IMPACTS OF TROPICAL LANDSCAPE CHANGE ON HUMAN DIET AND LOCAL FOOD SYSTEMS

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IMPACTS OF TROPICAL LANDSCAPE CHANGE ON HUMAN DIET AND LOCAL FOOD SYSTEMS

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Editorial: Impacts of Tropical Landscape Change on Human Diet and Local Food Systems

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Editorial on the Research Topic

Impacts of Tropical Landscape Change on Human Diet and Local Food Systems

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The impacts of changing diets on land use and land cover has been an important area of research in recent years (Foley et al., 2011; Tilman and Clark, 2014; Fanzo and Davis, 2019; Willett et al., 2019). This special issue looks at the reverse side of this relationship – how land use change affects the diets of local communities living in landscapes where change is taking place. Clear links between forest cover and diet and nutritional outcomes have been shown (Johnson et al., 2013; Ickowitz et al., 2014; Rasolofoson et al., 2018; Fisher et al., 2019), while more recent work has started to disentangle the differential impacts of land use type, composition and configuration on diets and the consumption of specific food groups (Rasmussen et al., 2019; Gergel et al., 2020). This special issue brings together a collection of papers that examine the effects of land use and land use change on diet and nutritional outcomes in the tropics. It assembles papers from a wide range of disciplines, covering the links between forest conservation, deforestation, hydropower development, and changing patterns of agricultural production on diets and nutrition across a range of settings.

Rasolofoson et al. use data from the Demographic and Health Surveys (DHS) to examine the effects of forests on nutritional status, particularly stunting, across 25 low and middle-income countries. The authors compare the prevalence of stunting for children with and without access to forest, with access being defined as living in communities within 3 km of the nearest forest edge and with at least 30% forest within a 5 km radius around the community center. They find that the percentage of stunted children among those with access to forest is 30.25%, while the stunting prevalence for children without access is 37.36%. The authors argue that access to forest significantly reduces child stunting (at least 7.11% points average reduction) – and that forest conservation therefore is a potentially effective nutrition-sensitive intervention.

Like Rasolofson et al. and Borgerson et al. also argue that forest conservation can address malnutrition, but they reach this conclusion through concerns about the sustainability of bush-meat hunting. In their study of 13 communities in Masoala National Park in Madagascar, they find high rates of food insecurity and malnutrition as well as high reliance on forests for food, particularly wild meat. They also find that although forests make important contributions to nutrient consumption, the extraction rates for wild meat are unsustainable. They thus advocate for conservation to reduce unsustainable hunting by helping communities to gain access to domestic sources of nutrient-rich foods.

Using data from 1,783 households across seven sites (in Bangladesh, Burkina Faso, Cameroon, Ethiopia, Indonesia, Nicaragua, and Zambia), Baudron et al. examine the pathways through which forests contribute to household dietary diversity and consumption of fruits, vegetables, and meat. Using piece-wise structural equation modeling, they compare the relative importance of a direct pathway (e.g., consumption of forest food), an income pathway (income from forest products used to purchase food from markets), and an agroecological pathway (forests and trees sustaining farm production). The results show major variation in the relationships between forest cover, pathways, and dietary outcomes across sites. Forest cover and dietary quality were positively related in some sites but negatively in others, and the importance of different pathways was also highly variable. The study highlights the significant variation in both the relationship between land use change and diets across settings, and the mechanisms that underly those relationships.

Rasmussen et al. examine how household wealth, on-farm production, and landscape context (forest cover and market access) are related to the dietary profiles of rural households in Ethiopia. Through cluster analysis of data from the World Bank's Living Standard Measurement Survey (LSMS), they identify three main household diet types: low diversity; high diversity rich in fruits and vegetables; and high diversity with increased consumption levels of oils, fats and sugars. The low diversity diet was mostly found among low- to middle-wealth households who farmed cereal grains. Households with diverse fruit-vegetable diets were most often engaged in coffee-agroforestry farming and tended to live in landscapes with higher forest cover. Finally, households with highly diverse oil-sugar diets tended to be wealthier and situated closer to roads. The study highlights the complex interactions among factors correlated with diverse diets and shows how even small increases in forest cover can increase dietary diversity and consumption of healthy foods.

While the previous papers look at how and under what conditions forests contribute to diets, Friant et al. and Acharya et al. investigate what happens to diets and nutrition when these forests are lost.

Friant et al. bring to light the ways in which dietary patterns differ across intermediate stages of deforestation and market integration in Cross River State in Nigeria. Using data on dietary diversity and food access collected from 528 households across six communities, they find that although forest-edge communities consumed less green leafy vegetables and less bushmeat than forest-interior communities, they consumed more dairy, eggs, beans, and other fruits and vegetables. Also, households from forest-edge communities exhibited significantly lower household food insecurity access scores. They conclude that in the intermediate stages of deforestation, communities may be able to get the "best of two worlds" with increased access to markets and continued access to forests.

Acharya et al. use Demographic and Health Survey data from 15 countries in Sub-Saharan Africa to explore the relationships between deforestation and the double burden of malnutrition. They find that forest cover loss is marginally associated with a higher probability of having an overweight woman and a stunted pre-school child in the same household, but not with having an overweight and anemic woman or an overweight and stunted child in the same household.

Golden et al. expands the focus of this collection of papers by examining dams as a unique form of land use change with the potential to have major impacts on human diets. Their paper examines the impact of dams on aquatic food resources in the Lower Mekong Basin (LMB), where over 100 dams are planned or in construction. Expanding on past modelling, they estimate that the loss of subsistence fish resources associated with dam building could greatly increase the number of people in the LMB who are at risk of protein, zinc, thiamine, niacin, calcium and iron deficiency.

Like Friant et al. and Sibhatu also finds some positive effects of land use change on diets. Sibhatu analyzes the impacts of oil palm adoption on about 700 households in Jambi, Indonesia over a 2-year period. He finds that oil palm adopters consumed more diverse foods at a household level than non-adopters and that they were less likely to be undernourished or to be micronutrient inadequate. This shows the potential positive effects of some land use changes; however, as Sibhatu himself (Sibhatu, 2020) and others have noted (Nurhasan et al., 2020), these findings may be very specific to the sample here in which both adopters and non-adopters of oil palm primarily cultivated plantation cash crops as opposed to food crops (Purwestri et al., 2019).

Finally, Sunderland and Vasquez address conservation of forests, warning against an overly protectionist stance that may have negative impacts on the food and nutrition security of local communities in forest-protected adjacent areas. After reviewing the many ways that forests contribute to the food and nutrition security of forest adjacent communities, they lay out some of the tensions between the conservation community and local people when protection of forests reduces access to forests for local communities that rely on them for their food and nutrition security. They call for greater integration and respect for the rights of local communities to access forests for food and a rights-based and participatory approach to conservation that emphasizes synergies between biodiversity conservation and food security.

As a collection, these papers have several implications for research and policy. Although previous papers examining large secondary data sets have shown fairly consistent relationships between forest cover, land use, and diet quality (Johnson et al., 2013; Ickowitz et al., 2014; Galway et al., 2018; Rasolofoson et al., 2018), this collection of new papers suggests that the impact of land use change on diet quality and food systems is heterogenous. The context-specific trajectories that explain these different results across sites remain very poorly understood and are fueling a dynamic area of research as several of these studies show. Understanding these complex relationships is imperative for designing policies that ensure peoples' access to sustainable sources of sufficient quantities of nutritious foods.

AUTHOR CONTRIBUTIONS

All authors contributed equally to this editorial and to editing the special issue.

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Oil Palm Boom and Farm Household Diets in the Tropics

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Farm households in the tropics are rapidly expanding oil palm monocultures, mainly at the expense of rainforests, agroforests, and traditional croplands. Although monetary gains and ecological consequences of such changes in land-use have been extensively documented, little is known about nutritional and dietary impacts on farm households despite those households being the most affected by nutritional problems. Here, this research gap is addressed with a 2-year panel data of farm households from Jambi province in Indonesia, a hot spot of tropical rainforest conversion into oil palm plantation. I use endogenous switching regression to better account for selection bias and obtain counterfactual outcomes. Results show high levels of undernourishment and micronutrient inadequacy in farm households in Jambi. Non-adopters are more likely to be undernourished and micronutrients deficient, consume less diverse foods, and eat low quantities of fruits and vegetables. The counterfactual analysis shows that oil palm adoption leads to significantly greater household dietary diversity, higher calorie consumption, more fruit and vegetable consumption, and higher food expenditure in farm households. These positive dietary impacts are observed irrespective of whether households belong to transmigrant or local communities. Panel regression results further show that oil palm cultivation reduces the prevalence of undernourishment and, at the same time, increases the mean probability of adequacy of consumed fruits and vegetables and micronutrients. This impact, leading to better diets, however, is complex and not straightforward; several socioeconomic, demographic, and farm factors have different dietary impacts for adopters and non-adopters. The findings highlight important policy implications: farm households adopt and expand land-uses that provide greater dietary benefits. Thus, policy-makers interested in maintaining the tropical rainforests, regulating oil palm plantations, and tackling nutritional deficiencies in the rural tropics should not overlook these dietary benefits for farm households.

Keywords: land-use change, oil palm, dietary diversity, food security, farm households, Indonesia, tropical rainforests

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INTRODUCTION

Changes in land-use have been altering ecosystems and livelihoods throughout history. Recent years have witnessed a massive expansion of oil palm monocultures in the tropics of Asia, mainly at the expense of rainforests, agroforests (timber and jungle rubber), and traditional crops such as rubber and rice (Daulay et al., 2016; Drescher et al., 2016; Byerlee et al., 2017). Although large-scale commercial plantations used to dominate this massive expansion, farm households are also adopting intensively managed oil palm monocultures at a high rate (Byerlee and Viswanathan, 2018). Current estimates show that smallholder farmers account for 40 to 50% of the total oil palm plantation areas (Euler et al., 2016; Byerlee et al., 2017). This crop's rapid expansion in farm households has fostered a growing body of literature focused mainly on the associated negative ecological impacts and socioeconomic implications for those households.

The negative ecological impact of farm households' oil palm expansion has been extensively documented. It is particularly associated with deforestation, ecosystem erosion, biodiversity loss, soil erosion, and greenhouse gas emission (Foster et al., 2011; Clough et al., 2016; Drescher et al., 2016). Mixed findings are reported on the socioeconomic implications. While some studies argue that oil palm expansion causes social conflicts on local communities (Overbeek et al., 2012; Krishna et al., 2017b; Hidayat et al., 2018), numerous other studies suggest that farm households' adoption of the crop contributes to alleviate poverty and improve households' income and living standards (Dewi et al., 2005; Sunderlin et al., 2008; Tscharntke et al., 2010; Euler et al., 2016, 2017). However, a closer look at those studies examining the impact of oil palm cultivation on growers' welfare reveals two critical shortcomings, strikingly vital for research and policy-making. First, the vast majority of these studies used cross-sectional data and expressed welfare benefits in terms of money and asset accumulation. Second, those studies have hardly looked at possible nutritional and dietary effects of oil palm adoption in farm households (Euler et al., 2017; Chrisendo et al., 2019). From a development-policy standpoint, however, longitudinal evidence and welfare analyses that go beyond monetary profits are critically important, particularly in rural areas of the tropics where the highest concentration of malnourished farm households are found. In this study, I contribute to the literature by addressing these two shortcomings.

In particular, using unique panel household survey data from the tropical rainforest areas of Indonesia and regression models that better account for selection bias, I examine the dietary impacts of oil palm adoption in farm households that have chosen to grow the crop in comparison to other farmers that do not. Examining the dietary impact of oil palm adoption in farm households of the tropics is important for several other reasons. First, despite significant progress in recent decades, nutritional deficiencies still pose serious problems in farm households; for instance, about 40% of the Indonesian population is affected by undernutrition and micronutrient malnutrition, and majority of the affected are farm households (Isabelle and Chan, 2011; FAO and WHO, 2014; Ickowitz et al., 2016). Second, tens of millions of farm households in the tropical areas continue to adopt oil palm (Byerlee et al., 2017). This increasing adoption by smallholder farmers is despite the crop, which yields the highest output per ha of all oil crops (FAO, 1990), requiring an expensive initial investment, managerial skills, and a switch to more capital-intensive farming practices (Euler et al., 2017). Third, oil palm cultivation has been seen as an opportunity for fighting against rural poverty and food insecurity in several Southeast Asian countries, including in Indonesia (Zen et al., 2005; FAO and WHO, 2014). Therefore, understanding how to make such an expensive agricultural strategy to be nutrition-sensitive and contribute to improving farm households' nutrition is vital for research and policy-making.

This study is based on a 2-year panel data from the Jambi province on the island of Sumatra. Indonesia is the largest oil palm—producing and—exporting country in the world (FAO, 2018), despite also boasting the highest deforestation rate (Margono et al., 2014). Jambi province is a study area of particular interest, having undergone a significant conversion of primary forests to oil palm plantations over the last few decades (Wilcove et al., 2013; Gatto et al., 2015). The province, like other rural areas in Indonesia, has high levels of underweight and stunted children, poor household dietary diversity, and pervasive micronutrient deficiencies (FAO and WHO, 2014). Additionally, the availability of a unique panel dataset of farm households from Jambi, as part of a larger interdisciplinary research project (Drescher et al., 2016), inspired the pursuit of this study. These data allow differentiating between adopters and non-adopters as well as to calculate various household-level dietary indicators, including dietary diversity scores, quantities of fruits and vegetables consumed, calories consumed, and the measures of food and micronutrient adequacies.

Rural markets in Indonesia are poorly developed (Ickowitz et al., 2016), but plantation farmers in Jambi hardly cultivate food crops for their own consumption (Sibhatu et al., 2015; Euler et al., 2017). This has significant implications in terms of food and nutrition security. Those plantation farmers heavily depend on agricultural cash income to purchase adequately diverse foods from such imperfect markets (Sibhatu et al., 2015; Sibhatu and Qaim, 2018), which consequently makes them vulnerable to substantial income and price shocks. Moreover, cultivating perennial and non-food commercial crops—that do not directly add to household dietary diversity through own consumption, are claimed to compete for resources (e.g., land) with other food crops that in turn negatively affects food availability and increase food prices (Li, 2015). Given these serious implications, I hypothesize that adopting oil palm worsens diversity and quality of diets, increases the prevalence of undernourishment, and aggravates micronutrient inadequacies in farm households. Whether oil palm adoption in farm households helps their diets meet the minimum adequacy level and contributes to reducing the prevalence of undernourishment is an empirical question that I also seek to answer with this study.

MATERIALS AND METHODS

Research Area

This study is conducted in the lowlands of Jambi province, Sumatra, Indonesia, as part of the CRC990/EFForTS investigating ecological and socioeconomic changes associated with the transformation of lowland rainforest into agricultural systems (Drescher et al., 2016). In this subsection, I briefly discuss the socio-environmental and agricultural nature in Jambi in order to set the background for the study. Jambi province is home to diverse ethnic groups and multiple indigenous languages and dialects. It is part of the tropical areas historically covered by rainforests (Foster et al., 2011; Clough et al., 2016).

Forest reserves and national parks in Jambi currently house some endangered wildlife, among them, the Sumatran orangutan, tiger, and elephant (Luskin et al., 2014). The humid tropical climate in the province is conducive to rich biodiversity and plantation crops, such as timber, rubber, and oil palm.

History of deforestation and logging in Jambi and other regions of Indonesia date back over a century, with the cultivation of timber and rubber and, more recently, with a massive expansion of oil palm monoculture. Between 1990 and 2010, commercial plantation expansion, mining, and logging activities had caused a loss of 2.65 million ha of primary forest in Jambi, approximately 43% of its total forest area (Margono et al., 2014).

Agroforestry (timber and jungle rubber) and commercial plantations (rubber and oil palm) are the most important income sources for the population of Jambi. Rubber was the most dominant crop before being recently overtaken by oil palm (**Figure 1**). The areas under rubber decreased by half since 2010. Contrarily, oil palm plantation, first introduced in the region through government support programs, kept expanding. In the 1980s, farm households started to cultivate oil palm with subsidized contract programs (Rist et al., 2010; Gatto et al., 2017). While subsidization stopped after 1999, independent adoption continued to grow steadily (Susanti and Budidarsono, 2014; Euler et al., 2017). Currently, Jambi is the sixth top producer of crude palm oil in Indonesia, with more than 200,000 smallholder farm households and ∼700,000 ha of oil palm plantation (FAO, 2018). Important to note is that some forest cover is still available in Jambi that could be converted into oil palm plantation at any time. Thus, understanding why farm households chose to adopt oil palm is of paramount importance to promote environment-friendly production practices and improve the livelihoods of farmers and rural laborers in Jambi and similar areas worldwide.

Panel Household Survey

To examine the dietary impacts of oil palm adoption in farm households, the analysis draws upon a 2-year panel data from Jambi province. The two farm household surveys were implemented during the dry seasons in 2012 and 2015. A multi-stage random-sampling approach was applied in order to select a representative sample. In the first stage, all regencies covering the province's tropical lowland areas, namely Batanghari, Bungo, Muaro Jambi, Sarolangun, and Tebo, were purposely selected. From these regencies, 20 districts were selected randomly. Likewise, in each district, two villages were selected randomly; that is, 40 villages in total.

Additionally, five other villages were included in the sample to cover some areas where other CRC990/EFForTS' subprojects had conducted ecological experiments. At the last sampling stage, between 6 and 25 farm households from each village (depending on the village population size) were randomly selected. In total, 700 households were selected and interviewed. Only those households owning agricultural land in the last 5 years were included since oil palm experiences yield-delay for about 4 years between establishing new plantation and harvesting the first fruit bunches. Each household interviewed in 2012 was revisited in 2015. The attrition rate between the two surveys was 6% (41 observations) because of outmigration (56%), refusal to be re-interviewed (24%), and the death or old age of respondents (20%). Moreover, a few households were dropped due to missing data. To reduce the effect of attrition, those farmers who were not available in 2015 were replaced with other households from the same village that was also randomly selected.

The sample households are relatively specialized farmers in plantation crops, either rubber, oil palm, or both. Some of the respondents produce food crops like rice and maize. Moreover, few of them also grow horticultural crops, rear livestock (mainly chicken) and supplement with aquaculture production.

Data were collected using a structured questionnaire and through face-to-face interviews by carefully-trained enumerators in Bahasa Indonesia. Pre-testing was carried out to assess the questionnaire's clarity. The panel dataset contains a wide range of information, including household demographics, socioeconomic characteristics, farm activities, and food and non-food consumption. The food consumption section includes detailed information related to the type and quantity of consumed foods over the past week. While most of the data were collected by interviewing the household head or the spouse, the information about food consumption was collected by interviewing the persons responsible for buying and preparing food, often the wife or an adult daughter. In total, detailed quantitative information of 120 food items was collected, which allowed for the calculation of the household dietary indicators explained in the following sub-section.

Dietary Indicators

The main aim of this study is to examine the effects of oil palm adoption on household dietary outcomes, for which I use household dietary diversity (HDDS), daily consumption of calories, daily consumption fruits and vegetables; and annual food expenditure per adult equivalent (AE) as the primary outcome variables. To better understand the implications for food and nutrition security, I also use dichotomous dependent variables, indicating whether household diets met the minimum adequacy level of consumed fruits and vegetables, calories, as well as iron, zinc, vitamin A, and an average of these three micronutrients (iron, zinc, and vitamin A). I focus on these three micronutrients since their deficiency affects millions of people, particularly women and children (Black, 2014). All dietary indicators are derived from a quantitative 7-day recall of food intake that has already been consumed by the household members. Food waste and foods consumed outside the home are not included in the calculation of the dietary outcomes. Recent studies have shown household dietary indicators based on a quantitative 7-day recall period of food consumption are strongly correlated with individual-level indicators constructed on 24 h recall period (Sununtnasuk and Fiedler, 2017; Fongar et al., 2019).

Diet diversity is often used to indicate food security in terms of both availability and access (Ruel, 2003). I calculate a nine food groups of HDDS, based on those used for the Minimum Dietary Diversity for Women and which contribute strongly to micronutrient adequacy (Martin-Prével et al., 2015; FAO and FHI 360, 2016). Food groups that have little or undesirable

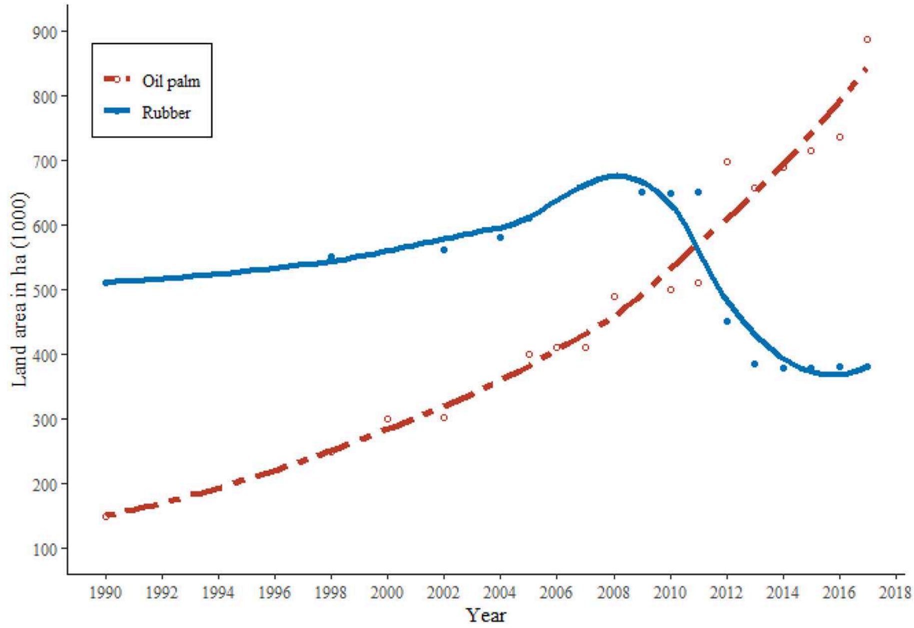

FIGURE 1 | Rubber and oil palm plantation areas in Jambi province, between 1990 and 2017. Author's presentation based on data taken from the Tree Crop Statistics of Indonesia yearbooks (BPS, 2017, 2018; DJP, 2017).

nutritional and health effects when consumed in large quantities are excluded (sugars and sweets, oils and fats, and condiments) from HDDS. Food consumption expressed in terms of calories is also a standard indicator used to assess food availability at the household-level (de Haen et al., 2011; Frelat et al., 2016; Sibhatu and Qaim, 2017). Using Food Composition Tables in Indonesia (Berger et al., 2013), the quantities of food items consumed are converted into calories. The USDA National Nutrient Database for Standard Reference (USDA, 2005) is used for a few food items not listed in the Indonesia Food Composition Table. The quantities of calories are adjusted by AE to account for differences in household demographic structure, such as age and gender. Besides, annual food expenditure is used as an additional outcome variable, as it is also one of the commonly used indicators of food security (de Haen et al., 2011; Lo et al., 2012). Annual food expenditure is also adjusted by the consumer price index for Indonesia across survey rounds and by AE. Alcohol, drinking water, and condiments are omitted in the food expenditure calculation.

In the same procedure that I calculate calories, quantities of the food items consumed are first converted into equivalent micronutrient consummation per AE. After that, the households are divided into two groups of food and micronutrient adequacy status, based on the estimated daily energy, fruits and vegetables, and micronutrient requirements of a male adult with normal physical activity. Households are classified as food abundant if the quantities of calories they consumed are above the estimated minimum threshold of 2,400 kcal per AE a day (FAO, 2001). For fruit and vegetables as well as for micronutrients consumed, the internationally recommended quantities are 400 g for fruits and vegetables, 18 mg for iron, 15 mg for zinc, and 625 μg retinol equivalents for vitamin A per AE a day (FAO et al., 2004). Following Hatløy et al. (1998), adequacy is expressed for each dietary indicator in terms of a dummy variable, being "*1*" if actual consumption is equal to or above the recommended level, and being "*0*" otherwise. Moreover, mean micronutrient adequacy is calculated for each household by averaging the adequacy indicators for iron, zinc, and vitamin A.

Covariates

Several other factors may influence the quality and diversity of diets in smallholder farm households. Moreover, the dietary impact pathways of oil palm cultivation might depend on the mediation of socioeconomic, cultural, farm, and demographic factors (Chrisendo et al., 2019). I estimate all regression models (see Econometric Analysis subsection) including farm, socioeconomic and demographic characteristics, such as farm and household sizes, credit access, as well as age, ethnicity, and the education level of the household head as covariates. These covariates may influence not only the adequacy of diets and micronutrients, but also the choice of what and how much to produce, and would hence mediate the impact of oil palm cultivation on household diets.

Econometric Analysis

In this subsection, I present the econometric models estimated in this paper, namely the endogenous binary switching regression. A simple linear panel estimation that assumes a set of explanatory variables have the same impact on adopters and non-adopters may not be appropriate in this study for at least two reasons. First, many of the oil palm farmers—in the dataset that I use in this study—acquired the plantations through a transmigration program (Euler et al., 2017). This implies that sample selection bias is possible in this dataset, and thus, several of the covariates might have different dietary impacts to adopters and non-adopters (Krishna et al., 2017a). Second, for the economic significance of the estimated coefficients, a challenge is that observed and unobserved factors might affect oil palm cultivation and household diets simultaneously, such as risk preferences, entrepreneurship and farm management knowledge, and skills. Thus, identifying the impacts of oil palm adoption on household diets requires dealing with such observable and unobservable characteristics as well as controlling for selection bias. To deal with these two methodological issues, I apply endogenous binary switching regression. Endogenous binary switching regression in a counterfactual framework allows identifying the effects on dietary outcomes in each of the two groups, instead of pooling the adopters and non-adopters in one regression model.

Modeling of the effects of oil palm adoption on dietary indicators under the endogenous specification framework is applied in two stages. First, the decision to adopt oil palm (adoption equation) is estimated using a probit model, in which the equation is specified as

$$OP_i^* = Z_i\beta + \varepsilon_i \quad with\ OP_i = \begin{cases} 1 & if\ OP_i^* > 0 \\ 0 & if\ otherwise \end{cases} \tag{1}$$

where OP_i^* denotes a latent variable for household i's adoption of oil palm; $OP_i^* = 1$ if a household cultivates oil palm and $OP_i^* = 0$ otherwise. β is a vector of parameters to be estimated; Z_i is a vector of control variables explaining the possibility of being an oil palm household. ε_i denotes the random error.

Second, panel linear regressions (outcome equations), with selection bias correction and conditional on adoption decision, are used to examine the relationship between the dietary indicators and a set of explanatory variables (Equations 2 and 3). Specifically, the outcome regression equations are specified in two separate equations—one for oil palm adopters and another for the non-adopters, and the equations estimated are such that:

$$N_{ai} = X_{ai}\beta_a + \nu_{ai}, \quad if\ OP_i = 1 \tag{2}$$

$$N_{ni} = X_{ni}\beta_n + \nu_{ni}, \quad if\ OP_i = 0 \tag{3}$$

where subscript a and n denote adopters and non-adopters, respectively. N represents the household diet indicators (i.e., HDDS, consumption of calories, grams of fruits and vegetables, and annual food expenditure). N_{ai}, refers to adopters (treatment group) and N_{ni}, to non-adopters (control group). β_a and β_n are vectors of parameters to be estimated. The error terms are ν_a and ν_n.

Equation (1), the adoption equation, decides which of the two types of groups (adopters or non-adopters) is applicable. Equations (2) and (3) describe the variables of concern (dietary indicators) in each of the two groups. The error terms ν_{ai}, ν_{ni}, and ε_i are assumed to have a trivariate normal distribution with a mean of zero. In particular, as the error term ε_i in Equation (1) is correlated with the error terms in Equations (2) and (3), the expected values of ν_{ai} and ν_{ni} conditional on the sample selection should be non-zero (Lokshin and Sajaia, 2004; Wooldridge, 2010). In simple words, statistically significant and non-zero coefficients of correlation of the error terms indicate that there was selection bias in adopting oil palm in Jambi; otherwise, no sample selection bias.

TABLE 1 | Conditional expectations and treatment effects.

Household type	Adopters	Non-adopters	Treatment effect
Adopters (A)	(a) $E(N_a \mid OP = 1$	(c) $E(N_n) \mid OP = 1$	$A_sN = (a - c)\ AAT$
Non-adopters (NA)	(d) $E(N_a \mid OP = 0$	(b) $E(N_n) \mid OP = 0$	$NA_sN = (d - b) ATU$

Cells (a) and (b) denote the diet indicators that are observed in a sample; cells (c) and (d) denote the counterfactual diet indicators.
E is the expected operator.
$OP = 1$ if the household is an adopter of oil palm; $OP = 0$ if the household is non-adopter of oil palm.
N_a is diet indicators for adopters; N_n = dietary indicators for non-adopters.
A_sN and NA_sN denote the expected diet indicators (N) effects of oil palm adoption for those households randomly chosen from the adopters and non-adopters, respectively.
ATT is the average treatment effect on the treated; ATU is the average treatment effect on the untreated.

The endogenous binary switching regression is estimated using full information maximum likelihood estimation. And for a maximum likelihood estimation to be robust, exclusion restrictions should be applied (Di Falco et al., 2011). I use altitude above sea level of the household residence as a selection instrument based on a falsification test. I use altitude above sea level as a selection instrument because, in the low altitudes of Jambi, altitude was found to be affecting oil palm adoption, but not household income and consumption expenditure (Krishna et al., 2017a). A variable (altitude above sea level of the household residence) is considered a valid selection instrument if it affects the decision to implement a particular farming system (statistically significant coefficients for oil palm adoption) but does not affect coefficients in outcome equation (statistically insignificant coefficients of the dietary indicators) (Di Falco et al., 2011).

In order to estimate and compare the impact of growing oil palm on the dietary outcomes of the adopters and non-adopters, I also use the endogenous switching regression model to obtain counterfactual dietary outcomes of each group. Estimating the counterfactual dietary outcomes enables to estimate the dietary indicators of non-adopters if they had adopted oil palm, or to estimate the dietary outcomes of the adopters if they had dis-adopted oil palm. Put differently, what the dietary status of the non-adopters would have been if their characteristics (coefficients the explanatory variables) had been the same as the adopters' characteristics, and vice versa. I follow Carter and Milon (2005) to compute the actual and counterfactual dietary outcomes of the adopters and non-adopters presented in **Table 1**.

Boxes (a) and (b) refer to the observed dietary outcomes (N) for adopters and non-adopters, respectively, while boxes (c) and (d) refer to the counterfactual dietary indicators. If the non-adopters had adopted oil palm, then the expected effect of oil palm adoption on non-adopters' dietary outcomes (NA_sN) would have been the difference between (d) and (b). Likewise, the expected effect of oil palm adoption on the adopters' dietary outcomes (A_sN) would have been the difference between (a)

and (c)—had they had dis-adopted oil palm. In econometric terms, the A_sN and NA_sN are equivalent to the average treatment effect on the treated (ATT) and average treatment effect on the untreated (ATU), respectively (Heckman et al., 2001). In other words, ATT refers to the estimated effect of oil palm adoption on the adopters, while ATU refers to the possible effect of oil palm adoption on the non-adopters.

Finally, I conduct further panel regression analyses to understand the impact of oil palm adoption on the dichotomous variables of households' diets indicating whether these diets met the minimum adequacy level of consumed fruits and vegetables, calories, iron, zinc, vitamin A, and an average of these three micronutrients. As these outcome variables are binary, panel logit regression is used for the estimations.

RESULTS AND DISCUSSIONS

In this section, I present the key findings. First, I explore the descriptive results. I then focus on the econometric estimation results in the following subsections.

Descriptive Characteristics

Table 2 presents descriptive statistics for the sample households, disaggregated by year and adoption. The adopters and non-adopters are similar in terms of demographic characteristics, including household size, age, and education. However, there is a significant difference in terms of farm characteristics between the two groups. Adopters cultivate significantly bigger farms with a higher proportion of their land having a formal title than non-adopters. Conversely, non-adopters are less likely to access formal credit services, probably since a significant portion of their land does not have a clear title. In terms of ethnicity, the Melayu—the largest local ethnic group in Jambi—account for the majority of the non-adopters. This is probably linked to their tradition of rubber cultivation (Euler et al., 2017). Adopters own more non-farm businesses (like cafes, small shops, and motorbike repair shops) than non-adopters in 2015. As there is no significant difference in off-farm activities between the adopters and non-adopters, I use the "own business" variable as a proxy for off-farm activities in the regression estimations, which I describe in more detail below.

Table 2 also displays summary statistics of the dietary indicators, outcome variables of interest in this study. A significant difference between adopters and non-adopters is observed for these outcomes variables. On overage, adopters consume significantly more diverse foods, particularly more fruits and vegetables. Approximately, 61 and 51% of non-adopters consumed more calories than the recommended 2,400 kcal/AE a day in 2012 and 2015 respectively, whereas 71 and 67% of the adopters consumed more than the recommended quantities of calories in 2012 and 2015 respectively. In other words, 39 and 49% of non-adopters and 29 and 33% of the adopters in 2012 and 2015 are classified as undernourished, respectively. In 2015, 82% adopters and 92% non-adopters consumed less zinc than the recommended amount on average. Furthermore, about 57% adopters and 69% non-adopters consumed less iron than the recommended amount in 2015. Similar results are observed in vitamin A consumption. In sum, these findings on dietary outcomes suggest two valuable lessons. First, there is a high prevalence of undernourishment and micronutrient inadequacy in Jambi, which is similar to the national average (see the Introduction section). Second, between the two types of households, the non-adopters are more likely to be undernourished, consume less diverse foods, and consume inadequate fruits and vegetables and micronutrients on average.

Moreover, **Figure 2** depicts the food groups consumed by all respondents over the previous 7-day period. The farmers' diet is mainly composed of cereals and starchy staples, meat and fish, nuts and seeds, and eggs. Organ meat, dairy products, and vitamin A-rich fruits and vegetables are relatively the least consumed groups.

Finally, over the 3 years, the number of oil palm adopters rose by 10% (from 248 to 272). Likewise, the proportion of titled land, access to formal credit, owning non-farm businesses increased in the two groups of adopters and non-adopters, Access to off-farm activities also slightly rose from 2012 to 2015, although no significant difference is observed between adopters and non-adopters. Strikingly, all indicators of household diets were lower in 2015 than in 2012 in both groups. This is because of the global drop in the prices of oil palm and rubber after 2012 (Kubitza et al., 2018).

Factors Affecting Oil Palm Adoption and Household Diets

I now discuss the econometric results, starting with findings from the endogenous switching regression model estimation. The results are presented in **Tables 3**, **4**, and in Tables A1–A3 of the **Supplementary Material**. The coefficients of correlation ($\boldsymbol{\Sigma_a}$ and $\boldsymbol{\Sigma_n}$), which are displayed in the lower part of **Table 3**, are significantly different from zero between the adoption (Equation 1) and outcome equations (Equations 2 and 3) in most of the model estimations. This confirms that there is self-selection in adopting oil palm in Jambi, supporting the notion that a panel linear regression estimation is not appropriate for the dataset at hand. Furthermore, the Wald test on the exclusion restriction of the variable "Altitude" is jointly significant (**Table 3**). Simultaneously, altitude does not have a statistically significant effect on all diet indicators of the adopters (Table A1 in the **Supplementary Material**). These results confirm that the falsification test is statistically valid and the endogenous switching regression estimation provides robust results.

Columns (1), (4), (7), and (10) of **Table 3**, which are the estimates from the adoption model explained in Equation (1), show that ethnicity, size of cultivated land, the proportion of titled land, and access to credit services influence oil palm adoption significantly. Ethnically, being a Melayu is negatively and significantly related to oil palm adoption. As mentioned earlier, the Melayu are the local people with the tradition of cultivating rubber, and thus less likely to switch to oil palm cultivation. Clearing rubber and setting up an oil palm plantation is also quite costly. Owning a larger cultivated area and a larger proportion of formally titled land positively and significantly influences the adoption of oil palm. This is expected and

TABLE 2 | Descriptive differences between adopters and non-adopters of oil palm by year.

	2012			2015		
	Adopters	Non-adopters	Difference	Adopters	Non-adopters	Difference
EXPLANATORY VARIABLES						
Age of household head (years)	45.19 (0.77)	45.62 (0.59)	−0.42 (0.97)	47.57 (0.67)	47.43 (0.057)	0.14 (0.88)
Education level of household head (years)	7.88 (0.22)	7.29 (0.18)	0.59** (0.28)	7.44 (0.21)	7.15 (0.18)	0.29 (0.28)
Household owns business (dummy)	0.24	0.19	0.05	0.35	0.22	0.13***
Off-farm income (1 = yes; 0 = otherwise)	0.61	0.60	0.01	0.73	0.69	0.05
Ethnicity: (1 = Melayu; 0 = others)	0.40	0.55	−0.16***	0.43	0.53	−0.10**
Cultivated land area (ha)	5.62 (0.43)	3.48 (0.23)	2.14*** (0.48)	6.74 (0.69)	3.43 (0.25)	3.31*** (0.74)
Share of titled land (%)	0.50 (0.03)	0.32 (0.02)	0.19*** (0.04)	0.54 (0.03)	0.37 (0.02)	0.17*** (0.03)
Migrant: (1 = transmigrant; 0 = otherwise)	0.44	0.24	0.20***	0.39	0.25	0.14***
Credit from formal source (dummy)	0.35	0.18	0.17***	0.47	0.28	0.19***
Altitude above sea level (meter)	50.58 (1.58)	57.23 (1.34)	−6.65*** (2.07)	47.51 (1.33)	56.80 (1.31)	−9.29*** (1.87)
Household size (number)	4.19 (0.09)	4.21 (0.07)	−0.03 (0.12)	4.15 (0.09)	4.11 (0.08)	0.04 (0.12)
OUTCOME VARIABLES						
Household dietary diversity score (HDDS; 9 food groups)	6.96 (0.07)	6.55 (0.06)	0.41*** (0.10)	6.94 (0.07)	6.57 (0.06)	0.36*** (0.10)
Fruits and vegetables consumed per day (grams/AE)	679.16 (28.88)	541.68 (15.42)	137.49*** (29.86)	400.25 (20.60)	308.35 (12.82)	91.90*** (22.99)
Calorie adequacy (dummy; ≥2,400 kcal/AE/day)	0.71	0.61	0.10***	0.67	0.51	0.16***
Fruits and vegetables adequacy (dummy; ≥400 g/AE)	0.75	0.61	0.11***	0.37	0.24	0.13***
Iron adequacy (dummy; ≥18 mg/AE)	0.57	0.39	0.18***	0.43	0.31	0.12***
Zinc adequacy (dummy; ≥15 mg/AE/day)	0.21	0.11	0.10***	0.18	0.08	0.10***
Vitamin A adequacy (≥625 ug RE/AE/day)	0.56	0.41	0.15***	0.48	0.35	0.13***
Mean adequacy of iron, zinc and vitamin A (dummy)	0.44	0.30	0.14***	0.36	0.25	0.12***
Food expenditure per year (000 IDR/AE)	7523.43 (285.38)	6262.68 (186.77)	1260.75*** (328.30)	8202.027 (290.27)	6489.72 (161.12)	1712.31*** (307.06)
Observations	248	440		272	431	

*Mean values with standard errors in parenthesis are reported. The t-test is conducted on mean differences. AE, adult equivalent; RE, retinol equivalent. Food expenditure is adjusted by the consumer price index for Indonesia across survey rounds. **, *** denote significance at 5, and 1% levels, respectively.*

plausible, since a larger land size, combined with a formal land title, is vital for long-term agricultural investments such as in oil palm. Having clear land title also allows farmers to use their land as collateral to access rural financial markets or to diversify their off-farm livelihood systems. As one would expect, there exists a positive and significant effect of formal credit access on oil palm adoption, with about 54% of the cultivated land in oil palm farmers having clear property rights, as compared to the 37% of non-adopters in 2015 (**Table 2**).

Now, I discuss the results from the outcomes equation. The estimated effects of the socioeconomic factors on the four outcomes variables are presented in **Table 3**. In general, the results show that the sign and magnitude coefficients of the covariates are different for the adopters and non-adopters. Some of the covariates that explain the dietary outcomes in the adopters also do not explain that of the non-adopters, and vice versa. This also supports that there was selection bias in the dataset that I use in this study. Moreover, within the dietary indicators, some of the covariates that influence the HDDS, for example, either do not significantly affect or significantly affect calorie consumption and food expenditure in the opposite direction. Thus, this effect difference within the dietary indicators should be taken into consideration when one aims to improve nutrition and food security in vulnerable farm households.

More specifically, unlike in the adopters (Column 2), HDDS of the non-adopters is positively affected by household size, owning a non-farm business, larger cultivated area, and a higher proportion of titled land (Column 3). This is probably implying that nutrition education and awareness are important to improve the diversity of diets in the non-adopters compared to the non-adopters. The results in column (5) show that owning a non-farm business positively and significantly affects the adopters' fruit and vegetable consumption, but no significant effect is observed in the non-adopters. Column (6) shows that household size, taking

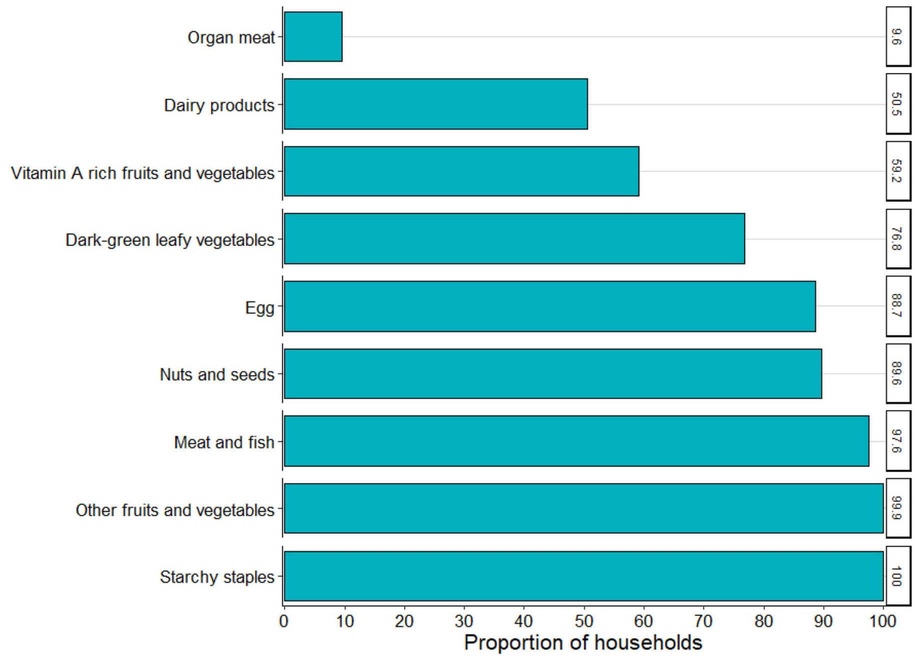

FIGURE 2 | The proportion of households who consumed each food group, 2012 and 2015 combined.

formal credit, a larger cultivated area, and a higher proportion of titled land influences the non-adopters' fruits and vegetables consumed both positively and significantly.

Column (8) shows that owning a non-farm business and a larger cultivated land area seems to positively and significantly affect the consumption of calories in the adopters, possibly suggesting that off-farm income indeed contributes to better household diets. Likewise, column (9) shows that education level of household head, size of cultivated land, and proportion of titled land have a positive and significant effect on the consumption of fruits and vegetables in the non-adopters. As columns (11) and (12) depict, the same set of factors that affect calorie consumption are observed to affect food expenditure to the same effect direction. Important to note is that in both groups, a larger household size affects calorie consumption and food expenditure negatively and significantly. This is to be expected since a family with more mouths to feed is more likely to be with a larger number of economically inactive members, yet spend more on food. Factors related to demographic characters seem to explain the dietary outcomes in the non-adopters than adopters; for example, age and education positively affect food expenditure in the non-adopters, but no significant effect is observed in the adopters.

The year dummy shows that there has been a significant reduction in terms of diversity, calories, the quantities of fruits and vegetables consumed, and food expenditure. As described earlier, this is possibly linked to the global drop in prices of oil palm and rubber. Therefore, the diets of plantation farmers are highly vulnerable to price shocks and the resultant decrease in income. Finally, it is important to stress that education significantly affects most of the dietary indicators and food expenditure, particularly in the non-adopters. This implies that the quality and diversity of these smallholders could be promoted through both education and awareness training.

Impact of Oil Palm Adoption on Household Diets

The next logical question, important for policy implications, is how the availability and diversity of food, quantities of fruits and vegetables consumed, and food expenditure would change if the non-adopters adopt oil palm or vice versa? This leads to the presentation of results from the counterfactual analyses. I have compared the actual dietary indicators of the oil palm adopters to the counterfactual dietary indicators of those farmers if they were non-adopters (ATT). Similarly, I have compared the actual dietary indicators of non-adopters with their counterfactual dietary indicators had they been cultivating oil palm (ATU). The dietary indicators are log transformed to be normally distributed and enable easy comparisons among the indicators. The results are presented in **Table 4**. Since the dietary indicators are in logarithmic form, I have interpreted the results in percentages. On average, I find that oil palm adopters would have fallen in calories, dietary diversity, quantities of fruits and vegetables consumed and food expenditure, if they become non-adopters by 11.0, 12.6, 16.5, and 51.3%, respectively. In contrast, the diets of non-adopters would have been higher than now if they had adopted oil palm. Specifically, had the non-adopters adopted oil palm, the diversity of their diets would have increased by 51.2 %, and the daily quantities of fruits and vegetables consumed by 66.7%, the daily calorie consumption by 31.4%, and annual food expenditure by 45.3%. Generally, these results show that non-adopters can significantly improve their diets in terms of

TABLE 3 | Binary switching regression for oil palm adoption and impact on household diets and food expenditure.

	Log HDDS (9 food groups)			Log fruits and Vegetables (g/AE)			Log calorie (kcal/AE)			Log food expenditure per AE		
		Outcome			Outcome			Outcome			Outcome	
	Selection (1)	Adopter (2)	Non-adopters (3)	Selection (4)	Adopters (5)	Non-adopters (6)	Selection (7)	Adopters (8)	Non-adopters (9)	Selection (10)	Adopters (11)	Non-adopters (12)
HH size	0.023	0.008	0.014***	0.026	0.009	0.030*	0.027	−0.085***	−0.045***	0.030	−0.097***	−0.056***
	(0.023)	(0.007)	(0.005)	(0.024)	(0.022)	(0.016)	(0.023)	(0.012)	(0.008)	(0.024)	(0.013)	(0.010)
Age of HH head	−0.001	−0.000	−0.001	−0.002	−0.000	0.002	−0.003	0.001	0.002	−0.003	−0.000	0.003**
	(0.003)	(0.001)	(0.001)	(0.003)	(0.003)	(0.002)	(0.003)	(0.002)	(0.001)	(0.003)	(0.002)	(0.001)
Education of HH head	0.007	0.004	0.009***	0.010	0.010	0.029***	0.010	−0.003	0.015***	0.011	0.003	0.023***
	(0.010)	(0.003)	(0.002)	(0.011)	(0.010)	(0.006)	(0.011)	(0.005)	(0.004)	(0.011)	(0.006)	(0.004)
HH owns business	0.096	0.005	0.029*	0.135	0.184**	0.011	0.129	0.100**	0.046	0.133	0.126***	0.101**
	(0.079)	(0.021)	(0.017)	(0.086)	(0.077)	(0.059)	(0.086)	(0.039)	(0.030)	(0.086)	(0.045)	(0.042)
Ethnicity: (1 = Melayu)	−0.147*	0.002	−0.033**	−0.186**	−0.078	0.007	−0.192**	0.011	−0.009	−0.178**	−0.035	−0.056
	(0.078)	(0.022)	(0.015)	(0.083)	(0.073)	(0.049)	(0.083)	(0.040)	(0.027)	(0.083)	(0.044)	(0.035)
Cultivated land area	0.038***	−0.003**	0.002	0.043***	0.004	0.020***	0.043***	0.005***	0.016***	0.045***	0.009***	0.011
	(0.010)	(0.001)	(0.001)	(0.012)	(0.004)	(0.005)	(0.012)	(0.002)	(0.004)	(0.013)	(0.003)	(0.007)
Titled land (%)	0.342***	−0.059**	0.032*	0.394***	−0.000	0.104*	0.388***	0.051	0.070**	0.394***	0.005	0.052
	(0.085)	(0.025)	(0.017)	(0.088)	(0.104)	(0.063)	(0.089)	(0.046)	(0.033)	(0.087)	(0.049)	(0.055)
Credit from formal source	0.408***	−0.093***	−0.007	0.432***	−0.115	0.165***	0.438***	−0.063	0.055*	0.427***	−0.073	−0.003
	(0.075)	(0.025)	(0.017)	(0.081)	(0.087)	(0.063)	(0.081)	(0.041)	(0.034)	(0.084)	(0.046)	(0.060)
Altitude	−0.005***			−0.009***			−0.009***			−0.008***		
	(0.001)			(0.002)			(0.001)			(0.002)		
Dummy 2015	−0.017	0.006	0.003	−0.023	−0.660***	−0.748***	−0.032	−0.092***	−0.104***	−0.029	0.074*	0.037
	(0.069)	(0.019)	(0.014)	(0.076)	(0.061)	(0.045)	(0.073)	(0.035)	(0.024)	(0.073)	(0.039)	(0.030)
Constant	−0.591**	2.202***	1.744***	−0.355	6.543***	5.558***	−0.323	8.443***	7.792***	−0.415	9.344***	8.374***
	(0.244)	(0.069)	(0.046)	(0.250)	(0.328)	(0.143)	(0.248)	(0.148)	(0.077)	(0.268)	(0.158)	(0.112)
Σa		0.269***			0.698***			0.399			0.441***	
		(0.022)			(0.056)			0.022			(0.024)	
Σn			0.208***			0.655***			0.363***			0.465***
			(0.007)			(0.022)			(0.016)			(0.030)
ρa		−0.958***			−0.432**			−0.393**			−0.433***	
		(0.020)			(0.209)			0.156			(0.147)	
ρn			−0.246***			−0.214			−0.320			−0.652***
			(0.046)			(0.148)			(0.211)			(0.129)
Wald test on exclusion restriction variable altitude:												
		$\chi 2 = 0.000$***			$\chi 2 = 0.071$*			$\chi 2 = 0.048$**			$\chi 2 = 0.044$**	
Observation		1378			1378			1378			1378	

*Coefficients are shown with standard errors in parentheses. HDDS, household dietary diversity score; HH, household. Subscript a and n denote adopters and non-adopters, respectively. *, **, *** denote significance at 10, 5, and 1% levels, respectively.*

TABLE 4 | Treatment effects of oil palm adoption.

Variables	Household type	Adopters	Non-adopters	Treatment effects	
Log HDDS (9 food groups)	Adopters	1.925	1.820	0.126***	ATT
		(0.002)	(0.003)	(0.002)	
	Non-adopters	2.374	1.862	0.512***	ATU
		(0.001)	(0.002)	(0.001)	
Log fruits and vegetables (g/AE/day)	Adopters	6.025	5.856	0.168***	ATT
		(0.016)	(0.020)	(0.009)	
	Non-adopters	6.460	5.793	0.667***	ATU
		(0.012)	(0.014)	(0.005)	
Log calorie (kcal/AE/day)	Adopters	8.005	7.895	0.110***	ATT
		(0.007)	(0.009)	(0.006)	
	Non-adopters	8.164	7.850	0.314***	ATU
		(0.005)	(0.005)	(0.004)	
Log food expenditure (000 IDR/AE/year)	Adopters	8.858	8.345	0.513***	ATT
		(0.009)	(0.010)	(0.005)	
	Non-adopters	9.100	8.647	0.453***	ATU
		(0.006)	(0.007)	(0.004)	

*Standard errors in parenthesis. *** denote significance at 1%, based on t-tests. HDDS, household dietary diversity score; IDR, Indonesian rupiah; ATT, the average treatment effect on the treated; ATU, the average treatment effect on the untreated.*

diversity, availability, and the consumption of micronutrients if they adopt oil palm.

Robustness Checks and Dealing With Endogeneity

As mentioned above, many of the adopters received oil palm plantations as part of the package of the transmigration program. This might bias impact estimates if not considered. Since the sampling strategy is random at the district, village, and household levels, and since I also use an econometric strategy that controls selection bias, this should not be a concern. However, as a robustness check, I have re-estimated all models by restricting the sample solely to the local communities. That is, I have excluded the transmigrant households from the estimation models. Results for the exclusion restriction test are presented in Table A1 in the **Supplementary Material**, while results from the switching regression estimation in a counterfactual framework are shown in Tables A2 and A3 in the **Supplementary Material**. The results tell a similar story as to those estimates in **Tables 3**, **4**. Indeed, regardless of a household's background, being from transmigrant or local communities, adopting oil palm improves household diets significantly.

Moreover, in the analyses so far I have not considered whether previous income (the wealth status of a farmer before adopting oil palm) affects the dietary benefits and oil palm adoption simultaneously; so it not yet clear whether the observed effects are actually due to oil palm adoption. In econometric terms, is there an endogeneity (reverse causality) problem in the data I use in this study? In order to test whether previous income status changes the findings, one needs to have income information of the two groups of adopters and non-adopters before 2012, that is, before the first wave of the dataset used in this study was collected. Unfortunately, such information is not available in this dataset. However, the CRC 990/EFForTS project is still tracking the sample households; and out of the 440 households that were non-adopters in 2012, 83 households have so far adopted oil palm [24 households adopted between 2012 (as mentioned earlier) and 2015 and another 59 households between 2015 and 2018]. I have therefore examined whether the average annual income and total expenditure in 2012 differ between the new adopters and those who have not yet adopted oil palm. The logic behind is that if the average annual income and expenditure differ between these two subsamples, there is highly likely that the average income and expenditure in the adopters and non-adopters in the whole dataset to be different as well. The estimated results are shown in Table A4 in the **Supplementary Material**. The findings show that there is no statistically significant difference between the two groups. This implies that the adopters were not wealthier than the non-adopters in 2012, and thus the observed effects on dietary outcomes are actually due to oil palm adoption. Besides, the majority of the adopters in the dataset I use in this study are transmigrant households who received oil palm plantations from the government (and as explained above, this difference is effectively dealt using appropriate econometric estimation model). Those transmigrant households did have noticeable wealth when they arrived in Sumatra; they were actually under food aid for several years (Fearnside, 1997). In other words, households in Jambi are adopting oil palm by the prospect to improve welfare (dietary outcomes), but it is unlikely that a priori wealth status systematically derives the observed dietary benefits in the adopters. Against this background, I conclude that reverse causality is not a problem in the data I use in this study.

Effects of Oil Palm Adoption on Food and Nutritional Adequacy

In the final analysis, I present and discuss the effects of oil palm adoption on food and micronutrient adequacy. I mainly discuss the impact of oil palm adoption on the mean

TABLE 5 | Effects of oil palm adoption on household diet outcomes.

	(1) Fruits and vegetable adequacy	(2) Calorie adequacy	(3) Iron adequacy	(4) Zinc adequacy	(5) Vitamin A adequacy	(6) Mean micronutrient adequacy
Oil palm dummy (dummy)	0.336**	0.386***	0.364***	0.549***	0.331***	0.350***
	(0.131)	(0.132)	(0.125)	(0.175)	(0.126)	(0.130)
HH size	−0.026	−0.320***	−0.182***	−0.177***	−0.136***	−0.194***
	(0.039)	(0.040)	(0.039)	(0.060)	(0.039)	(0.038)
Age of HH head	0.013**	0.002	0.001	0.012	0.001	−0.004
	(0.006)	(0.005)	(0.005)	(0.008)	(0.005)	(0.005)
Education of HH head	0.051***	0.051***	0.024	0.030	0.092***	0.067***
	(0.018)	(0.018)	(0.017)	(0.025)	(0.018)	(0.018)
HH owns business	0.489***	0.450***	0.293**	0.168	0.376***	0.362**
	(0.144)	(0.146)	(0.137)	(0.192)	(0.137)	(0.143)
Ethnicity: (1 = Melayu)	0.062	0.189	0.127	−0.051	−0.171	−0.099
	(0.150)	(0.149)	(0.146)	(0.213)	(0.144)	(0.147)
Cultivated land area	0.029**	0.058***	0.048***	0.048***	0.058***	0.082***
	(0.011)	(0.017)	(0.013)	(0.011)	(0.014)	(0.018)
Titled land (%)	0.346**	0.163	0.421***	0.289	0.386***	0.420***
	(0.157)	(0.154)	(0.149)	(0.218)	(0.149)	(0.153)
Migrant: (1 = transmigrant)	−0.092	0.135	0.207	0.287	−0.147	−0.034
	(0.169)	(0.168)	(0.161)	(0.222)	(0.162)	(0.167)
Credit from formal source	0.261*	0.323**	0.143	0.373**	0.028	0.081
	(0.140)	(0.139)	(0.132)	(0.181)	(0.132)	(0.137)
Dummy 2015	−1.730***	−0.401***	−0.512***	−0.440***	−0.333***	−0.547***
	(0.125)	(0.119)	(0.117)	(0.170)	(0.116)	(0.118)
Observations	1378	1378	1378	1378	1378	1378
Prob $\chi 2$	0.000***	0.000***	0.000***	0.000***	0.000***	0.000***

*Marginal effects after panel logit regression are shown with standard errors in parentheses. HH, household. *, **, *** denote significance at 10, 5, and 1% levels, respectively.*

probability of adequacy of fruits and vegetables consumed, calories, iron, zinc, vitamin A, and the average of the three micronutrients. The estimated results of the panel logit models are presented in **Table 5**. Since the estimated coefficients are reported in marginal effects at mean values, I have interpreted the results in percentages. Regardless of the diet indicator, oil palm adoption has statistically significant and positive effects, indicating that oil palm cultivation leads to higher food and micronutrient adequacy in general. On an average, the adoption of oil palm increases the probability of consuming fruits and vegetables by 33.6%, calorie adequacy by 38.6%, iron adequacy by 36.4, zinc adequacy by 54.9%, vitamin A adequacy by 33.1%, and average adequacy of the three micronutrients by 35%.

Together, with the results from the endogenous switching regression, it can be deduced that oil palm adoption improves the diets of farm households in the tropics, whether they belong to the transmigrant or the local communities. Hence, the nutritional impact might justify why farm households in the tropical are rapidly expanding oil palm cultivation. Moreover, several socioeconomic, farm, and demographic factors impact oil palm adoption and, at the same time, shape the diversity and adequacy of diets in those households.

CONCLUSION

I have analyzed the dietary impacts of oil palm adoption in farm households of the tropics, which has received little attention in the existing literature. In particular, using panel farm household data from Jambi province on the island of Sumatra, Indonesia, the effects of oil palm adoption on dietary diversity, quantities of fruits and vegetables consumed, calories, and food expenditure have been examined. Endogenous switching regression is applied to control for selection bias and to obtain counterfactual outcomes. Also, using panel logit regression estimation, I have examined the impact of oil palm adoption on dichotomous variables of household's diets, indicating whether the diet met the minimum adequacy level of fruits and vegetables consumed, calorie, iron, zinc, vitamin A, and the average of the three micronutrients. Rather than support the idea that adopting a perennial and non-food commercial crop worsens dietary quality and diversity in farm households, my findings support the opposite. The results illustrate that land-use change through oil palm adoption significantly improves the diets of farm households in the tropics. Positive effects are observed, notwithstanding, whether households belong to transmigrant or local communities.

Notably, oil palm adopters consume more diverse foods than non-adopters. The prevalence of undernourishment and micronutrient inadequacies are significantly less in such households. I also find that oil palm significantly increases household dietary diversity, food availability, food expenditure, and the adequacy of micronutrients through the income pathway. Furthermore, several socioeconomic, farm related, and demographic factors influence both oil palm adoption and the diversity and adequacy of household diets. This impact, leading to better diets, however, is complex and not straightforward; several the socioeconomic, demographic, and farm factors have different dietary impacts for adopters and non-adopters, which indicates that dietary diversity and quality in farm households should be promoted through strategies specifically tailored to the needs of these households.

Beyond the possible dietary impacts of oil palm adoption and the confounding factors, the findings also show that diets of farm household in the tropics are highly vulnerable to price and income shocks; the diets in both types of households were worse in 2015 than in 2012, as a consequence of the global drop in prices of oil palm and rubber. Moreover, oil palm is more capital-intensive as compared to the labor-intensive rubber, rice, or alternative crops cultivated in the tropics. This implies that expansion of the crop in smallholder setting of the rural tropics releases more labor to the rural labor market; women and older people might be more affected as they are less likely to work in oil palm plantations (Kubitza et al., 2018). Thus, creating off-farm work opportunities might help address issues related to price and income shocks, as well as to absorb the excess rural labor released because of the expansion of oil palm in the tropical rural areas.

In sum, the findings suggest that smallholder farmers seem to adopt and expand land-use types that provide greater dietary benefits. From a policy perspective, policy-makers interested in maintaining the tropical rainforests, regulating oil palm plantations and tackling nutritional deficiencies in the tropics should not overlook these dietary benefits for farm households.

I conclude by highlighting a few limitations that might be beneficial in future research. The focus of this study is to better understand the impact of oil palm adoption through the agricultural income pathway, not to capture all potential impact pathways. Future studies should, therefore, identify and investigate other impact pathways on how oil palm adoption influences household diets. Moreover, I have used a statistically valid selection instrument to account for selection bias as well as have tested for possible endogeneity issues. However, this is not to claim that the selection instrument used and the endogeneity tested in this paper are absolutely perfect. Yet this still helps to reduce the level of the statistical endogeneity problem often faced in this kind of study by reducing the level of bias in analyzing the dietary and nutritional impacts of oil palm adoption in the tropics. Similarly, I note that the results here reflect the situation in Jambi and may not be generalized. Follow-up research might test with data from other regions and use different estimation techniques to increase the reliability of the findings in this study. Finally, household-level dietary indicators are used in this study. Household-level food consumption is the entry point for any policy that targets individuals because a person can access and consume only the foods present in her/his household. Recent evidence also supports that household-level food consumption is strongly correlated with individual-level dietary intakes. However, it remains to be studied about how household-level food consumption could be proxy for the mean probability of adequacy of micronutrients in multi-site quantitative datasets.

DATA AVAILABILITY

The dataset analyzed for this study can be openly accessible from EFForTs-Information System: University of Goettingen (https://efforts-is.uni-goettingen.de).

ETHICS STATEMENT

Oral informed consent was obtained from village heads and all respondents before the survey was implemented. All study participants were informed that any information collected was to remain confidential and solely used for research purpose. Every identifying datum was also anonymized in the final dataset used for the analyses. As per the Collaborative German—Indonesian Research Project CRC990 and national guidelines, written legal consent from study participates were not required.

AUTHOR CONTRIBUTIONS

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

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2019.00075/full#supplementary-material

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Impacts of Mainstream Hydropower Development on Fisheries and Human Nutrition in the Lower Mekong

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Dams provide energy and irrigation water, but also alter natural water flows that support fisheries. This tradeoff presents a risk for human nutrition in regions dependent on aquatic foods, including the Lower Mekong Basin (LMB), where over 100 dams are planned or in construction. Previous models estimate significant reductions in fishery production resulting from these dams. This study estimates the number of new nutritionally insecure people (i.e., those at risk for nutritional deficiencies) associated with Mekong damming. We calculated population-level nutritional needs based on the Estimated Average Requirements (EARs) for Cambodia and the entire LMB. We then estimated fish-derived nutrient supplies by integrating data on annual fishery production and fish nutrient content for a wide range of species. Finally, we synthesized available literature and modeling results on the impacts of damming on fisheries production, and estimated the consequent impact on inadequate intakes of protein, zinc, niacin, thiamin, riboflavin, and calcium, as well as potential vulnerability to losses of dietary iron. Hydropower development could restrict access to subsistence fish from the Mekong River Basin and lead to increased risk of nutritional deficiencies in Cambodia and the LMB. Our median estimates suggest that by 2030, relative to 2010, inadequate intakes could lead to an increased population at risk of nutritional deficiencies in the LMB by 0.21 to 2.23 million people for protein, 0.12 to 1.17 million people for zinc, 0.41 to 1.58 million people for niacin, 0.47 to 0.87 million people for thiamin, 0.70 to 2.31 million people for riboflavin, and approximately 10,000 people for calcium. This increased population at risk is additional to those currently malnourished. We then calculated that the average iron intake of many age-sex groups (constituting 58% of the population) will be below 150% of their EAR in 2030, indicating a potential risk of increased inadequate iron intake. Fish is the main source of animal source foods and critical micronutrients in the LMB. In the absence of mitigation efforts, any reductions in fishery production could increase already high levels of nutrient deficiency, creating a widespread risk of nutrition insecurity.

Keywords: food security, micronutrient deficiencies, fisheries, malnutrition, dams, health impact assessment

INTRODUCTION

Growing human populations and economic development drive increases in global demand for food and energy, interlinked sectors that heavily rely on limited water resources (World Economic Forum, 2011). Dams produce hydropower, provide water for food production, modify water flows, and restructure fisheries, placing them at the center of the challenge to sustainably manage the food-energy-water (FEW) nexus (D'Odorico et al., 2018). With 3,700 large hydropower dams planned worldwide (Zarfl et al., 2015), understanding FEW tradeoffs is increasingly important.

At the forefront of the FEW nexus and large-scale damming conversations is the Mekong River, which starts in the Tibetan Plateau and empties into the South China Sea. The Upper Basin is primarily in China and Myanmar, and the Lower Mekong Basin (LMB) spreads through Lao PDR, Thailand, Cambodia, and Vietnam (Mekong River Commission, 2010). The Mekong is a key trade route, provides irrigation water and electricity, and is one of the world's most productive inland fisheries (Baran and Myschowoda, 2009). With over 100 dams planned in the LMB, including eleven planned on the mainstream by 2020 (Dugan et al., 2010), electricity availability is projected to increase by 900% (Pittock et al., 2016).

However, these dams severely affect fisheries by disrupting flows, sediment load, migration routes, and downstream flood regimes. The quantity and timing of water flows shape Lower Mekong Basin (LMB) fisheries. Flood pulses are particularly integral for the ecology of Tonle Sap Lake (Kummu and Sarkkula, 2008), a major regional fishery, which receives 53.5% of its flow from the Mekong (Kummu et al., 2014) and supports the fisheries of one million people (Stone, 2011). By reducing downstream sediment loads, dams restrict nutrient inputs, limiting the primary production that supports fishery production (Baran et al., 2001). Dams also block spawning routes for migratory fish, which constitute at least 35% of fish caught in the Mekong (ICEM (International Centre for Environmental Management), 2010; Stone, 2011). As a result, the LMB provides an important case for studying the effects of damming on nutrition from lost fisheries, an often-overlooked cost to dam construction.

Reductions in fishery production will affect the nearly 50 million people dependent on LMB fisheries for food and livelihoods (Orr et al., 2012; Pittock et al., 2017). Migratory fish groups that are highly vulnerable to damming represent \$1.4 billion of the \$2.4–3 billion of first-sale price values in the LMB (Barlow et al., 2008). Throughout the LMB, fish are a critical part of the diet (Hortle, 2007; Buoy et al., 2009), and a major source of protein, iron, zinc, calcium, B vitamins, and other micronutrients (Roos et al., 2007a,b; Chamnan et al., 2009; Golden et al., 2016; Vilain and Baran, 2016; Vilain et al., 2016). However, the contribution of fish to nutrition security is often overlooked in decision-making.

Without targeted mitigation efforts, reductions in fishery production seriously impact nutrition and malnutrition-related diseases (ICEM (International Centre for Environmental Management), 2010; Pittock et al., 2016, 2017). While people will adapt to lower fish production in a variety of ways and to varying degrees, thus offsetting some nutritional losses, the capacity for adaptation is uncertain. The objective of this study is to quantify the number of nutritionally insecure people (i.e., those at risk of nutritional deficiencies)—that is, the population estimated to be below the Estimated Average Requirement (EAR) cut-point for adequate intake of nutrients—when fish consumption changes from Mekong damming.

Previous studies estimated that damming may decrease LMB fishery production by 4 to 42% (ICEM (International Centre for Environmental Management), 2010; Mekong River Commission, 2010; IFReDI (Inland Fisheries Research Development Institute), 2013). These studies range in methodology from process-based models of hydrology, fish populations, and ecology (Baran et al., 2007; ICEM (International Centre for Environmental Management), 2010; Ziv et al., 2012; IFReDI (Inland Fisheries Research Development Institute), 2013) to causal loop diagrams (Pittock et al., 2016) and time series modeling (Sabo et al., 2016). Studies have also estimated protein losses associated with fishery production reductions (ICEM (International Centre for Environmental Management), 2010; Pittock et al., 2016, 2017) and the water and land resources required to replace fish catch declines with terrestrial foods (Orr et al., 2012). However, without quantifying micronutrient losses or comparing nutrient losses to nutrient requirements, the literature may underestimate the human health risks of hydropower development.

In this work, we calculated population-level nutrient (protein, zinc, niacin, thiamin, riboflavin, and calcium) requirements for Cambodia and the entire LMB from existing demographic data. We used published fishery production projections to calculate nutrient supply from fishery production. By comparing nutrient requirements and fishery nutrient supply, we calculated the increase in prevalence of those at risk of nutrient deficiencies in 2030 relative to 2010. For iron, we quantified the population currently near the threshold of deficiency with the population near the same threshold as fish availability shrinks under damming scenarios. This analysis is the first study to connect projected fishery declines under damming to a potential increased risk of nutrient deficiencies, a critical step to fully accounting for the human health consequences of damming.

METHODS

Fisheries Production Literature Review

Our analysis of nutritional insecurity from damming in the LMB utilizes GENuS, a recently constructed database of fish nutrient content, to enrich previous modeling efforts on the impacts of damming on fish catch. To identify published estimates of fish catch declines under different damming scenarios, we first reviewed gray literature and government reports that we located during an in-country consultation in Cambodia. We then included journal and white paper projections of fisheries production declines attributed to damming. We searched Web of Science for publications in English with projections that met the following criteria: (1) included quantitative estimates of fish catch reductions for specific damming scenarios; (2) had a geographic scope of either Cambodia or the entire LMB, as defined by ICEM (International Centre for Environmental Management) (2010);

TABLE 1 | Summary of publications used for micronutrient analysis.

References	Geographic extent	Baseline year	Mechanisms of fish catch change
IFReDI (Inland Fisheries Research Development Institute) (2013)	Cambodia	2011	- Barrier effect - Decreased flood plains - Changes in aquaculture production - Imports - Exports - Non-consumptive use - Changes in population
ICEM (International Centre for Environmental Management) (2010)	LMB	2000	- Barrier effect - Decreased floodplains - Increased reservoir capacity - Changes in aquaculture production
Mekong River Commission (2010)	LMB	2000	- Changes in aquaculture production - Exports - Imports - Barrier effect - Decreased floodplains - Increased reservoir capacity

and (3) was published in 2010 or after to ensure that the most recent data were being used. Of the eleven studies considered, only three met these criteria. Unused publications calculated reductions in fisheries production as a function of hydrological factors (Sabo et al., 2016, 2017) or in geographical areas that are not comparable to Cambodia or the LMB (Baran et al., 2007; Ziv et al., 2012). Changes within ecosystems or bodies of water that do not align geographically with the scope of this study cannot easily be scaled up or scaled down to produce comparable results.

The studies included in our analysis are: (ICEM (International Centre for Environmental Management), 2010; Mekong River Commission, 2010; IFReDI (Inland Fisheries Research Development Institute), 2013), the first for Cambodia and the latter two for the entire LMB (**Table 1**). Each model included multiple projections, representing various damming scenarios. We classified damming scenarios according to damming extent, where "none" represents construction of no mainstream dams, "max" represents construction of eleven mainstream dams, and "some" represents construction of an intermediate number of dams, which varies across the studies.

The IFReDI (Inland Fisheries Research Development Institute) (2013) estimated 2011 baseline fisheries production in Cambodia as 889,000 tons/year (damming extent: none). With only one (the Stung Treng dam) of the possible 11 dams constructed in the LMB, fisheries production is projected to decrease by 34,000 tons/year (4%) in Cambodia by 2030 (damming extent: some; **Table 2**). With all 11 mainstream dams, fisheries production is projected to decrease by 183,000 tons/year (21%) (damming extent: max; **Table 2**). ICEM (International Centre for Environmental Management) (2010) estimated 2,000 baseline fisheries production in LMB as 2,100,000 tons/year, similar to the Inland Fisheries Research and Development Institute baseline estimates for Cambodia on a tons/year/km^2 basis. Without additional mainstream dams in 2030, this study estimated fisheries production to decline by 480,000 tons/year (23%) in the LMB (damming extent: none; **Table 2**). Fish catch was estimated to decline by 600,000 and 680,000 tons/year with the addition of six mainstream dams built upstream of Vientiane and nine mainstream dams operating upstream of Khone Falls, respectively (damming extent: some; **Table 2**). With 11 mainstream dams by 2030, fisheries production is estimated to decline by 880,000 tons/year (42%) (damming extent: max; **Table 2**). The Mekong River Commission (2010) estimated year 2000 baseline fisheries production in the LMB as 2,300,000 tons/year (damming extent: none). With construction of 11 mainstream dams by 2030, the study estimated fisheries production could decrease by 900,000 tons/year (41%) (damming extent: max; **Table 2**). Taken together, fishery catch is projected to decline by 4–33% with some damming and 21–42% with all 11 proposed dams.

Estimated Average Requirements (EARs)

The Estimated Average Requirement (EAR) is the amount of a given nutrient that is assumed to be sufficient to meet the needs of 50% of the population, and is used as an input in our deficiency risk calculations (section Prevalence of Inadequate Intake). EARs are generally calculated based on the reference nutrient intake (RNI)—the amount sufficient to meet the needs of ∼97% of the population—and an RNI-to-EAR conversion factor. RNIs for calcium, thiamin, riboflavin, niacin, and iron are provided for each age and sex group by the World Health Organization (WHO) Food Agriculture Organization (FAO) of The United Nations (2004) and the conversion factors are provided separately by the World Health Organization (2006). For some age-sex groups, these conversion factors for iron did not exist, and we used the relationship between EAR and RNI in the Institute of Medicine's guide to nutritional requirements (Institute of Medicine of the National Academies, 2006) to derive them.

EARs for iron range from 2.2 to 42.8 mg/day (World Health Organization (WHO) Food Agriculture Organization (FAO) of The United Nations, 2004), depending on age sex, and iron availability in the diet. The capability of the body to absorb iron is controlled by a range of factors, including foods and nutrients eaten with the iron in the same meal (e.g., ascorbic acid, calcium, or polyphenols), the type of iron consumed (heme or non-heme iron), as well as each person's overall iron status or concurrent disease load (Lynch et al., 2018). However, we do not have sufficient data for several of these factors within GENuS (polyphenols, alcohol) to allow for explicitly calculating the absorbable iron supply from the diet. Therefore, we instead account for the differences in dietary bioavailability of iron by estimating the differences in each country's iron requirement based on the bioavailability of their diet. Each country was assigned to a bioavailability category based on criteria meant to serve as proxies of the relevant components of the diet which control iron absorption: overall meat, fruit, and vegetable intake. These criteria were derived based on guidance from Hurrell and Egli (2010) and Hallberg and Rossander-Hultén (1991), and previously applied to a similar analysis (Golden et al., 2016). The exact consumption thresholds and the

TABLE 2 | Summary of projected capture fishery production changes for each reviewed scenario.

References	Damming scenario	Damming level	Freshwater fishery production (kilotons)	Production relative to 2010 (%)	Per capita freshwater fishery production (kg/person/year)
IFReDI (Inland Fisheries Research Development Institute) (2013)	Baseline	None	596–602	74–75	33.6–33.9
	With Stung Treng dam	Some	480–591	59–73	27.1–33.3
	With Sambor dam	Some	443–527	55–65	25.0–29.7
	With Stung Treng and Sambor dam	Some	443–527	55–65	25.0–29.7
	Eleven mainstream	Max	442– 526	55–65	24.9–29.6
ICEM (International Centre for Environmental Management) (2010)	Baseline	None	1,560–1,890	71–86	19.3–23.4
	Six mainstream dams built upstream of Vientiane	Some	1,500–1,830	68–83	18.6–22.7
	Nine mainstream dams operating upstream of Khone Falls	Some	1,420–1,750	65–80	17.6–21.7
	Eleven mainstream	Max	1,220–1,550	55–70	15.1–19.2
Mekong River Commission (2010)	Baseline	None	1,700–2,400	68–95	21.1–29.7
	Six to nine mainstream dams	Some	1,400–2,350	56–93	17.3–29.1
	Eleven mainstream	Max	1,400–2,000	56–79	17.3–24.8

corresponding bioavailability categories are described in detail in **Supplementary Table 1**.

Physiological zinc requirements were estimated by the International Zinc Nutrition Consultative Group (IZiNCG) et al. (2004). We estimated protein requirements following a methodology first used by Medek et al. (2017) and refined by Smith and Myers (2018). Because protein requirements are determined as grams per kilogram of body weight, we first estimated the baseline body weight for all age-sex groups in each country. For adults, we used mean height for each age, sex, and country (NCD Risk Factor Collaboration (NCD-RisC), 2016), paired with WHO's minimally acceptable BMI of 18.5 to estimate baseline weight. For adolescents and children over five, we used WHO height-for-age growth curves (World Health Organzation Multicentre Growth Reference Study Group, 2006), paired with the measured adult height in each country, to extrapolate the corresponding height of the younger age groups. For children under five where no height data was available, we used 50th percentile weight-for-age to determine inadequate protein intake (World Health Organzation Multicentre Growth Reference Study Group, 2006). The weight of each age-sex group was then multiplied by the corresponding WHO protein requirement to derive final requirements by demographic group (Joint FAO/WHO/UNU Expert Consultation, 2007).

For nearly all nutrients for which we estimate the risk of deficiency, the requirements for pregnant and lactating women are greater than those of non-pregnant or non-lactating women of the same age. The only exception is calcium for lactating women, for which the requirement is the same as for non-lactating women. For all other nutrients, we estimated the additional requirements of these women using the 2010 age-specific fertility rate from the United Nations World Population Prospects for each age category (United Nations, 2017). We multiplied the birth rate by the fraction of the year constituting the average gestational length (40 weeks) to estimate the population of pregnant women requiring additional nutrition. We estimated the population of lactating women by multiplying the median duration of breastfeeding in each country from the WHO Global Data Bank on Infant and Young Children Feeding (World Health Organization, 2018a); Laos and Thailand had missing data, and thus these countries were assigned the regional average.

Nutrient Supply

To identify the nutritional adequacy in the diet after damming scenarios relative to the baseline, we needed to first estimate the total protein, zinc, niacin, thiamin, riboflavin, and calcium coming from both fish and from the total diet. Fish also provide many other important nutrients in the diet (Vilain and Baran, 2016; Vilain et al., 2016), but we were unable to include them in our analysis because we either lacked sufficient data on nutrient intake (vitamins B12, iodine, omega-3 fatty acids) or there were no established nutritional requirements (copper, manganese). Furthermore, fish can also significantly contribute to vitamin A intake (Roos et al., 2007a), but our dietary intake estimates were not sufficiently precise to be able to identify which parts of the fish were eaten in each country. This information exerts a large influence on any estimate of how much vitamin A is provided via fish, as eyes and liver contain nearly all the vitamin A in a given fish. Therefore, we did not feel justified in estimating vitamin A given the large uncertainties in our data.

We used the Global Expanded Nutrient Supply (GENuS) model to assign nutrient supplies for each age-sex group in 2010. The methodology of GENuS is described in detail in Smith et al. (2016) (Smith et al., 2016) and datasets are available at the Harvard Dataverse (Smith, 2016). To briefly summarize the construction of GENuS, the dataset uses FAO food balance sheet data, combined with additional production and trade data, to estimate the food supply for 225 foods in 151 countries since 1961. Of the 225 foods and food groups in GENuS, seven of these—freshwater fish, demersal fish, pelagic fish, crustaceans, cephalopods, molluscs, marine mammals, and other marine fish—capture the contributions of fish and seafood to the diet. Per capita food supplies are then paired with their corresponding nutrient densities from six regional food composition tables to estimate the nutrient supply in the diet across 23 different nutrients. Nutrients were only included in GENuS if they were found in more than one table, but most nutrients were found in all six. For this analysis of the four Mekong Basin countries, all nutrient density information was drawn from the Institute of Nutrition, Mahidol University (2014), which includes information on 71 different species of fish and seafood consumed regionally. Finally, the national nutrient supplies were then combined with data from the Global Dietary Database (GDD) (Global Nutrition Policy Consortium, 2018) on food consumption patterns by age and sex to generate detailed food and nutrient supply estimates across 34 demographic groups. GDD data were only used to identify different eating patterns of food groups between age and sex groups, but were not used to convert food supplies to food intakes due to the GDD data lacking full information on the entire diet. Because GENuS relies on FAO data to estimate the per capita supply of fish, and because it has been shown that freshwater fish catch data collected by FAO is particularly inaccurate (Bartley et al., 2015), we performed an additional correction factor on the supply of freshwater fish in the diet. A recent analysis by Fluet-Chouinard et al. (2018) re-derived the estimated catch of freshwater fish in many countries using a range of complementary data sources, particularly household expenditure surveys. We used the ratio between their corrected catch estimates and the primary FAO data and multiplied the supply of freshwater fish in each country by that value. Those factors were 1.48 (Cambodia), 3.02 (Laos), and 2.54 (Thailand). For Vietnam, Fluet-Chouinard et al. (2018) has insufficient data to derive corrected catch estimates, so we simply used the FAO values for freshwater fish supply. For comparison, we also ran a parallel set of analyses using the uncorrected GENuS estimates (**Supplementary Table 2**).

For zinc, we converted total zinc supplies to absorbable zinc to account for the absorption-inhibiting influence of phytate. We estimated paired phytate and zinc values for all foods in GENuS using a composite food composition table that was constructed using an array of data sources (Smith and Myers, 2018). For grains, we also accounted for the effect of processing and fermenting on each grain's zinc and phytate content. We used regional estimates of the percentage of grains processed and nutritional impact of processing from Wessells et al. (2012) and applied these to per capita food and nutrient supply estimates. We used the GENuS methodology to estimate dietary phytate and zinc from edible food supply, as well as associated uncertainties. We estimated absorbable zinc from dietary zinc and dietary phytate using the Miller equation (Miller et al., 2007; Hambidge et al., 2010). For protein, we followed established methodology (Medek et al., 2017) and assumed that plant-based protein was 80% digestible and animal-based protein was 95% digestible.

Prevalence of Inadequate Intake

For all nutrients except iron, we estimated distributions of intra-individual intake to determine prevalence of inadequate intake. Distributions were assumed normal (in line with previous studies, e.g., Arsenault et al., 2015; Beal et al., 2017) and we estimated standard deviation using coefficients of variation (CV) based on nutrient: calcium (30%), zinc (25%), thiamin (30%), riboflavin (30%), and niacin (25%). For protein, we assumed that intake distributions were log-normal, with CV derived from a relationship between the Gini coefficient and the shape of the distribution. The methodology for CV derivation is further explained in Smith and Myers (2018) and Medek et al. (2017). The CV for each country is as follows: Cambodia (37%), Laos (47%), Thailand (50%), and Vietnam (46%).

Prevalence of inadequate intake was estimated using the EAR cut-point method (Institute of Medicine (US) Subcommittee on Interpretation Uses of Dietary Reference Intakes; Institute of Medicine (US) Standing Committee on the Scientific Evaluation of Dietary Reference Intakes, 2000). If certain key criteria are satisfied, the EAR cut-point method suggests that the number of people whose intake falls below the EAR is equivalent to the number of people who are inadequate in that nutrient. The necessary criteria are that each person's intake is independent of their requirement, the variability of intakes is greater than the variability of requirements, the distribution of requirements is approximately symmetrical, and the actual prevalence of inadequacy is neither very low nor high. For the nutrients studied here (except iron), each of these criteria hold, and we can apply this method. The cut-point method provides an estimate of the population that is consuming an inadequate amount of nutrients to meet their physiologic needs. Other intervening factors may affect whether a person with inadequate nutrient intake is physiologically deficient in that nutrient (e.g., consuming supplements, suffering additional illnesses), so we only characterize populations at *risk of deficiency*, not *deficient*, and this is the terminology we apply throughout the rest of our study. This method has been applied in numerous studies assessing the population-level status in the risk of dietary nutritional inadequacy (Wessells et al., 2012; Arsenault et al., 2015; Beal et al., 2017; Medek et al., 2017).

This calculation was performed using 2010 as a baseline. For each of the Mekong damming scenarios, we then reduced nutrient intake provided by fish by an amount proportional to the forecasted decline in per capita fish catch by 2030. The change in the prevalence of those at risk of deficiency for each scenario was measured relative to baseline, then multiplied by the population of each age-sex category in 2030 and by the proportion of each country that is located within the Mekong Basin.

We were unable to estimate the change in the risk of deficiency for iron, unlike with the other nutrients, because iron does not meet the criteria for the application of the cut-point method; namely, the distribution of requirements is not normal, particularly for menstruating women. Furthermore, we are unable to apply the probability method here because we do not have sufficient data to generate a distribution of requirements for the population, primarily because there are many confounding factors that influence the body's absorption of iron. However, we use GENuS data to provide estimates of iron contribution from seafood, as we know that iron is often sourced from Mekong fisheries (Roos et al., 2007b), calculating the sum of the populations of age-sex groups for whom the mean intake of the nutrient falls below 150% of their EAR, indicating a very rough measure of those that are close to inadequacy. This method was used similarly in a previous study as a proxy for risk of deficiency to lost fish in the diet (Golden et al., 2016).

RESULTS

Across the six reviewed damming scenarios (baseline plus five combinations of different damming futures), increasing human population and decreasing fishery production leads to hundreds of thousands to millions of newly nutritionally insecure people in Cambodia and the LMB by 2030, as compared to 2010 (**Figure 1**). For each nutrient, we present the medians of all projections simulating different damming scenarios and fish production (**Figure 1**). The scenarios range from a best case with only the Stung Treng dam as a mainstream dam to the worst case with 11 additional mainstream dams (**Supplementary Table 2**). The three main findings are: (1) all scenario projections are positive with error bars that do not include zero for all micronutrients and protein; (2) variance is higher in the worst-case scenarios; and (3) variance differs by nutrient (**Figure 1**). There is high uncertainty in the models, but all results indicate unequivocal and severe negative nutritional impacts. Results for the parallel model without the correction factor for freshwater fish are presented in **Supplementary Table 3**. All values reported below are from the model which includes the freshwater correction factor.

The median best- and worst-case scenarios indicated that, by 2030, there would be up to an additional 190,000–350,000 people newly at risk for deficiency for zinc in Cambodia and 90,000–910,000 in the LMB (**Figure 1**; **Supplementary Table 3**). For protein, as many as 340,000–660,000 more individuals would be at risk for deficiency in Cambodia and 160,000–1,690,000 more in the LMB (**Figure 1**; **Supplementary Table 3**). Projections for the maximum increase in the population at risk for niacin deficiency spanned 160,000–310,000 in Cambodia and 90,000–940,000 in the LMB (**Figure 1**; **Supplementary Table 3**). By 2030, we projected the population newly at risk for thiamin deficiency to increase by up to 70,000–130,000 people in Cambodia and 50,000–520,000 people in the LMB (**Figure 1**; **Supplementary Table 3**). Riboflavin projections indicated that there would be as many as 260,000–470,000 additional people at risk for deficiency in Cambodia and 140,000–1,380,000 in the LMB. For calcium, median projections indicated no additional risk of deficiency for Cambodia and an additional 10,000 people at risk in the LMB (**Figure 1**). This is largely because 99% of the Cambodian population is currently at risk for calcium deficiencies, not leaving much space to increase risk. Finally, we projected that up to 58% of the population (or 47 million people) in the LMB are in age-sex groups where their mean iron intake may be below 150% of their EARs by 2030, indicating a dire need to increase consumption of iron-rich foods such as fish. These populations include all age-sex groups in Cambodia and Laos, women aged 10–49 in Thailand and Vietnam, and men aged 10–24 and 50+ in Thailand.

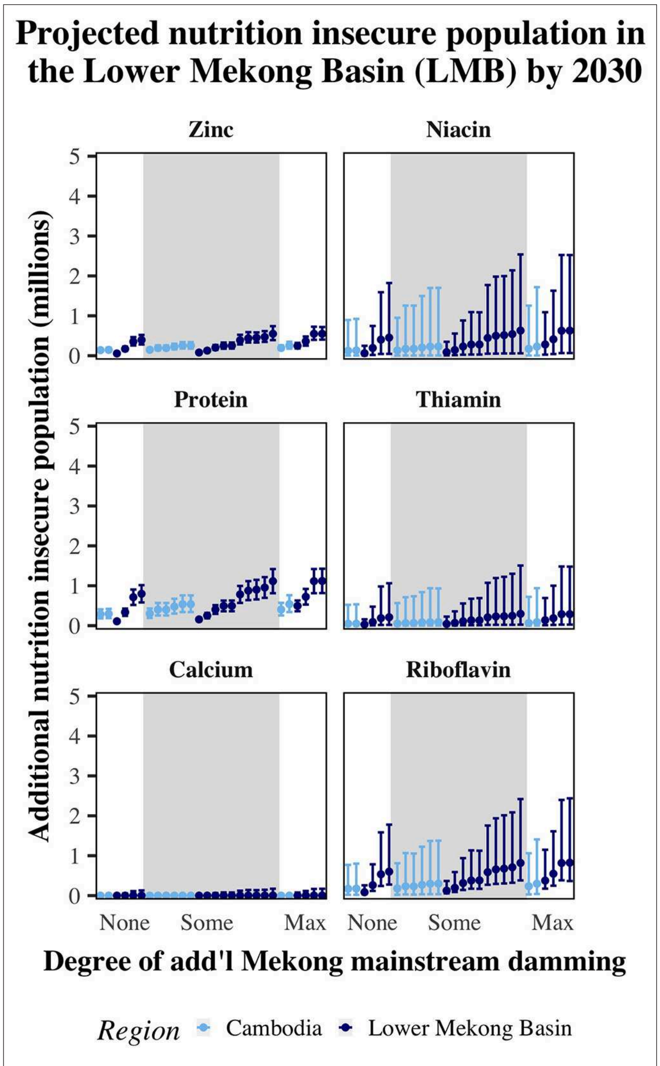

FIGURE 1 | Estimated increases in 2030, relative to 2010, of the human population classified as nutrition insecure (i.e., below the EAR cut-point) for zinc, protein, niacin, thiamin, riboflavin, and calcium, according to each damming scenario projection. We classified each damming scenario according to the extent of additional mainstream damming ("none," "some," or "max") which corresponds to 0, 1–9, and 11 additional dams. The white and gray panels break the bars into these categories. The error bars represent the 95% confidence interval for each scenario.

DISCUSSION

In this study, we calculated the increased risk of potential nutritional deficiencies that could arise from hydropower damming across a broad range of micronutrients and protein for Cambodia and the entire LMB. Our results extend prior studies which have primarily considered reductions in fishery production alone or have focused only on the protein contribution of fish. By comparing human nutrient requirements and shifts in fishery nutrient supplies resulting from future damming scenarios, we calculated the increased risk of inadequate intakes in 2030 relative to 2010. This analysis demonstrates that, in the absence of mitigation efforts, all forms of damming will increase nutritional insecurity in Cambodia and the LMB. Our calculations also add nuance to the nutritional needs of people in this region, by accounting for varying nutritional demands by individuals of different ages and sexes, as well as increased nutritional demands during pregnancy and lactation.

By comprehensively estimating a broad suite of potential nutritional outcomes that could arise from damming, we provide a foundation for more precisely quantifying the health costs of hydropower development. Each of these risks of nutritional deficiency has consequent health impacts that lead to profound risks of disability and disease. For instance, protein deficiencies can influence almost all aspects of physiological functioning, and we estimated a median increase of up to 2.2 million people at risk of protein deficiency by 2030 in the LMB. Our estimates of protein deficits improve on previous calculations of protein reductions in the Mekong river basin (ICEM (International Centre for Environmental Management), 2010; Orr et al., 2012; Pittock et al., 2016, 2017) by incorporating multiple fishery projection estimates and relating the protein losses to actual population requirements (Pittock et al., 2016). Our results reinforce the previous findings that protein availability from fish catch will decrease and will broadly impact regional food security.

Beyond protein benefits, fish from the Mekong river basin provide significant micronutrient supplies to the region, especially iron and zinc. Iron-deficiency anemia is responsible for high maternal mortality rates, reduced school performance in children, and lost economic productivity (World Health Organization, 2018b). In fact, iron deficiency is estimated to account for 2.29% of all disability-adjusted life years (DALYs) lost in Cambodia and for 3.21% of all DALYs lost among Cambodian women (Hay et al., 2017). In 2030, we project that 58% of the population will have an average iron intake that is already lower than 150% of their estimated requirement, which will only worsen under projected damming scenarios.

Zinc is critical for immune function and preventing serious childhood illnesses such as diarrhea (National Institute of Health, 2018c). We predicted a median increase in those at risk for zinc deficiency for more than 1 million people in the LMB by 2030 relative to baseline. Niacin deficiency can lead to diarrhea, wasting, and dementia (World Health Organization (WHO) Food Agriculture Organization (FAO) of The United Nations, 2004). In the best-case scenario, by 2030, we estimate a potential increase of up to 1.5 million people at risk of niacin deficiency in the LMB. Under the worst-case damming scenarios, niacin deficiencies could reach an additional 4.4 million people (5.5% of the population within the LMB). Thiamin is essential for cell growth and development, and thiamin deficiency can cause weight loss, cognitive deficits, muscle weakness, and cardiovascular symptoms (National Institute of Health, 2018a). We estimate a median increase of more than 800,000 people at risk for thiamin deficiency in the LMB by 2030, which could lead to an array of health consequences. Maximal worst-case scenarios could lead to an additional 4.0 million people at risk of riboflavin deficiencies, known to contribute to an increased risk of anemia and cardiovascular diseases (Powers, 2003). Calcium deficiencies, leading to increased risk of reduced bone mass and osteoporosis (Nordin, 1997), are expected to increase the least of all modeled nutritional deficiency risks, with 370,000 people projected to be at risk of deficiency in maximal worst-case scenarios in the LMB.

Although we were unable to quantify vitamin A and B12 deficiency, we expect reduced fishery production to greatly impact vitamin intake in LMB. Vitamin B12 is only found in animal products, and recent estimates indicate that about 19% of the global population is at risk for deficiencies of nutrients that derive mostly from animal consumption (which includes vitamin B12) (Golden et al., 2016). Given that Cambodians obtain 80% of animal source foods intake from fish (Royal Government of Cambodia, 2010), we expect the population to have few other sources of vitamin B12, which is essential for blood cell formation and neurological functioning (National Institute of Health, 2018b). Although found in colorful fruits and vegetables, freshwater fish are very rich sources of vitamin A (Roos et al., 2007a). Without adequate consumption of vitamin A, immunity is compromised leading to increased risk of infectious disease and night blindness (Lim et al., 2012).

Increases in the risk of protein and micronutrient deficiencies are among a wide array of social costs related to hydropower development. There are also clear benefits. Hydropower can help satisfy the Mekong region's electricity demand, which is increasing at a rate greater than the rest of the world [ICEM (International Centre for Environmental Management), 2010]. Increased hydropower can enhance the region's trade, reduce its reliance on fossil fuels, and reduce greenhouse gas emissions for the power sector [ICEM (International Centre for Environmental Management), 2010]. Cambodia and Lao PDR also plan to export power, the revenue from which can develop education and health care [ICEM (International Centre for Environmental Management), 2010].

Our analysis is limited by risks of both over- and under-estimation. We may be underestimating the impact of fish catch declines on nutrition security because we are not accounting for indirect pathways to nutrition, such as through income generation. We may also be biasing our results by using nutrient supply data as a proxy for nutrient intake data, though the GENuS data apportions the FAO food supply data to age-sex groups using nationally representative food consumption survey recalls conducted in 187 countries (Smith et al., 2016). Although 24 h recalls are often seen as the gold standard for food consumption statistics, GENuS provides us with systematic data across the LMB that proxies consumption. Furthermore, we may also be underestimating the amount of fish in the diets based on comparison between FAO per capita fish availability and more detailed information on fish consumption (Hortle, 2007). There is also a possibility that we are overestimating the nutritional impacts from hydropower development because we do not consider replacement diets when fish is lost (e.g., through livestock, agriculture, or aquaculture development).

It is clear that dams positively contribute to other food production systems by storing irrigation water and creating habitat for aquaculture. Unfortunately, there are numerous barriers to adopting small-scale aquaculture practices (Beveridge et al., 2013; Béné et al., 2016; Richardson and Suvedi, 2018), making it unlikely that aquaculture could compensate for losses in fish catch in providing food for those vulnerable to malnutrition [ICEM (International Centre for Environmental Management), 2010]. Replacement of lost fish with other food sources is beyond the scope of this analysis, but is an important future direction for research. Some groups have even demonstrated, through randomized control trials, the importance of local fish food products in improving Cambodian nutrition (Sigh et al., 2018).

New hydropower management approaches may be able to reduce food-energy tradeoffs, particularly if efforts are made to retrofit hydropower development to mitigate environmental impacts (Arias et al., 2014). One recent study argues that operating dams to better match natural hydrologic regimes could increase fish catch by ∼50% within 8 years, while producing the same amount of power (Sabo et al., 2017). Many experts doubt the analysis and interpretations of this study (Halls and Moyle, 2018; Williams, 2018), however, and other experts doubt that necessary investments will be made to retrofit the dams if they are constructed (Dugan et al., 2010). Regardless of the exact predictions of fish futures in the Lower Mekong, accounting for the full nutritional impacts of dams may incentivize prevention or mitigation actions. This type of evidence could also enable collective action from affected citizens to integrate fisheries management into decentralized development planning (Ratner et al., 2017).

CONCLUSION

Our study estimates the nutritional risk associated with declines in fisheries production driven by hydropower development. Millions of people are at risk for nutritional deficiencies if proper attention is not paid to the health consequences of damming. It is clear that fisheries are important for Cambodia's path to meeting the Sustainable Development Goals to eliminate poverty and ensure food security and nutrition (Ratner et al., 2017), as the region's fish provide employment and food for millions of people (Hortle, 2007; Orr et al., 2012). But in a region that is already stretched for nutritional resources, further damming has the potential to place millions of people at risk of new nutritional deficiencies. This is not a challenge unique to the Lower Mekong Basin but affects all major riverine systems in nutritionally vulnerable countries around the world.

AUTHOR CONTRIBUTIONS

CG, BV, and JG designed the study. AS, ES, and JG conducted the literature review, data extraction, and analysis and production of visualizations. MS and SM provided analytical support in estimating dietary intake and the contribution of fish. CG and BV visited the Lower Mekong to discuss these issues with local stakeholders, NGOs, and government officials who provided input on our data analysis and interpretation. CG and AS drafted the article. All authors contributed to revising and approving the manuscript.

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SUPPLEMENTARY MATERIAL

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Food Insecurity and the Unsustainable Hunting of Wildlife in a UNESCO World Heritage Site

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Madagascar faces dual challenges in biodiversity conservation and public health. In order to identify strategies to reduce the unsustainable hunting of threatened species while maintaining or improving child nutrition, we quantified interactions among ecosystem indicators (lemur density and habitat biodiversity indices), health indicators (stunting, underweight, wasting, and anemia), nutrition, food security, and wildlife hunting through interviews of 1,750 people in 387 households and surveys of 28 wildlife transects with 156 habitat plots at 15 sites on Madagascar's Masoala Peninsula, a UNESCO World Heritage Site. The surveyed population ate 6,726 forest animals (mammals and birds), or a mean of 3.27 kg of wild meat per person (4.48 kg per adult equivalent) during the prior year. Local Malagasy were also highly food insecure (78% of households) and malnourished (for children under five, as many as 67% were stunted, 60% were underweight, 25% were wasted, and 40% were anemic). In some communities, nearly 75% of animal-sourced calories, 76% of protein, and 74% of iron came from forest animals-demonstrating a strong dependence on wild foods. Few micronutrient-rich alternatives to wild meats were available in adequate supply and many were highly volatile; for example, 79% of chickens died from Newcastle disease in the prior year. The survivorship of lemurs (94% of lemur species are threatened with extinction) depends on providing food security to a malnourished human population who commonly hunts wildlife for food. Currently, wildlife provides a critical source of micronutrients, yet the hunting of threatened species is an untenable solution to poor diet and food insecurity. Given the established connection between wild foods and human nutrition, reductions in forests and wildlife populations will also threaten the local food supply. In order to reduce the unsustainable hunting of threatened species while improving household food security and child health, we suggest testing the effects of increasing the affordability, accessibility, and stability of micro-nutrient rich animal-sourced foods in communities where forests contribute the most to food security.

Keywords: bushmeat, conservation, food security, hunting, lemur, Madagascar, Masoala National Park, nutrition

INTRODUCTION

While unsustainable hunting is widely recognized as a primary contributor to global biodiversity loss (Wilkie et al., 2016), little is known about the relationships between hunting, biodiversity, nutrition, and food security surrounding many of the world's protected areas. Malnutrition is a primary driver of the global burden of disease (International Food Policy Research Institute, 2016); with half of the deaths of all children worldwide associated with undernutrition (Black et al., 2013). If a child's diet is deficient in key micronutrients, such as zinc, iron, and vitamin A, their cognitive and physical growth can be delayed or impaired and their risk of infection and early death is significantly increased (Ezzati et al., 2002; Lopez et al., 2006; Black et al., 2013).

The country of Madagascar is one of the least food secure nations in the world (Economist Intelligence Unit, 2017). Its people spend proportionally more of their cash income on food than anywhere else on the planet (Economist Intelligence Unit, 2014, 2016). Further, Madagascar has one of the highest rates of stunting in the world (International Food Policy Research Institute, 2016), and faces high health burdens including poor maternal outcomes and high rates of both anemia and malaria (WHO, 2012; Mould et al., 2016; Rice et al., 2016).

Madagascar is also one of the most biodiverse places on earth. The nation has long been a global priority for conservation; most of the plant and animal species in Madagascar are found nowhere else on earth (Myers et al., 2000). Among these animals are lemurs, euplerid carnivorans, and tenrecs. Nearly 90% of all tenrecs, and 100% of all lemurs and euplerid carnivorans are only found in Madagascar. Lemurs are the most threatened group of primates on earth, and nearly all species (94%) are threatened with extinction because of habitat loss and unsustainable hunting (Schwitzer et al., 2013).

Recent global evidence shows many complex pathways by which forests can improve dietary diversity, health, and nutrition (Golden et al., 2011; Food Agriculture Organization., 2013; Johnson et al., 2013; Ickowitz et al., 2014; Rowland et al., 2016; Tata et al., 2019). While forest foods are rarely the staple food in a diet (Rowland et al., 2016), children who live in areas with greater forest cover eat more nutritious diets than those who do not (Ickowitz et al., 2014; Tata et al., 2019). In addition to primary forests, swidden and agro-forests also provide opportunities for families to increase the diversity of their diet and their access to many micronutrient rich foods (Ickowitz et al., 2016). Among these wild foods, wild animals provide an important source of calories, fat, protein, and bio-available micronutrients (Fa et al., 2003; Siren and Machoa, 2008; Golden et al., 2011; Sarti et al., 2015; van Vliet et al., 2017) Yet the unsustainable hunting of many wild species threatens their survival, the functioning of their ecosystems, and the food security and cultural identity of many people worldwide (Wilkie et al., 2016).

The interactions between biodiversity and food security are poorly understood within Madagascar. The Masoala National Park—a UNESCO World Heritage site—is one of Madagascar's most intact and biodiverse forest ecosystems (Kremen et al., 1999; Kremen, 2003). Yet, this national park is under significant threat from deforestation and unsustainable hunting (Allnutt et al., 2013; Borgerson, 2016; Zaehringer et al., 2017). Here, we aim to (a) describe the state of food security, nutrition, biodiversity, and hunting on the Masoala Peninsula; and (b) understand how their interactions affect the future of public health and biodiversity conservation in Madagascar.

METHODS

We used the following four multi-disciplinary methods to examine the interactions among human health, nutrition, and biodiversity at 13 sites surrounding the Masoala National Park. We collected this data during May until December 2015. All research was approved by Human Subjects Institutional Review Boards (Protocols #15-0331 Wildlife Conservation Society and #15-2230 Harvard T. H. Chan School of Public Health), the Republic of Madagascar and Madagascar National Parks (Permits 111/13, 325/14, 111/15, 218/15, 270/15, /MEEF/SG/DGF/DCB.SAPP/SCB). We obtained oral informed consent and/or assent from all participants.

Extensive Structured Interviews

All authors are either fluent in, or native speakers of, the local dialect of Betsimisaraka Malagasy. CB and BJRR asked members of 387 households in 13 communities about their hunting, collection of forest products for food, demographics, diet, health, income, and food security in a 1–2 h interview. We surveyed all households in small communities. In communities with >50 households, we randomly selected study households by using a grid system in each village, assigning a number to each household in each grid, and selecting a subset of households in all quadrants using a random number array. Individuals provided information about each type of cash-generation activity in Malagasy Ariary (MGA) and we converted estimates of cash income to United States Dollar (USD) at a rate of 3,000 MGA to the dollar (the conversion rate at the time of data collection). Because subsistence (and not cash) income was high, we also scored houses based on their size and the building materials used for their walls, floors, and roofs (ranked 1–3 based on local perceptions of quality), to provide a secondary indicator of wealth in addition to reported cash income. This total score was divided by the number of household members to control for the possibility that house size may increase with household size.

We determined food security using multiple methods. We used the Coping Strategies Index (CSI) (CARE, 2008) and the Household Food Insecurity Access Scale (HFAIS) (Coates et al., 2007) to measure changes in feelings, perceptions, and behaviors during the prior year (HFAIS) and prior week (CSI) in response to insufficient access to food (coping strategies). We then weighted CSI values based on the qualitative perception of the severity of each coping strategy in each community (categorically ranked on a scale of 1–4). A CSI or HFAIS score of 0 reflects a household which perceives itself as food secure and higher CSI or HFAIS scores reflect greater perceived food insecurity. We defined a food insecure household as any household that could not access adequate food to feed their family one or more days during the prior week. In order to

TABLE 1 | Masoala food security (2015)[†].

	Village 1	Village 2	Village 3	Village 4	Village 5	Village 6	Village 7	Village 8	Village 9	Village 10	Village 11	Village 12	Village 13	Masoala average
1) Affordability														
1.1) Food consumption as a share of household expenditure	45.3	64.8	61.6	58.6	57.2	60.4	60.8	34.5	29.0	41.4	49.3	46.0	38.3	51.6
1.2) Proportion of population under global poverty line	87.5	96.7	95.0	92.5	93.3	90.0	100.0	100.0	85.7	100.0	100.0	95.1	96.7	95.09
1.3) Gross domestic product per capita (PPP)	271	117	200	1,136	204	247	34	89	409	15	38	233	217	266.5
1.4) Presence of food safety net programs	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0
1.5) Access to financing for farmers	0	0	0	1	1	0	0	0	0	0	0	0	2	0.3
2) Availability														
2.1) Sufficiency of supply														
2.1.1) Average food supply	3495.8	3642.7	2820.7	2908.9	2646.9	2443.5	2558.5	2173.0	2542.6	1781.4	2297.7	2368.8	2558.9	2672.7
2.1.2) Dependency on chronic food aid	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0
2.2) Agricultural infrastructure														
2.2.1) Existence of adequate crop storage facilities	1	1	1	1	0	0	1	0	1	1	0	0	1	0.6
2.2.2) Road infrastructure	0	0	0	1	1	2	0	0	0	0	0	1	1	0.5
2.2.3) Port infrastructure	1	1	1	0	0	0	0	0	0	0	0	2	1	0.6
2.3) Corruption	1	1	2	3	2	2	1	3	0	0	0	1	2	1.6
2.4) Food loss	68.7	71.3	70.3	63.5	66.6	53.2	40.9	53.8	79.3	19.4	15.3	56.4	56.0	55.3
3) Quality and safety														
3.1) Diet diversification	37	33	35	42	35	47	29	33	21	15	21	46	61	37.0
3.2) Micronutrient availability														
3.2.1) Dietary availability of vitamin A	0.19	0.30	0.20	0.30	0.23	0.43	0.11	0.20	0.43	0.26	0.20	0.27	0.23	0.2
3.2.2) Dietary availability of animal iron	3.7	2.4	5.3	1.4	2.7	2.3	0.9	1.4	2.1	0.8	1.4	4.9	2.3	2.6
3.2.3) Dietary availability of vegetal iron	17.8	17.2	11.5	15.6	11.9	12.5	12.3	10.7	12.3	9.0	12.3	9.4	14.8	13.0
3.3) Protein quality	103.6	86.3	78.0	66.4	68.8	72.7	59.3	54.5	61.3	40.7	54.6	80.1	70.6	71.0
3.4) Food safety														
3.4.1) Percentage of population with access to potable water	100	100	100	100	100	100	100	100	100	100	0	100	100	92.2

(Continued)

TABLE 1 | Continued

	Village 1	Village 2	Village 3	Village 4	Village 5	Village 6	Village 7	Village 8	Village 9	Village 10	Village 11	Village 12	Village 13	Masoala average
3.4.2) Presence of a store	1	1	1	1	1	1	1	1	0	0	1	1	1	0.93
4) Background variables														
4.1) Prevalence of undernourishment	15.6	0.0	20.0	25.0	10.0	20.0	28.6	26.7	0.0	57.9	40.0	31.7	20.0	23.3
4.2) Child growth														
4.2.1) Percentage of children under 5 stunted	40.0	11.1	25.0	33.3	0.0	12.5	33.3	27.3	66.7	27.3	30.0	28.6	33.3	28.9
4.2.2) Percentage of children under 5 underweight	42.9	12.5	33.3	38.9	0.0	11.1	9.1	15.4	33.3	30.8	25.0	60.0	16.7	27.4
4.2.3) Percentage of children under 5 wasted	20.0	0.0	25.0	6.7	0.0	0.0	0.0	9.1	0.0	0.0	0.0	4.8	0.0	5.0
4.3) Anemia														
4.3.1) Percentage of pop anemic	22.2	16.8	32.7	31.2	17.7	19.5	22.4	14.6	23.3	10.4	22.5	40.1	27.0	24.5
4.3.2) Percentage of children under 5 anemic	16.7	18.2	27.8	27.8	40.0	18.2	20.0	16.7	28.6	11.1	17.7	27.8	40.0	23.7
4.4) Intensity of food deprivation	347.9	363.9	280.2	289.3	263.9	242.1	254.7	213.5	253.0	177.8	228.7	235.7	255.7	265.9
4.5) Prevalence of obesity	7.5	2.5	9.8	5.4	2.1	10.3	0.0	3.4	0.0	0.0	0.0	1.4	5.9	4.4
5) Forest provisioning services														
5.1) Availability														
5.1.1) Abundance of plant life	266846.6	228049.2	111555.9	182937.5	203982.2	106778.3	145019.3	221906.0	231200.1	234808.2	265571.2	43589.7	163031.0	175519.0
5.1.2) TBA trees	31828.3	34787.4	587.9	1382.4	12146.3	294.4	4891.5	7027.3	10024.4	3770.9	5520.7	9.4	334.6	8216.6
5.1.3) Biomass wildlife	1344.5	916.9	2103.0	1365.3	2641.3	8344.8	2602.7	2204.5	4511.5	4418.4	4052.4	8272.4	4944.8	3624.0
5.2) Quality														
5.2.1) Habitat SWDI	3.8	3.7	2.1	2.9	3.1	1.2	2.8	2.7	3.0	3.2	3.0	1.1	2.4	2.6
5.2.2) Wildlife SWDI	0.6	0.0	0.0	0.1	0.3	0.0	0.2	0.2	0.2	0.5	0.3	0.0	0.0	0.2
5.3) Contribution/Use														
5.3.1) Food supply														
5.3.1.1) Average vegetable food supply from forest	18.3	3.4	7.6	22.2	13.4	10.0	37.6	9.5	24.6	6.6	0.0	9.5	6.1	12.4
5.3.1.2) % of vegetable food supply from forest	0.4	0.1	0.3	0.9	0.5	0.4	1.8	0.3	1.2	0.4	0.0	0.4	0.2	0.5
5.3.1.3) Average animal food supply from forest	22.5	15.0	20.0	17.7	9.7	22.9	14.3	40.9	48.5	7.8	14.4	12.2	15.5	18.5
5.3.1.4) % of animal food supply from forest	20.3	17.3	18.1	43.9	39.2	18.8	73.7	61.4	75.2	61.2	55.1	10.5	31.9	36.2

(Continued)

TABLE 1 | Continued

	Village 1	Village 2	Village 3	Village 4	Village 5	Village 6	Village 7	Village 8	Village 9	Village 10	Village 11	Village 12	Village 13	Masoala average
5.3.1.5) % of all food supply from forest	1.2	0.5	1.0	1.4	0.9	1.3	2.0	2.3	2.9	0.8	0.6	0.9	0.8	1.2
5.3.2) Diet diversification														
5.3.2.1) Dietary availability of vitamin A from forest	0.2	0.2	0.1	0.1	0.1	0.0	0.0	0.1	0.4	0.1	0.1	0.0	0.1	0.1
5.3.2.2) Dietary availability of iron from forest	1.3	0.4	0.8	1.2	0.7	0.8	1.9	1.2	2.5	0.5	0.5	0.8	0.8	0.9
5.3.2.3) % of Dietary availability of iron from forest	5.1	2.2	3.7	6.5	3.3	5.4	8.5	5.8	14.3	5.1	4.0	4.9	4.9	5.1
5.3.3) Protein														
5.3.3.1) Protein quality from forest	3.5	1.6	2.3	2.7	1.5	2.2	3.8	4.0	8.3	1.4	2.0	2.2	2.9	2.6
5.3.3.2) % of quality protein from forest	3.9	1.9	2.3	4.0	2.0	2.6	5.7	4.5	9.9	3.7	3.8	2.6	4.0	3.5

† *The unit and method used to calculate each variable is explained in detail in* ***Table 2****.*

gain a more detailed and holistic perspective of household food security, we also used a modified version of the Economist Intelligence Unit's (EIU) food security index (Myers et al., 2000), with additional data collected on forest provisioning services (**Tables 1**, **2**).

We asked households about the quantity of 160 different types of food they ate, including 24 forest mammals (**Table 3**), during the prior 24-h, week, month, and year (depending on how frequently the food was regularly consumed) and used data from the Food and Agriculture Organization of the United Nations (FAO) (FAO, 2012) and GENuS (Smith et al., 2016) to calculate dietary nutrient intake. We converted all household members into their adult-equivalent score using FAO guidelines (Food Agriculture Organization, 2004; Weisell and Dop, 2012). We recorded the quantities of foods eaten in local serving sizes and the weighed each serving size 10 times to determine a mean weight for calculations. For rarely consumed animals, we calculated the mean body mass of animals using previously published data (Goodman, 2011, 2012; Soarimalala and Goodman, 2011; Borgerson, 2015, 2016). Dietary diversity was measured using the Women's Dietary Diversity Scale (WDDS) (FAO, 2010).

Measurements of Human Health

During interviews, CB and BJRR collected 27 indicators of human health from 1,750 individuals aged 2 weeks to 91 years old (all available members of the 387 interviewed households). We measured individual height and weight, and used non-invasive photospectrometry to assess oxygen saturation and hemoglobin status using a MASIMO Pronto 7 hemoglobinometer. We also asked about each individual's overall morbidity. We used WHO and CDC guidelines to determine whether individuals were stunted, underweight, wasted, had severely low BMIs, or were anemic (WHO, 2006, 2011; Centers for Disease Control Prevention, 2012).

Lemur and Bird Surveys

BR used distance sampling methods (Buckland et al., 1993; Buckland, 2001) to assess the density, biomass, abundance, and population demographics of all diurnal lemur and bird species (groups of animals which are hunted and can be reliably surveyed using distance sampling methods). We established a total of 172 kilometers of transects (using a GPS) on the peninsula. Each of the 13 village sites contained two 2 km long transects. BR and a local field assistant walked each transect line at a maximum rate of 1 km/h, a minimum of 20 times. Two additional transects (total of 140 areal km in length) extended from the western border of the Masoala National Park through the interior, and ended at the eastern border, through the parks northern and southern regions. Each interior transect was walked twice. Each time we saw a lemur or bird, we recorded the age class, sex, group size/composition, height (m), angle, and the perpendicular distance (m) of the animal (or the center of the group of animals) from the transect line. We used published and estimated body weight data (Goodman, 2011, 2012; Soarimalala and Goodman,

TABLE 2 | Units and methods used to calculate Masoala food security variables (2015).

Variable	Basis of calculation
1) Affordability	
1.1) Food consumption as a share of household expenditure	% of total household expenditures that were spent on food during prior week from household weekly dietary and economic recall surveys
1.2) Proportion of population under global poverty line	% of population (individuals) living under $2/day PPP (exchange rate 3,000 MGA:$US1) from household and individual annual income recall surveys
1.3) Gross domestic product per capita (PPP)	Mean cash income last year in US$ at PPP / capita from household and individual annual income surveys
1.4) Presence of food safety net programmes	Village qualitative assessment (0–4) from village group interviews, e.g., Seecaline, Care, school breakfasts
1.5) Access to financing for farmers	Village qualitative assessment (0–4) from village group interviews, e.g., OTIV, CARE
2) Availability	
2.1) Sufficiency of supply	
2.1.1) Average food supply	Mean calories/adult male equivalent/day calculated from household 24 h dietary recalls
2.1.2) Dependency on chronic food aid	Village qualitative assessment (0–2) from village group interviews
2.2) Agricultural infrastructure	
2.2.1) Existence of adequate crop storage facilities	Village qualitative assessment (0–1) from village group interviews
2.2.2) Road infrastructure	Village qualitative assessment (0–4) from village group interviews (incl. quality of roads, transportation availability, quality, frequency, cost, and capacity)
2.2.3) Port infrastructure	Village qualitative assessment (0–4) from village group interviews (incl. transportation availability, quality, frequency, cost, and capacity)
2.3) Corruption	Rating 0–4; 4 = highest risk from village group interviews (incl. frequency and perceived severity)
2.4) Food loss	Total % of chickens that died of disease from annual household recalls of livestock loss
3) Quality and safety	
3.1) Diet diversification	Average % of grams of diet from non-starchy foods calculated from household 24 h dietary recalls
3.2) Micronutrient availability	
3.2.1) Dietary availability of vitamin A	Proportion of population that ate foods high in Vit A in the last 24 h, from 24 h dietary recall surveys
3.2.2) Dietary availability of animal iron	Average mg/person/day calculated from household 24 h, weekly, and annual diet dietary recalls
3.2.3) Dietary availability of vegetal iron	Average mg/person/day calculated from household 24 h, weekly, and annual dietary recalls
3.3) Protein quality	Average grams/person/day calculated from household 24 h, weekly, and annual diet dietary recalls
3.4) Food safety	
3.4.1) Percentage of population with access to potable water	% of population with access to clean rivers (no upstream villages) or improved water sources (wells, pumps, etc.) from village group interviews
3.4.2) Presence of a store	Village qualitative assessment (0–2) (0 no stores, 1 stores, 2 bazaar) from village group interviews
4) Background variables	
4.1) Prevalence of undernourishment	% of households whose members (adult male equivalent) do not receive the min calories/person as defined by the FAO/WHO/UNU expert consultation 2001, mean age 21.11 = 1,680 cal
4.2) Child growth	
4.2.1) Percentage of children under 5 stunted	% of children under 5 whose height is more than two standard deviations below the mean for their age and sex, using WHO standards, calculated from individual health assessments
4.2.2) Percentage of children under 5 underweight	% of children under 5 whose weight is more than two standard deviations below the mean for their age and sex, using WHO and CDC standards, calculated from individual health assessments
4.2.3) Percentage of children under 5 wasted	% of children under 5 whose weight for height is more than two standard deviations below the mean for their age and sex, using WHO and CDC standards, calculated from individual health assessments
4.3) Anemia	
4.3.1) Percentage of pop anemic	% of total population with Hb levels below the min recommended range for their respective age and sex, calculated from individual health assessments
4.3.2) Percentage of children under 5 anemic	% of children under 5 with Hb levels below the min recommended range for their respective age and sex, calculated from individual health assessments
4.4) Intensity of food deprivation	Average kcal/person/day calculated from household 24 h, weekly, and annual diet recall surveys
4.6) Prevalence of obesity	% of population over age 20 with a BMI >30, calculated from individual health assessments
5) Forest provisioning services	
5.1) Availability	
5.1.1) Abundance of plant life	Mean N plants per hectare, calculated from habitat plots

(Continued)

TABLE 2 | Continued

Variable	Basis of calculation
5.1.2) TBA trees	TBA of trees >3 cm in diameter per hectare, calculated from habitat plots
5.1.3) Biomass wildlife	Estimated biomass (in grams) of diurnal lemur and birds per hectare, calculated from transects
5.2) Quality	
5.2.1) Habitat SWDI	SWDI of trees over 10 cm in diameter, calculated from habitat plots
5.2.2) Wildlife SWDI	SWDI of diurnal wildlife species, calculated from transects
5.3) Contribution/Use	
5.3.1) Food supply	
5.3.1.1) Average vegetable food supply from forest	Mean kcal/capita/day calculated from household 24 h, weekly, and annual diet dietary recalls
5.3.1.2) % of vegetable food supply from forest	% of total kcal/capita/day from vegetable/fruit products calculated from household 24 h, weekly, and annual diet dietary recalls
5.3.1.3) Average animal food supply from forest	Mean kcal/capita/day calculated from household 24 h, weekly, and annual diet dietary recalls
5.3.1.4) % of animal food supply from forest	% of total kcal/capita/dayfrom animal products calculated from household 24 h, weekly, and annual diet dietary recalls
5.3.1.5) % of all food supply from forest	% of total kcal/capita/day from all food and drinks calculated from household 24 h, weekly, and annual diet dietary recalls
5.3.2) Diet diversification	
5.3.2.1) Dietary availability of vitamin A from forest	Proportion of population that ate foods high in Vit A yesterday from forest, calculated from household 24 h dietary recalls
5.3.2.2) Dietary availability of iron from forest	% of total mg/person/day, calculated from household 24 h dietary recalls
5.3.2.3) % of Dietary availability of iron from forest	Mean mg/person/day calculated from household 24 h dietary recalls
5.3.3) Protein	
5.3.3.1) Protein quality from forest	Mean grams/person/day calculated from household 24 h, weekly, and annual diet dietary recalls
5.3.3.2) % of quality protein from forest	% of total grams/person/day calculated from household 24 h, weekly, and annual diet dietary recalls

2011; Borgerson, 2015, 2016) for different age/sex classes to determine biomass.

Habitat Sampling

DR and a local assistant collected botanical information using 156 forest plots, each 20 m in diameter. We employed the same 15 transect lines used for wildlife surveys for habitat sampling. At each of the village sites, 10 habitat plots were established in 200 meters increments at a 20 m distance from each of the wildlife transects. For the interior transects, plots were located every 5 km. Each plot was composed of three concentric circles. In the first circle (1 m radius), we sampled all small plants, i.e., woody seedlings and herbaceous ground cover with a diameter <2.5 cm, and estimated the percentage of ground cover for each species. In the second circle (3 m radius) we identified, counted, and measured the diameter at breast height (DBH) and height of woody stems of all medium plants, i.e., shrubs, saplings, and woody and herbaceous climbers (vines and lianas) between 2.5 and 10 cm in diameter. In the third circle (10 m radius), we identified the local species name of each large plant with a DBH ≥10 cm, and measured its DBH, crown width, height, bole height, and angle and distance from the plot center.

RESULTS

Food Security

We found a high prevalence of both food insecurity and poverty on the Masoala Peninsula with 78% of Malagasy households being highly food insecure, experiencing food insecurity a mean of 3.26 days during the prior week [CSI mean 4.88 (range: 0–42); weighted CSI mean 11.74 (range: 0–99)]. We recorded data across 8 months which included prior week recalls of food insecurity during seasons of both low and high food security. The most frequently reported mechanisms for coping with food insecurity during the prior week were to: limit the portion size of all household members (16% of all incidences of coping strategies used); eat at the households of friends or family (13%); and rely on less preferred or less expensive foods (12%).

The mean household HFAIS score was 2.86 (household range 0–11). Participants reported two lean seasons, one in the austral winter and one in the summer, just before each rice harvest. The winter lean period was reported to be the most severe. During the prior year, nearly half of households (43.9%) worried that their household would not have enough food. Because of a lack of food-resources, 38.0% were unable to eat preferred foods and a third of households ate just a few kinds of food day after day (30.2%). A quarter of households ate less during meals than needed (27.1%) and believed they lost weight because of inadequate food (24.6%). One in every five households experienced a time during the prior year when there was no food at all in their household and/or fields and no financial resources to acquire more (19.4%), and/or reduced the number of meals they ate in a day (17.6%). Nearly one in ten (8.0%) households sold assets, land, or livestock to buy food during the prior year.

Over 95% of the surveyed population was under the global poverty line (defined as the percentage of individuals living

TABLE 3 | Consumption and hunting of forest mammals during the prior year by members of 387 households in 13 communities surveyed near the Masoala National Park (2015).

Species	Mean eaten per household	Total number eaten†	Eaten as a guest	Opportunistic hunting	Pursuit hunting	Trapping	Purchasing	Unknown provenance	Price per whole animal
	(N)	(N)	(N)	(N)	(N)	(N)	(N)	(N)	(USD)
Lemurs	**0.62**	**240**	**19**	**58**	**31**	**103**	**17**	**12**	
Eulemur albifrons	0.35	136	10	7	22	79	9	9	2.70
Varecia rubra	0.05	19	4	0	5	3	7	0	6.00
Microcebus sp.	0.02	9	0	9	0	0	0	0	–
Allocebus trichotis	0.01	2	0	2	0	0	0	0	–
Cheirogaleus sp.	0.05	19	0	16	0	2	0	1	–
Phaner furcifer	0.00	1	0	0	1	0	0	0	–
Lepilemur scottorum	0.01	5	1	4	0	0	0	0	–
Hapalemur griseus	0.06	24	2	7	0	14	1	0	1.00
Avahi mooreorum	0.06	24	2	12	3	5	0	2	–
Daubentonia madagascariensis	0.00	1	0	1	0	0	0	0	–
Carnivorans	**0.22**	**85/100**	**18**	**23**	**8**	**48**	**3**	**0**	
Galidia elegans	0.06	25	1	13	5	6	0	0	
Galidictis fasciata	0.00	1	0	0	0	1	0	0	
Salanoia concolor	0.01	4	0	3	0	1	0	0	
Cryptoprocta ferox (min whole animals caught/portions)	0.06	23/38	15	4	3	14	2	0	0.33 per kg
Fossa fossana	0.03	13	0	1	0	12	0	0	–
Eupleres goudotii	0.05	19	2	2	0	14	1	0	1.00
Tenrecs	**4.53**	**1,755**	**116**	**1,031**	**532**	**8**	**47**	**21**	
Tenrec ecaudatus	4.13	1599	115	913	500	8	47	16	0.36
Setifer setosus	0.18	71	1	53	12	0	0	5	–
Hemicentetes semispinosus	0.22	85	0	65	20	0	0	0	–
Bats	**0.57**	**222**	**17**	**7**	**63**	**38**	**93**	**4**	
Pteropus rufus	0.52	201	17	0	49	38	93	4	0.88
Rousettus madagascariensis	0.02	9	0	7	2	0	0	0	–
Microchiroptera sp.	0.03	12	0	0	12	0	0	0	–
Introduced species	**1.05**	**407**	**17**	**1**	**53**	**105**	**224**	**7**	
Potamochoerus larvatus††	0.90	347	12	0	53	54	224	4	1.15
Viverricula indica	0.16	60	5	1	0	51	0	3	–
Total mammals	**7.00**	**2,709/2,724**	**187**	**1120**	**687**	**302**	**384**	**44**	

† *Total number of whole animals of each species eaten by the 387 households in 13 communities during the prior year.*
†† *Pieces of meat weighing a mean of 1.1 kg.*
The bold text is the total number eaten in each animal group.

under $2 per person per day) (**Tables 1**, **2**), and cash income was primarily spent on food (Mean = 51.7%). Of food expenses, 73.2% of these were used to purchase ingredients for a meat or vegetable sauce to complement their rice staple. Comparatively, only 22% was spent on rice, 1% on tubers, and 4% on snacks.

Nutrition, Dietary Diversity, and Health

Overall nutrition was poor and dietary diversity was low (**Tables 1**, **2**). A mean of 37.0% of all grams of food eaten during the prior week were from non-starchy foods. The diets of most households (77.3%) were moderately diverse during the prior week (**Tables 1**, **2**, **4**). One in four households ate any food high in Vitamin A within the previous 24 h (**Tables 1**, **2**), and individuals ate a mean of 2.63 mg of iron from animal sources, 12.97 mg from all other food sources, and 70.97 g of quality protein per day.

We found high levels of stunting, underweight, wasting, and anemia and a moderate to high prevalence of anemia throughout all sub-populations measured on the Masoala (**Tables 5**, **6**). There was notable variation between communities in these variables. Focusing on children under five, some communities reached levels of 67% stunting, 60% underweight, 25% wasting, and 40% anemia (**Tables 1**, **5**). On average, however, the severity of the prevalence of stunting, underweight, wasting, and anemia in children under five on the peninsula is classified as medium for stunting, high for children who are underweight, and medium for wasting (**Table 6**; WHO, 2012).

Natural Resource Use

Local people reported a high reliance on the forest for food. Surveyed households ate 6,726 forest animals during the prior year (**Table 3**). Of the micro- and macro-nutrients provided by domestic and wild animal products (including fish, eggs, insects, honey, etc.), a mean of 36.2% of all calories (kcal) from animal products, 44.5% of all animal iron, and 38.4% of animal protein came from wild forest animals (**Tables 1**, **2**). There was significant variation among communities. In some communities, as much as 75.2% of animal products came from forest animals, while in others, as little as 10.5% did (**Tables 1**, **2**). People relied less on the forest for vegetables than they did for meat. Only 0.5% of all vegetables (kcal) eaten came from the forest, and communities were similar in their reliance on the forest for vegetable foods (**Tables 1**, **2**). In total, an average of 1.2% of all kcal consumed per day came from forest products (village range = 0.5–2.9%).

Members of almost all households reported eating the meat of forest animals during the prior year (89.1%); 73.4% had eaten at least one forest mammal and 72.9% at least one forest bird. Household members ate a mean of 6.9 forest mammals and 10.4 forest birds, or 14.56 kg of wild meat per household per year. Nearly a third of households (30.0%) had eaten a threatened mammal during prior year. These households ate a mean of 1.3 threatened mammals. Tenrecs were eaten by the greatest percentage of households (50.9%), followed by the meat of bushpigs (34.9%), lemurs (18.9%), euplerids (13.7%), introduced carnivorans (10.6%), and bats (10.6%). The vast majority of catch was eaten by members of the hunter's own household and was not sold (**Table 3**). Of the 6,726 forest animals eaten, 40.3% were mammals and 59.7% were birds. Tenrecs were the most frequently caught forest mammal (64.8% of the total number of forest mammals caught), followed in number by bushpigs (12.8%), lemurs (8.9%), bats (8.2%), native euplerid carnivorans (3.1%), and introduced carnivorans (2.2%) (**Table 3**). Because the hunting of many species is prohibited, the hunting and consumption of many animal species is likely under-reported. Actual levels of hunting are likely higher than reported here.

Eighty-six percent of measured households ate any kind of fish or meat during the prior week, and 57.1% ate the meat of domestic animals. Chickens were the most commonly owned domestic livestock, followed by ducks (**Table 7**). Yet, over three-quarters of poultry (79.0%) died during the prior year from an illness consistent with the symptoms and timing of Newcastle disease.

TABLE 4 | Foods characterizing diets with low, moderate, and high diversity using a WDDS scale†.

Low dietary diversity (WDDS 0–3) 1.6% of households	Moderate dietary diversity (WDDS 4–6) 77.3% of households	High dietary diversity (WDDS 7–9) 2.6% of households
Starchy staples	Starchy staples	Starchy staples
Dark green leafy veg.	Dark green leafy veg.	Dark green leafy veg.
	Other fruits and veg.	Other fruits and veg.
	Fish/seafood and meat	Fish/seafood and meat
		Organ meat
		Eggs
		Legumes, nuts, seeds

†*Food categories listed were found in >75% of households in that subclass.*

Biodiversity

The availability and quality of habitat varied greatly between villages (**Tables 1**, **2**). A total of 0.64 km^2 of forested land were cleared for new (not in fallow) agricultural lands over the prior year; 13.7% of households cleared this land at a mean distance

TABLE 5 | Percentages of individuals (n = 1,750) classified as stunted, underweight, wasted, and anemic in the 13 communities surveyed near the Masoala National Park (2015)†.

Age range (yrs)	Sex	Sample size *(n)*	Underweight	Stunted	Wasted	Sample size *(n)*	Anemic
0 < 6	Male and Female	192	27.87	33.33	4.76	215	19.53
6 < 13	Male and Female	192	22.92	24.85	5.88	312	12.18
13 < 21	Male	104	25.96	27.66	7.45	188	29.51
13 < 21	Female	87	35.63	33.33	8.00	153	45.75
0 < 21	Male and Female	859	25.61	31.32	6.18	–	–
0 < 21	Male	433	23.09	29.79	5.43	–	–
0 < 21	Female	424	28.30	32.89	6.95	–	–
21 < 55	Male and Female	–	–	–	–	466	24.03
55+	Male and Female	–	–	–	–	70	42.86
0+	Male and Female	–	–	–	–	1,338	24.51
0+	Male	–	–	–	–	613	17.33
0+	Female	–	–	–	–	725	30.69

†*Children are defined as stunted, underweight, or wasted if their height-for-age, weight-for-age, or weight-for-height is more than two standard deviations below the CDC (2000) or WHO (2006) Child Growth Standards median.*

of 81 min from their home. Households cleared an average of 12,065 m^2 of land (or 1,652 m^2 when including households that did not clear land); 84.2% of this land was subsequently used for subsistence agriculture, 11.3% for cash crops, 1.9% for livestock, and 1.9% was sold.

Lemurs were more abundant and found in larger cluster sizes within the Masoala National Park than near villages. *Varecia rubra* were found throughout the park interior at a mean density of 8.3 animals per square kilometer (expected cluster size = 5.6), and *Eulemur albifrons* were found at a mean density of 58.1 animals per square kilometer (expected cluster size = 5.5). *V. rubra* were present at five of the thirteen village sites at densities ranging from 0–13.6 per km^2. The mean density of *V. rubra* at all village sites was 2.9 animals per square kilometer (expected cluster size = 2.2). *E. albifrons* were present at eight of the thirteen village sites at densities ranging from 0 to 91.3 animals per square kilometer. The mean density of *E. albifrons* at all village sites was 16.5 animals per square kilometer, with an expected cluster size of 4.3.

Forest plots were significantly richer, in all plant size classes, within the Masoala National Park than near villages (Small plants: $T = 17.56$, DF = 151, $P < 0.0001$; Medium plants: $T = 16.66$, DF = 151, $P < 0.0001$; Large plants: $T = 14.40$, DF = 151, $P < 0.0001$). Medium and large plants also had significantly larger DBH (Medium plants: $T = 5.33$, DF = 1,036, $P < 0.0001$; Large plants: $T = 2.29$, DF = 1,315, $P = 0.02$) and were significantly taller (Medium plants: $T = 7.21$, DF = 1,036, $P < 0.0001$; Large plants: $T = 22.06$, DF = 1,315, $P < 0.0001$). There were significantly more medium and large plants and significantly fewer small plants in forest plots within the Masoala National Park than those plots near villages (Small plants: $T = 4.11$, DF = 151, $P < 0.0001$; Medium plants: $T = 11.82$, DF = 151, $P < 0.0001$; Large plants: $T = 6.10$, DF = 151, $P < 0.0001$). Further, there were 10 times as many stems of small plants whose identity was unknown within the park. These differences in habitat resulted in a total basal area of plants per hectare 50% larger and a total available crown area per hectare twice as large within the park than outside of it (Total basal area: 9,027.13 vs. 6,927.63 m^2; Total available crown area: 19,112.39 vs. 7,606.61 m^2).

TABLE 6 | The percentage of communities measured ($n = 13$) experiencing different levels of severity for child malnutrition (under age 5) on the Masoala Peninsula (2015) using WHO standards (2017).

Measurement of malnutrition	Low severity %	Medium severity %	High severity %	Very high severity %
Stunting	30.8	30.8	30.8	7.7
Low weight	15.4	30.8	7.7	46.2
Wasting	69.2	7.7	0.0	15.4

DISCUSSION

The future of lemurs, 94% of which are threatened with extinction (Schwitzer et al., 2013), depends on the sustainable diets of a malnourished human population who commonly hunts them for food (Borgerson, 2015; Borgerson et al., 2016). We found a high prevalence of both food insecurity and poverty surrounding the Masoala National Park, a UNESCO World Heritage Site. Eighty percent of households on the Masoala experienced food insecurity over the course of a year and 95% lived in persistent poverty. Both food insecurity and poverty on the Masoala are higher than national averages (WHO, 2012), which already place Madagascar as the third least food secure nation in the world (Economist Intelligence Unit, 2017). The prevalence of poverty on the Masoala exceeds that of Burundi and the Democratic Republic of Congo, the only two nations that are ranked below Madagascar in food security, and, unlike Madagascar, are recovering from violent civil wars (Economist Intelligence Unit, 2017).

Children on the Masoala were highly malnourished and one-quarter of the population was anemic, far higher than in other regions of Madagascar (Mould et al., 2016). In order to meet the requirements for a healthy diet, local people relied on the forest for food, yet still often failed to meet these objectives. While wild meats are not a staple food, most households depended on them. Wildlife consumption was common throughout the region; 89% of households ate wildlife within the prior year on the Masoala, nearly twice the prevalence of other regions worldwide (Rowland et al., 2016). Those surveyed ate 6,726 forest animals (mean of 7 mammals and 10 birds per household), or ∼3.27 kg of wild meat per person (4.48 kg per adult equivalent) during the prior year. This amount of wildlife is much higher than that reported in other regions of Madagascar including Kianjavato (Borgerson et al., 2018a), Alaotra (Borgerson et al., 2018b), and Betampona (Golden et al., 2014b), but it is similar to that reported in nearby Makira (Golden et al., 2014a; Brook et al., 2019), and far less than the amount eaten in the Amazon and Congo basins (63 and 51 kg/capita/year, respectively; Nasi et al., 2011). Further, as much as 75% of all meat eaten in some communities was from forest animals. People ate very low quantities of iron and protein and in some communities as much as 76% of protein and 74% of iron came from forest meats.

Healthy forests can support the food security of the most vulnerable households in a region by directly supplying a wide variety of wild foods that increase the quality, security, and diversity of local diets. The meat of wild animals provided

TABLE 7 | The range and mean of household livestock assets on the Masoala Peninsula (2015).

Type of livestock	Cows	Pigs	Ducks	Chickens	Geese	Cats	Dogs
Range (*n* per household)	0–18	0–12	0–43	0–69	0–41	0–6	0–5
Mean (*n* per household)	1.19	0.20	2.73	9.62	0.42	0.40	0.41

valuable micronutrients to people experiencing food insecurity and malnutrition, yet the hunting of threatened species is an untenable solution to food insecurity. Further, continued habitat loss in the region will likely only reduce access to wild foods in the future and increase the reliance on less-diverse foods (Powell et al., 2011; Sunderland, 2011; Sunderland et al., 2017). In order to improve human well-being in the long term, forests must be conserved and hunting must be reduced to sustainable levels.

Conservation efforts to reduce the unsustainable hunting of threatened species are unlikely to alter the behavior of hunters unless they address the goals, reasons, and incentives for hunting. Improving food security can increase the sustainability of hunting, improving both forest conservation and human nutrition in the long term, allowing for forests to provide essential services to those who live near them. Yet, few micronutrient-rich alternatives to wild meats were available in adequate supply and many were highly volatile; 79% of all chickens died from Newcastle disease in the prior year. In order to reduce the unsustainable hunting of threatened species while improving household food security and child health, we suggest testing the effects of increasing the affordability, accessibility, and stability of micro-nutrient rich animal-sourced foods in remote communities, where forests contribute the most to food security.

In conclusion, we believe that by using an integrated approach to improve food security in one of the world's most biodiverse and least food secure nations, we can both conserve Madagascar's unique biodiversity and improve the nutrition and health of Madagascar's people.

ETHICS STATEMENT

We collected this data during May until December 2015 and all research was approved by Human Subjects Institutional Review Boards (Protocols #15-0331 Wildlife Conservation Society and #15-2230 Harvard T. H. Chan School of Public Health), the Republic of Madagascar and Madagascar National Parks (Permits 111/13, 325/14, 111/15, 218/15, 270/15, /MEEF/SG/DGF/DCB.SAPP/SCB). We obtained oral informed consent and/or assent from all participants.

AUTHOR CONTRIBUTIONS

CB, BR, DR, and BJRR collected the data. CB designed the study and completed data analysis. CB and CG interpreted the data and wrote this manuscript. All authors approved the final manuscript.

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Testing the Various Pathways Linking Forest Cover to Dietary Diversity in Tropical Landscapes

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A diverse diet is important to address micronutrient deficiencies and other forms of malnutrition, one of the greatest challenges of today's food systems. In tropical countries, several studies have found a positive association between forest cover and dietary diversity, although the actual mechanisms of this has yet to be identified and quantified. Three complementary pathways may link forests to diets: a direct pathway (e.g., consumption of forest food), an income pathway (income from forest products used to purchase food from markets), and an agroecological pathway (forests and trees sustaining farm production). We used piece-wise structural equation modeling to test and quantify the relative contribution of these three pathways for households in seven tropical landscapes in Bangladesh, Burkina Faso, Cameroon, Ethiopia, Indonesia, Nicaragua, and Zambia. We used survey data from 1,783 households and determined forest cover within a 2-km radius of each household. The quality of household diets was assessed through four indicators: household dietary diversity and consumption of fruits, vegetables, and meat, based on a 24-h recall. We found evidence of a direct pathway in four landscapes (Bangladesh, Cameroon, Ethiopia, and Zambia), an income pathway in none of the landscapes considered, and an agroecological pathway in three landscapes (Bangladesh, Ethiopia, and Indonesia). We also found evidence of improved crop and livestock production with greater forest cover in five landscapes (Bangladesh, Burkina Faso, Cameroon, Ethiopia, and Indonesia). Conversely, we found negative associations between forest cover and crop and livestock production in three landscapes (Cameroon, Indonesia, and Zambia). In addition, we found evidence of forest cover being negatively related to at least one indicator of diet quality in three landscapes (Indonesia, Nicaragua, and Zambia) and to integration to the cash economy in three landscapes (Cameroon, Ethiopia, and Nicaragua). This is one of the first studies to quantify the different mechanisms linking forest cover and diet. Our work illuminates the fact that these mechanisms can vary significantly from one site to another, calling for site-specific interventions. Our results also suggest that the positive contributions of forests to rural livelihoods cannot be generalized and should not be idealized.

Keywords: nutrition, hidden hunger, multifunctional landscapes, ecosystem services, structural equation modeling

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INTRODUCTION

An estimated two billion people are currently affected by "hidden hunger," i.e., micronutrient deficiencies (Muthayya et al., 2013; IFPRI, 2014; HLPE, 2017). Poor quality diet is now one of the leading risk factors for mortality globally (Afshin et al., 2019; Willett et al., 2019) and nutrition-related chronic diseases are increasingly a problem in developing counties (IFPRI, 2014). Poor diet quality contributes to both micronutrient deficiency and chronic nutrition-related diseases. Dietary diversity is increasingly accepted as a good measure of diet quality (Foote et al., 2004; Steyn et al., 2006; Kennedy et al., 2010). Dietary diversity may increase with improved market access (Sibhatu et al., 2015). However, protein-rich and micronutrient-rich food can be several times more expensive than staple food, particularly in low-income countries (Headey and Masters, 2019). In such countries, with largely rural population, dietary diversity may thus be improved through diversity in farm production (Jones et al., 2014; Powell et al., 2015; Jones, 2017).

Several studies have also found more diverse and nutritious diets consumed by people living in or near areas with greater tree cover (Dounias and Froment, 2006; Powell et al., 2011; Johnson et al., 2013; Ickowitz et al., 2014; Baudron et al., 2017; Galway et al., 2018; Rasolofoson et al., 2018). Three main complementary pathways may link forest cover to dietary diversity: (1) a "direct pathway," (2) an "income pathway," and (3) an "agroecological pathway" (**Figure 1**). (1) Forests may contribute directly to people's diets through the harvest of bushmeat, wild fruits, wild vegetables, and other forest-sourced foods (Hladik et al., 1990; Fa et al., 2003; Vinceti et al., 2008; Nasi et al., 2011; Termote et al., 2011; Powell et al., 2015; Rowland et al., 2017). (2) The sale of non-timber forest products, and timber to a lesser extent, may contribute to people's income (Williams, 1998; Beck and Nesmith, 2001; Kaschula et al., 2005; Pfund et al., 2011; Angelsen et al., 2014), potentially leading to the purchase of a diversity of food items from markets. (3) Finally, forests and trees may support diverse crop and livestock production through an array of ecosystem services (Reed et al., 2017) such as maintenance of soil fertility and water regulation (Young, 1989; Sanchez et al., 1997; Ong et al., 2000), pollination (Garibaldi et al., 2011), pest control (Dix et al., 1995), and regulation of micro- and regional climate (Zheng and Eltahir, 1998; Fu, 2003; Shiferaw Sida et al., 2018). Forests may also be grazed and sustain livestock production (Baudron et al., 2017). An additional aspect of this agroecological pathway may come from the availability of fuelwood from forests allowing the production of nutritious crops, which, on average, require a long cooking time, e.g., pulses (Wan et al., 2011; Remans et al., 2012). The availability of fuelwood from forests may also result in the use of more crop residues and livestock dung as soil amendment rather than as fuel, with positive impact in soil fertility and crop diversity (Baudron et al., 2017).

While there is a growing body of evidence in support of each of the above pathways, their relative importance to each other remains poorly understood. This is the first study to our knowledge to attempt to quantify the relative contribution of different pathways. The objective of this study was to test and quantify the various pathways linking forest cover to dietary diversity—direct, income, and agroecological—using piece-wise structural equation modeling spanning seven contrasting tropical landscapes with a novel combination of household and forest cover information. Due to the importance of these food groups for adequate nutrition and because they are most commonly missing in households with low dietary diversity, the linkages between forest cover and the consumption of (1) fruit, (2) vegetable, and (3) meat (and other animal products, excluding dairy products) were also tested.

MATERIALS AND METHODS

Household Survey Data From Seven Study Sites

We use previously published and publicly available household survey data from the Agrarian Change Project implemented by the Center for International Forest Research (CIFOR)[1]. This dataset was collected through a standardized questionnaire that addressed household composition, dietary diversity, crop and livestock management, and income. Seven tropical landscapes spanning three continents were selected for the study: (1) the Bosawas Biosphere Reserve in Nicaragua, (2) Cassou District in Burkina Faso, (3) Nguti District in Cameroon, (4) Arsi Negele in Ethiopia, (5) Nyimba District in Zambia, (6) Chittagong Hill Tracts Region in Bangladesh, and (7) Kapuas Hulu Region in Indonesia (**Figure 2**). While we will refer to these locations by their respective country names in the rest of the paper, it should be noted they are not representative of national-level conditions. Although each landscape is very different in some respects (e.g., differing forest types, levels of biodiversity, agricultural practices, market influence, and forest dependency; **Table 1**), the main characteristic comparable across all seven landscapes is that they exemplify clear gradients of agricultural expansion and intensification across the forest transition (Deakin et al., 2016; Sunderland et al., 2017). In this regard, they are representative of similar sites throughout the tropics exhibiting rapid rural change.

The data were collected between December 2014 and August 2016 from 275 farming households in Bangladesh, 281 in Burkina Faso, 242 in Cameroon, 219 in Ethiopia, 239 in Indonesia, 253 in Nicaragua, and 274 in Zambia, for a total of 1783 households (see survey questionnaire in **Supplementary Material**). In each landscape, households were selected using a stratified random sampling scheme across a gradient of forest-agricultural intensification (see Sunderland et al., 2017). As such, approximately a third of households were distributed in each of three zones: relatively high tree cover/low level of agricultural intensification; relatively low tree cover/high agricultural intensification; and intermediate tree cover/agricultural intensification. While differences between zones were not the focus of this analysis, it is possible that this stratified sampling introduced confounding social, cultural, or economic factors our analysis was not completely able to control for (such as differences in diet between social–ecological

[1]https://data.cifor.org/dataset.xhtml?persistentId=doi:10.17528/CIFOR/DATA.00101

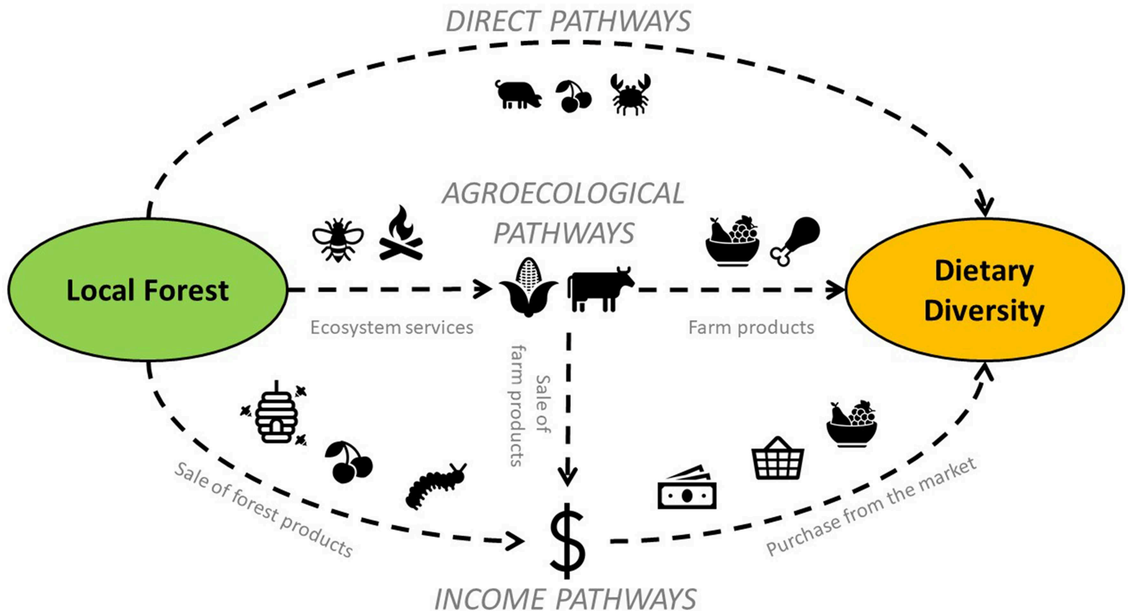

FIGURE 1 | Conceptual diagram representing the three main pathways linking forest cover to dietary diversity.

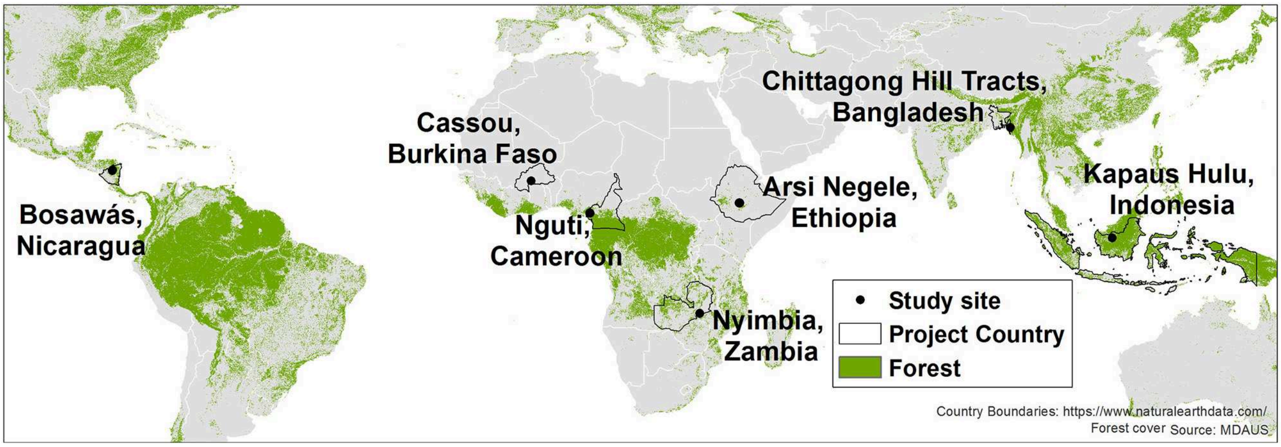

FIGURE 2 | Location of the seven study landscapes. For the global context, forest cover shown here is derived from the commercially available MDA US dataset produced from Landsat 8. However, for our analysis, we conducted site-specific forest cover mapping to ensure locally accurate results.

systems of different ethnic groups) (see Sunderland et al., 2017 for more description).

The survey data contained information on presence or absence of a home garden, total area farmed (as estimated by the head of the household, and referred to as "farm area" in the rest of the paper), numbers of different livestock species, ownership of various assets, main sources of income, and consumption or not of 11 food groups in the household during the 24 h that preceded the survey: (1) cereals, grains, and cereal products; (2) roots and tubers; (3) pulses and nuts; (4) vegetables; (5) meat and animal products; (6) fruits; (7) milk and milk products; (8) oils and fats; (9) sugar, sugar products, and honey; (10) spices and condiments; and (11) snacks and processed foods. These 11 food groups were used to construct dietary diversity scores following the Food and Agriculture Organization of the United Nations' Household Dietary Diversity Score (HDDS; Kennedy et al., 2010), modified to match food groups used in another research project (the Sentinel Landscapes Project, https://www1.cifor.org/sentinel-landscapes/home.html). The 24-h household dietary diversity score is referred to as "dietary diversity" in the rest of the paper, and the consumption of fruits, vegetables, and meat and other animal products, excluding dairy products, in the 24-h preceding the interview are referred to

TABLE 1 | Description of the seven study landscapes.

	Chittagong Hill Tracts Region (Bangladesh)	Cassou District (Burkina Faso)	Nguti District (Cameroon)	Arsi Negele (Ethiopia)	Kapuas Hulu Region (Indonesia)	Bosawas Biosphere Reserve (Nicaragua)	Nyimba District (Zambia)
Natural vegetation	Tropical wet evergreen/semi-evergreen and deciduous forest	Mixture of gallery forest, tree savannah, shrub savannah and grass savannah	Atlantic Biafran forest	Dry afromontane Forest	Tropical forest	Humid broadleaf evergreen forest	Mopane woodland, scrub woodland, Zambezian miombo woodland
Elevation (m.a.s.l.)	300–1,000	300	1,600	2,050–2,214	25–500	150–580	300–800
Precipitation (mm)	2,540	800–1,000	2,864	1,075	3530	2,500–3,500	700–900
Average temperature (°C)	23	27.6	26	15	27	25.7	29
Population density	120	35	47	142	7.7	14	8.1
Main crop	Rice, maize, fruits, vegetables, tobacco, sugarcane, cotton, teak plantation	Sorghum, maize, yam, sweet potatoes, cotton, sesame, rice	Maize, cassava, cocoyam, cocoa, yam, oil palm, groundnut, coffee, plantain, banana, rubber	Wheat, maize, potato, enset, bean, barley, teff, sorghum	Rice, palm oil, rubber, vegetables	Maize, rice, beans, banana, plantain, cacao, coffee	Maize, sunflower, groundnut, soybean, cotton, sweet potato
Main livestock	Cattle, pigs, chickens	Cattle, sheep, goats, poultry	Goats, pigs, sheep, chickens	Cattle, sheep, goats, horses, donkeys, and chickens	Chickens	Cattle, pigs, chickens, horses	Goats, chickens, pigs, cattle and ducks
Diets and culture	Strong reliance on wilds foods and agroforestry systems	Well-developed agroforestry systems and strong market integration	Very strong cultural attachment to forest foods. Considerable rural to urban transfer of forest foods for sale in urban centers	Fast moving shift into greater market integration	Diet transition already taken place across the landscape. Heavy emphasis on processed foods and marketed products	Site with least reliance on wild foods due to loss of availability and strong market access	Currently strong traditional food culture but moving fast toward greater market integration and western foodstuffs

as "fruit consumption," "vegetable consumption," and "meat consumption," respectively.

Remote Sensing

Contemporary forest cover surrounding all households included in the household survey data was characterized with Landsat imagery (30-m resolution) using the best available imagery from years closest to the dates of household surveys within each country gathered from the United States Geological Survey's GLOVIS earth explorer tool (http://glovis.usgs.gov/). Images contained varying amounts of cloud cover and atmospheric haze, which presented challenges for identifying forest cover. Dry season imagery was selected to help minimize cloud cover and to help distinguish agricultural land from other vegetation types, except in Burkina Faso where tree canopies were most visible during the wet season. Selected years resulted in 2010 for Indonesia; 2013 for Burkina Faso, Ethiopia, and Zambia; 2014 for Bangladesh; and 2015 for Cameroon and Nicaragua. Images were classified into three basic classes: forest, non-forest, and no data (consisting of clouds, water bodies, and cloud shadows) using ENVI software (Exelis Visual Information Solutions, Boulder, Colorado). We used a combination of image thresholding (based on vegetation indices such as the Normalized Burn Ratio, Tasseled Cap Transformations, and Disturbance Index, Healey et al., 2005) and Maximum Likelihood-based classifications, as well as Support Vector Machine (SVM) classifiers, as appropriate, to best capture forest/non-forest at sites. Because image availability due to excessive cloud cover made image analysis most challenging at the Indonesia sites, we adapted published forest cover maps (from Hansen et al., 2013) that became available near the end of the project. Thus, rather than create our own forest cover maps in Indonesia, we used forest cover (as defined by Hansen et al., 2013) and further conducted a supervised classification within the forested areas to distinguish several types of plantations (rubber and oil palm) that we then reclassified as agriculture. Classification accuracy was assessed with a combination of field verification and high-resolution imagery (e.g., RapidEye, Google Earth) which aimed to use a minimum of 100–200 verification points at each site, as available. The proportion of forest within a 2-km radius of each household (termed "forest cover" here) was then determined using R package *raster*. A 2-km buffer approximated the average travel distance to forests at most sites, as determined in scoping exercises and key informant interviews conducted at all sites.

Indicators of Farm Production and Wealth

For each farming household, livestock numbers reported in the survey were converted into Tropical Livestock Units (TLU). Following the method of Jahnke (1982), sheep and goats were assumed to be equivalent to 0.1 TLU; donkeys, 0.5 TLU; and all types of cattle, 0.7 TLU.

In addition, each farm was qualified as integrated to the cash economy or not based on their reported sources of income. If the household reported wage labor, salary, a trade, or any form of business, they were classified as market integrated, while other households were classified as not market integrated.

Approximately 45% of households were classified as market integrated by this method.

Piece-Wise Structural Equation Modeling

To test and quantify the various pathways linking forest and dietary diversity, fruit, vegetable, and meat consumptions, structural equation models were used. Structural equation modeling has been used extensively in psychology, and increasingly in natural science. Structural equation modeling can be defined as "*the use of two or more structural [cause-effect] equations to model multivariate relationships*" (Grace, 2006). As such, structural equation models are generally represented as more or less complex networks of relationships. Structural equation modeling is related to regression, principal components analysis, and path analysis (McCune and Grace, 2002). However, a major difference is that structural equation modeling provides a means to evaluate the structure of the model (pattern of relationships among variables) as well as the model parameters using observed data (McCune and Grace, 2002). By model structure, we mean the correlations, direct, and indirect relationships among variables. Therefore, structural equation modeling can be used to test construct models (i.e., hypothesized models) and quantify relationships between model components (Grace, 2006). Although not used in this study, structural equation modeling also allows for the inclusion of unobserved (latent) variables as theoretical variables reflected by several indirect observed (manifest) variables (Grace et al., 2010).

A construct model was developed to test the three pathways linking forest cover to dietary diversity (**Figure 3**). Dietary diversity was hypothesized to be influenced by (1) forest cover (e.g., Ickowitz et al., 2014), representing the direct pathway described in the introduction; by (2) farm production (e.g., Jones, 2017), proxied by farm area, presence/absence of a home garden, and livestock ownership; and by (3) improved market access (e.g., Sibhatu et al., 2015), proxied by integration to the cash economy. Relationships between forest cover and farm area, between forest cover and presence/absence of a home garden, and between forest cover and livestock ownership were included to represent different ways in which forests can support farm production (e.g., Reed et al., 2017), i.e., different dimensions of the agroecological pathway described in the introduction. As crop residues and weeds often represent a major part of the diet of livestock in tropical countries (e.g., Baudron et al., 2014), a relationship between farm area and livestock ownership was included. Similarly, manure produced by livestock being often concentrated in home gardens (e.g., Baudron et al., 2017), a relationship between livestock ownership and presence/absence of a home garden was also included. In addition, a relationship between farm area and presence/absence of a home garden was included to test possible correlation between these two dimensions of crop production. Finally, a relationship between forest cover and integration to the cash economy was included to represent the possible sale of forest product (e.g., Angelsen et al., 2014), i.e., income pathway described in the introduction, and relationships between farm area, presence/absence of a home garden, livestock ownership, and integration to the cash economy, to represent the possible sale of farm products. A continuous variable was used for livestock ownership in the models representing Burkina Faso and Ethiopia, but a binary variable (presence/absence) for the models representing the five other landscapes, as a large fraction of households in these sites did not own any livestock. The same model was used for fruit consumption, vegetable consumption, and meat consumption, giving a total of 28 models (four per country). Global goodness of fit of the models was assessed by tests of directional separation. We ensured through these tests that all interactions were included, e.g., with Fisher's C of 0 and P-value of 1.

Structural equation modeling assumes that all variables are derived from a normal distribution, while dietary diversity can be assumed to follow a Poisson distribution, and fruit, vegetable, and meat consumption, presence/absence of a home garden, integration to the cash economy, and livestock ownership in five of the seven countries can be assumed to follow a binomial distribution. In response, piece-wise structural equation modeling is recommended, whereby paths are estimated in individual models and then pieced together to construct the causal model (www.jonlefcheck.net/2014/07/06/piecewise-structural-equation-modeling-in-ecological-research). This was performed using the R package *piecewiseSEM*.

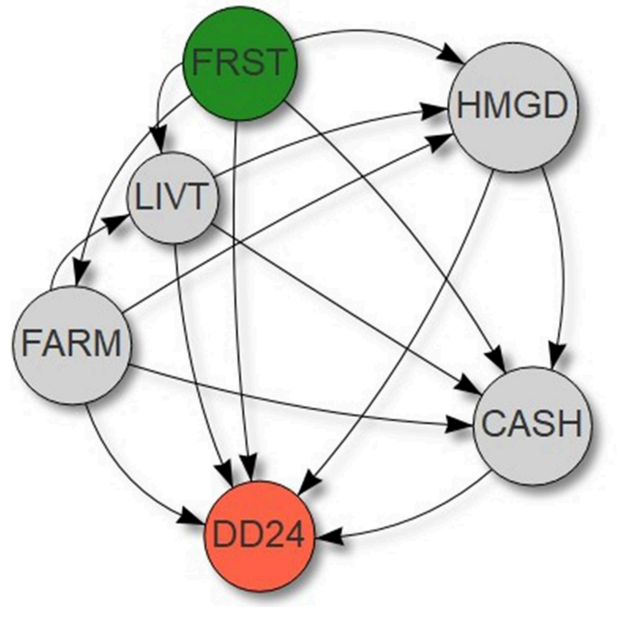

FIGURE 3 | Construct models used to test and quantify the pathways from **Figure 1** using piece-wise structural equation modeling. FRST: percentage of forest in a 2-km radius surrounding each household, DD24: household dietary diversity score recorded over the past 24 h, FARM: farm size (ha), HMGD: presence/absence of a home garden, LIVT: livestock ownership (TLU), CASH: integration to the cash economy (yes/no). See text for detailed description of the interactions included in this model. The same model was used with the other dietary measures: consumption of fruits in the past 24 h (yes/no), consumption of vegetables in the past 24 h, and consumption of meat and other animal product (excluding milk and milk products) in the past 24 h.

RESULTS AND DISCUSSION

Forest and non-forest were distinguished with quite high accuracy, with a few clear exceptions, as simplified land cover classifications with few classes (such as the ones we used) tend to be quite accurate. Overall accuracies exceeded 90–95% at most sites and at a few locations approached or surpassed only 80%. In Indonesia, the rubber and oil palm plantations were accurately discriminated from other forest cover to 93% accuracy. Locations with lower overall map accuracy were evident, as follows. In Bangladesh, teak plantations could not be discriminated from surrounding forests as they were spectrally similar and mostly quite small (<1 ha). In addition, this landscape also encompassed narrow, small linear non-forest features that were not well captured using 30-m imagery. Dry tropical forests, such as in Burkina Faso, achieved overall accuracy of only 86% largely driven by errors of omission whereby scattered trees as well as small forest patches with very sparse canopy cover were not detected by the 30-m Landsat imagery.

All piece-wise structural models fitted the observed data well, with a Fisher's C-value of 0 and a P-value of 1.

General Characteristics of the Seven Landscapes Studied

The seven landscapes were found to be characterized by varying levels of forest cover, with Bangladesh having the highest average proportion of forest surrounding (2 km radius) the studied farming households (88.1%) and Ethiopia having the lowest (12.0%) (**Table 2**). The variability in forest cover—measured by standard deviations in **Table 2**—was the greatest in Indonesia and the lowest in Burkina Faso.

The largest farms were found in Nicaragua (average of 10.97 ha) and the smallest ones were found in Ethiopia (average of 1.01 ha). About ¾ or more of the farms were cultivating a home garden in Cameroon, Ethiopia, Indonesia, and Nicaragua. The lowest proportion of farms cultivating a home garden was found in Zambia (11.3%). The largest livestock herds were found in Burkina Faso (average of 6.18 TLU) and the smallest ones were found in Indonesia (average of 0.75 TLU). The majority of households were considered integrated to the cash economy in Bangladesh and Indonesia. The lowest proportion of households considered integrated to the market was found in Ethiopia (15.1%).

The highest dietary diversity was recorded in Nicaragua (average score of 8.74) and lowest in Burkina Faso (average score of 6.14). The largest variability in dietary diversity was found in Zambia and the lowest was found in Indonesia. The majority of households consumed fruits in the 24 h that preceded the interview in Burkina Faso, Ethiopia, Nicaragua, and Zambia. Fruit consumption was the lowest in Bangladesh and Cameroon (around 40% in both landscapes). More than 90% of households consumed vegetables in the 24 h that preceded the interview in Bangladesh, Ethiopia, Indonesia, and Zambia. The lowest proportion of households consuming vegetables (but still high) was found in Nicaragua (76.6%). In all the landscapes except Ethiopia, about 80% of households or more consumed meat in the 24 h that preceded the interview (the percentage was 47.3% in Ethiopia).

Forest and the Direct Pathway to Diet Quality

The results of this piece-wise structural equation modeling identified a direct relationship or pathway between forest cover and dietary diversity in two landscapes: Bangladesh and Ethiopia (**Figure 4** and **Table 3**). This appeared to be linked, at least in part, to meat consumption in Bangladesh, and meat and fruit consumption in Ethiopia (**Table 3**). In addition, forest was found to support fruit consumption in Cameroon and meat consumption in Zambia (although no association between forest cover and dietary diversity was found for these landscapes; **Table 3**). No link between forest cover and vegetable consumption was found in any of the landscapes investigated (**Table 3**). This lack of relationship could be in part due to the high percentage of households that had consumed vegetables during the 24 h preceding the interview (**Table 2**).

Forest cover was positively associated with fruit consumption in Cameroon and Ethiopia. It was the only statistically significant predictor of fruit consumption in Cameroon and the predictor with the largest value in Ethiopia, thus underscoring the importance of forest access in these two landscapes (**Table 3**). Wild fruits are important food items in the diet of many rural communities around the world. These wild fruits are partly harvested not only from forests but also from trees retained on farmland (Campbell, 1987; Herzog et al., 1994; Kalenga Saka and Msonthi, 1994). Similar to wild fruits, wild vegetables are also often harvested from the farmland, as part of a "hidden harvest" (Scoones et al., 1992; Powell et al., 2015). Therefore, part of the positive associations found between farm area and fruit consumption in Cameroon (**Table 3**) and between farm area and vegetable consumption in Burkina Faso and Nicaragua (**Table 3**) may be explained by wild fruits and wild vegetables harvested from the farmland, not only from cultivated sources. Part of the positive association between farm area and meat consumption in Indonesia and Nicaragua may also be explained by wild animals that are often hunted from the farmland and not exclusively from forests (e.g., Smith, 2005). Similarly, the positive association found between livestock ownership and fruit consumption in Bangladesh and Nicaragua may be explained by collection of wild fruits during herding.

Forest cover was positively associated with meat consumption in three out of the seven landscapes considered (Bangladesh, Ethiopia, and Zambia; **Table 3**). Forest cover was the only statistically significant predictor of meat consumption in Bangladesh and Zambia, and the predictor with the largest value in Ethiopia, pointing to the importance of bushmeat and wild fish in many of the sites considered (**Table 3**). Nasi et al. (2011) estimated the total quantity of bushmeat extracted annually from tropical forests of Africa and South America to six million tons. Many of the African countries considered in their assessment do not produce enough non-bushmeat animal products to meet the requirements of their growing populations (Fa et al., 2003).

TABLE 2 | General characteristics of the 1,783 farms in the seven study landscapes analyzed in this study (% and mean ± standard deviation).

	Bangladesh	Burkina Faso	Cameroon	Ethiopia	Indonesia	Nicaragua	Zambia
Proportion of forest in the 2-km radius surrounding the farm (%)	88.1 ± 5.0	22.8 ± 4.0	78.3 ± 8.2	12.0 ± 13.1	27.4 ± 30.4	30.2 ± 11.1	19.4 ± 9.8
Farm size (ha)	2.35 ± 2.62	5.81 ± 3.93	4.97 ± 3.93	1.01 ± 0.63	6.84 ± 12.22	10.97 ± 23.03	2.08 ± 2.79
Presence of a home garden (%)	54.9	62.3	74.4	71.7	82	77.1	11.3
Livestock ownership (TLU)	2.42 ± 4.02	6.18 ± 8.31	0.10 ± 0.35	2.85 ± 2.24	0.75 ± 6.66	3.16 ± 9.84	3.08 ± 4.74
Integration to the cash economy (%)	67.6	29.9	39.7	15.1	83.7	17.8	44.5
24-h household dietary diversity score	8.07 ± 1.48	6.14 ± 1.53	6.75 ± 1.68	7.75 ± 1.99	8.04 ± 1.44	8.74 ± 1.94	8.31 ± 2.46
Fruit consumption in the last 24 h (%)	39.8	52	39.5	54.4	47.8	65.1	56.7
Vegetable consumption in the last 24 h (%)	99.7	81.1	84.9	99.1	98.4	76.6	94.2
Meat consumption in the last 24 h (%)	91.2	84.7	89.3	47.3	91.6	79.1	79.6

Indeed, in large parts of tropical Africa, livestock production is limited by diseases such as trypanosomiasis (Kristjanson et al., 1999). Bushmeat and wild fish thus represent a critical source of quality proteins and readily available micronutrients to millions in and around tropical forests. In Northeastern Madagascar, it was established that the loss of bushmeat in local diets would increase the incidence of anemia in children by 30% (Golden et al., 2011).

Securing access to forest food where it is of critical importance to local diets may be challenged when these forests are protected (Pimbert and Pretty, 2013), which is the case in Cameroon, Indonesia, and Zambia in particular. Forest protection, and enforcement of stricter conservation legislations, can limit access to critical resources that contribute to diets and there is often a trade-off between biodiversity conservation and dietary diversity (Hutton et al., 2005; Sylvester et al., 2016). The issue is particularly sensitive for bushmeat, as bushmeat harvesting for subsistence generally coexists with—often very lucrative—bushmeat trades and may affect endangered species (Maxwell et al., 2016). Commercial hunting for meat is seldom sustainable (Robinson and Bennett, 2004; Maxwell et al., 2016), but see Cowlishaw et al. (2005).

In addition to the positive associations between forest cover and diet quality reported above, negative associations were also uncovered. Forest cover was found to be negatively related to dietary diversity and fruit consumption in Zambia, and to fruit consumption in Indonesia (**Figure 4** and **Table 3**). In some circumstances, forest people may be vulnerable to seasonal gaps in some or all food groups, if wild food availability or access fluctuates seasonally (De Souza, 2006; Gabriele and Schettino, 2007). There may also be cultural differences and different dietary habits between populations living in the more forested and in the less forested parts of the same study site. Cultural differences may explain why we see positive relationships with forests in some sites and negative or neutral ones in others. For example, the communities in the Nicaragua site were non-Indigenous and lack the knowledge and tradition of wild food use (fruits, vegetables, bushmeat, etc.) seen in Indigenous populations of central America (Sylvester et al., 2016), helping to explain the lack of significant relationship between forest cover and any of the indicators of diet quality found in the Nicaragua site. The fact that forest cover covaried with ethnicity in some sites may also help to explain some of the weak and variable relationships seen (cultural variation in diet can be very large and could account for a lot of variation in our dietary variables). For example, in the Indonesian site, the communities with most forest cover were largely Dayak while the less forested communities were a mix of ethnic groups, including many immigrants from Java who have very different dietary traditions to the Dayak (Dove, 1999).

Forest, Integration to the Cash Economy, and the Income Pathway to Diet Quality

A positive association between integration to the cash economy and diet quality was found in three landscapes: Burkina Faso, Ethiopia, and Zambia. Integration to the cash economy was positively related to vegetable and meat consumptions in Burkina Faso, to fruit consumption in Ethiopia, and to dietary diversity and fruit consumption in Zambia (**Table 3**). These results concur with past findings that highlight the fact that improved market access tends to be associated with improved dietary diversity (Jones, 2017).

However, a positive association between forest cover and integration to the cash economy as well as a positive association between integration to the cash economy and diet quality—i.e., evidence of an income pathway—was not found in any of the landscapes studied. A positive association between forest cover and integration to the cash economy was found in Indonesia—where high-value forest products such as resin (e.g., "gaharu") and swiflet nests are harvested and traded (Leonald and Rowland, 2016)—but no association between integration to the cash economy and dietary quality was found in this landscape (**Tables 3**, **4**).

A negative association between forest cover and integration to the cash economy was found in three landscapes: Cameroon,

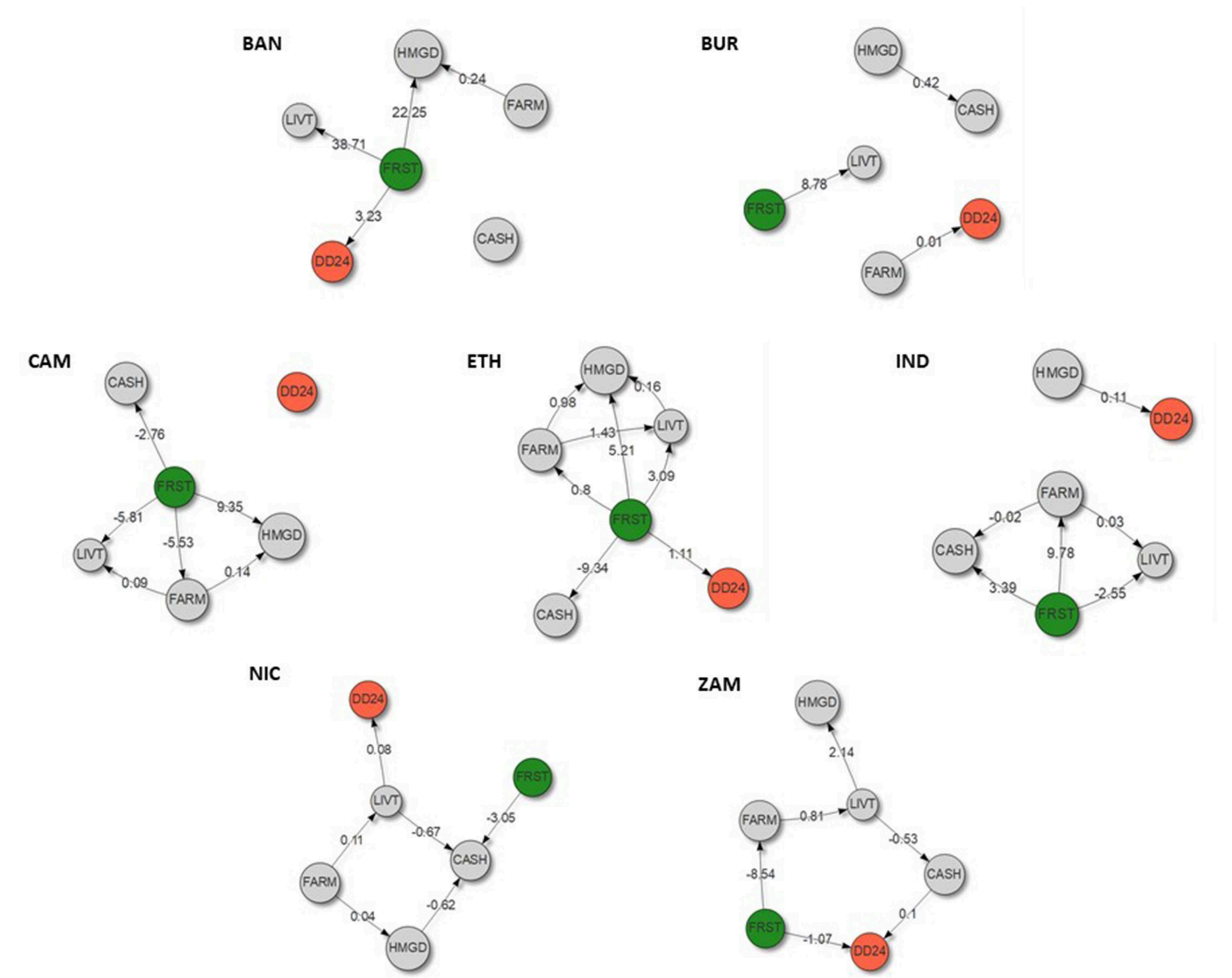

FIGURE 4 | Piece-wise structural equation models linking the percentage of forest in a 2-km radius surrounding each household (FRST) to its household dietary diversity score recorded over the past 24 h (DD24) for Bangladesh (BAN), Burkina Faso (BUR), Cameroon (CAM), Ethiopia (ETH), Indonesia (IND), Nicaragua (NIC), and Zambia (ZAM). Only relationships that are statistically significant ($P < 0.1$) are represented with their coefficient in the networks. FARM: farm size (ha), HMGD: presence/absence of a home garden, LIVT: livestock ownership (TLU or presence/absence), CASH: integration or not to the cash economy.

Ethiopia, and Nicaragua (**Table 4**). Forested areas of the tropics tend to be remote rural areas, which are often characterized by poverty (Bird et al., 2011). Income-earning opportunities tend to be limited, and markets distant (Angelsen and Wunder, 2003).

Forest, Crop and Livestock Production, and the Agroecological Pathway

Our results suggest that agricultural production supports diet quality in five out of the seven countries studied. Farm area was positively associated with dietary diversity and fruit and vegetable consumptions in Burkina Faso, with vegetable and meat consumptions in Nicaragua, and with meat consumption in Indonesia. Home gardens were positively associated with dietary diversity and meat consumption in Indonesia, with fruit consumption in Burkina Faso, and with meat consumption in Ethiopia. Finally, livestock ownership was positively associated with improved dietary diversity and fruit and meat consumptions in Nicaragua and with improved fruit consumption in Bangladesh (**Figure 4** and **Table 3**).

We found evidence of an agroecological pathway—positive associations between forest and agricultural production (farm, home garden, or livestock), combined with a positive association between agricultural production and diet quality—in three landscapes: Bangladesh, Ethiopia, and Indonesia. This was characterized by positive associations between forest cover and livestock ownership, and between livestock ownership and fruit consumption in Bangladesh; positive associations between forest cover and presence of a home garden, and between presence of a home garden and meat consumption in Ethiopia; and by positive associations between forest cover and farm area, and between farm area and meat consumption in Indonesia.

Though generally not combined with positive relationships with diet quality, evidence of positive association between forest cover and crop and livestock production was found in five

TABLE 3 | Estimates and their confidence intervals and associated *P*-values for the predictors of household dietary diversity scores (DD24), fruit consumption (FT24), vegetable consumption (VG24), and meat (and other animal product excluding dairy) consumption (MT24) for Bangladesh (BAN), Burkina Faso (BUR), Cameroon (CAM), Ethiopia (ETH), Indonesia (IND), Nicaragua (NIC), and Zambia (ZAM).

Pred	Bangladesh			Burkina Faso			Cameroon			Ethiopia			Indonesia			Nicaragua			Zambia		
	Est	Std	*P*-val	Est	Std	*P*-val	Est	Std	*P*-val	Est	Std	*P*-val	Est	Std	*P*-val	Est	Std	*P*-val	Est	Std	*P*-val
DD24																					
FRST	**3.23**	**1.65**	**0.05**	0.07	0.17	0.68	0.43	0.33	0.19	**1.11**	**0.19**	**0.00**	0.00	0.08	0.97	−0.32	0.20	0.10	**−1.07**	**0.24**	**0.00**
CASH	0.00	0.05	0.97	0.01	0.05	0.89	0.08	0.05	0.12	0.01	0.07	0.84	0.07	0.06	0.28	−0.06	0.06	0.34	**0.10**	**0.04**	**0.02**
FARM	0.00	0.01	0.76	**0.01**	**0.01**	**0.07**	0.00	0.01	0.58	0.05	0.04	0.20	0.00	0.00	0.30	0.00	0.00	0.65	0.01	0.01	0.25
HMGD	0.01	0.05	0.82	0.07	0.05	0.19	−0.01	0.06	0.89	0.06	0.06	0.31	**0.11**	**0.06**	**0.08**	0.04	0.05	0.50	0.10	0.06	0.13
LIVT	0.01	0.05	0.89	0.00	0.00	0.48	0.01	0.06	0.94	0.01	0.01	0.37	0.00	0.07	0.98	**0.08**	**0.05**	**0.09**	0.04	0.05	0.43
FT24																					
FRST	−4.35	9.74	0.66	0.86	0.91	0.35	**5.34**	**1.68**	**<0.01**	**11.22**	**1.81**	**0.00**	**−1.54**	**0.49**	**0.00**	1.17	1.29	0.37	**−7.00**	**1.61**	**0.00**
CASH	0.23	0.33	0.48	0.11	0.28	0.68	0.27	0.27	0.33	**0.68**	**0.41**	**0.10**	0.04	0.36	0.91	**−0.64**	**0.34**	**0.06**	**0.80**	**0.28**	**0.00**
FARM	−0.02	0.05	0.69	**0.07**	**0.04**	**0.06**	−0.05	0.04	0.22	0.46	0.29	0.12	0.00	0.01	0.82	0.00	0.01	0.81	0.12	0.09	0.18
HMGD	0.31	0.31	0.31	**1.11**	**0.26**	**<0.01**	0.13	0.31	0.68	−0.07	0.38	0.86	0.48	0.34	0.16	0.17	0.32	0.60	0.63	0.44	0.15
LIVT	**0.71**	**0.32**	**0.03**	0.01	0.02	0.73	−0.31	0.37	0.41	0.08	0.09	0.37	0.13	0.39	0.75	**0.50**	**0.30**	**0.10**	−0.09	0.30	0.76
VG24																					
FRST				0.62	1.15	0.59	−2.76	2.30	0.23	40.20	86.20	0.64	0.52	2.32	0.82	−2.96	1.33	0.03	−2.57	2.74	0.35
CASH				**0.69**	**0.39**	**0.08**	−0.02	0.37	0.96	−2.82	3.61	0.44	0.50	1.22	0.68	−0.39	0.38	0.30	−0.49	0.53	0.35
FARM				**0.19**	**0.06**	**<0.01**	0.02	0.05	0.63	−0.48	1.40	0.73	0.28	0.26	0.29	**0.03**	**0.02**	**0.09**	0.11	0.21	0.60
HMGD				−0.41	0.34	0.24	−0.10	0.43	0.82	18.37	3181	1.00	0.30	1.19	0.80	**−0.09**	**0.37**	**0.81**	0.82	1.07	0.44
LIVT				0.01	0.02	0.62	**−1.08**	**0.42**	**0.01**	0.55	0.75	0.47	−1.05	1.21	0.39	0.34	0.33	0.30	−0.83	0.62	0.18
MT24																					
FRST	**31.92**	**16.23**	**0.05**	0.42	1.23	0.73	−1.64	2.54	0.52	**8.62**	**1.61**	**<0.01**	0.96	1.12	0.39	−2.20	1.41	0.12	**4.05**	**1.94**	**0.04**
CASH	−0.24	0.61	0.70	**0.70**	**0.42**	**0.09**	0.65	0.44	0.14	**−1.06**	**0.51**	**0.04**	0.53	0.58	0.36	**−0.71**	**0.38**	**0.07**	0.46	0.32	0.15
FARM	0.02	0.10	0.87	0.08	0.05	0.16	0.08	0.07	0.24	0.42	0.30	0.17	**0.18**	**0.10**	**0.06**	**0.04**	**0.02**	**0.07**	0.15	0.13	0.24
HMGD	0.83	0.54	0.13	**−2.03**	**0.55**	**<0.01**	−0.12	0.48	0.80	**0.84**	**0.40**	**0.04**	**1.29**	**0.49**	**0.01**	−0.58	0.41	0.16	0.46	0.57	0.42
LIVT	0.17	0.50	0.74	−0.01	0.02	0.48	0.66	0.65	0.31	0.10	0.09	0.24	−0.73	0.63	0.25	**0.55**	**0.34**	**0.10**	0.55	0.34	0.11

Predictors with an associated P-value lower than 0.1 are in bold.

landscapes. Forest cover was positively related to farm area in Ethiopia and Indonesia; to the presence of a home garden in Bangladesh, Cameroon, and Ethiopia; and to livestock ownership in Bangladesh, Burkina Faso, and Ethiopia (**Table 4**). This positive relationship could be explained by ecosystem services provided by forests. In particular, soil fertility maintenance, micro-climate regulation, and pollination may be critical to crop species found in home gardens (Islam et al., 2008; Garibaldi et al., 2011; Baudron et al., 2017). Larger farm areas and larger livestock herds in the more forested sites may also be explained by lower population densities, resulting in greater availability of land for local farmers (Dzingirai et al., 2013).

Conversely, forest cover was negatively associated with farming in three landscapes, as reflected in smaller farm areas in Cameroon and Zambia, and reduced livestock ownership in Cameroon and Indonesia (**Table 4**). This negative association could be the reflection of policies that encourage conventional forms of intensification and not tree-based crop and livestock production systems (agroforestry and silvopastralism) and other production systems based on agroecology (Garibaldi et al., 2019). For instance, both the Cameroon and the Indonesia landscapes are characterized by a rapid expansion of large-scale plantations (Asaha and Deakin, 2016; Leonald and Rowland, 2016). This negative association could also be the result of lost opportunities to convert forests—particularly if they are protected—to cropland and pastures (Balmford and Whitten, 2003). It could as well be the result of crop destruction and livestock depredation by wildlife in the most forested parts of these landscapes (Choudhury, 2004; Michalski et al., 2006; Yirga and Bauer, 2010; Baudron et al., 2011). Forests may also act as reservoirs of crop pests and the wildlife they host may transmit diseases to livestock (Bengis et al., 2002; Blitzer et al., 2012). Much more emphasis is placed on ecosystem services than ecosystem disservices in the scientific literature. However, considering both is crucial in the design of multifunctional landscapes that deliver net benefits to local residents, in terms of diet quality but also other aspects of human well-being.

Limitations of the Study

Although illuminating regarding the pathways linking forest cover to dietary diversity, this research suffered from a number of limitations, which should be considered by future studies.

While our cross-site comparison allowed us to evaluate if patterns occurred across countries and forest types, higher spatial and temporal resolution of both forest cover and dietary diversity datasets may allow us to better distinguish the pathways from forests to diets. With the use of 30-m-resolution satellite images (from Landsat imagery), some small forest patches were likely undetected, particularly in the most sparsely forested landscapes. Forest detection could be improved with the use of images of higher resolution (Sentinel-2 images have a 10-m resolution, RapidEye images have a 5-m resolution, and Quickbird images have a 2.5-m resolution). Diet quality was only assessed once in each household, missing the temporal dynamic of availability and consumption of the different food groups. The sources of the different food groups (forest, farm, and market) were also not recorded, reducing the power of our analysis. The proxies

TABLE 4 | Estimates and their confidence intervals and associated *P* values for the predictors of integration to the cash economy (CASH), farm size (FARM), home garden presence/absence (HMGD), and livestock ownership (LIVT) for Bangladesh, Burkina Faso, Cameroon, Ethiopia, Indonesia, Nicaragua, and Zambia, derived from the piece-wise structural equation models using household dietary diversity scores.

Pred	**Bangladesh**			**Burkina Faso**			**Cameroon**			**Ethiopia**			**Indonesia**			**Nicaragua**			**Zambia**		
	Est	**Std**	***P*-val**	**Est**	**Std**	***P*-val**	**Est**	**Std**	***P*-val**	**Est**	**Std**	***P*-val**	**Est**	**Std**	***P*-val**	**Est**	**Std**	***P*-val**	**Est**	**Std**	***P*-val**
CASH																					
FRST	−4.89	5.73	0.39	0.06	0.78	0.94	**−2.76**	**1.40**	**0.05**	**−9.34**	**2.95**	**<0.01**	**3.39**	**1.11**	**<0.01**	**−3.05**	**1.80**	**0.09**	1.21	1.40	0.39
FARM	0.00	0.03	0.91	0.03	0.02	0.13	−0.01	0.03	0.83	0.33	0.34	0.33	**−0.02**	**0.01**	**0.10**	−0.03	0.02	0.16	−0.10	0.08	0.22
HMGD	−0.25	0.17	0.15	**0.42**	**0.24**	**0.09**	−0.11	0.24	0.63	0.51	0.45	0.26	−0.08	0.47	0.86	**−0.62**	**0.37**	**0.09**	0.26	0.40	0.51
LIVT	−0.18	0.17	0.31	0.00	0.01	0.74	−0.33	0.28	0.24	−0.06	0.11	0.57	0.00	0.49	1.00	**−0.67**	**0.36**	**0.06**	**−0.53**	**0.28**	**0.05**
FARM																					
FRST	−7.82	12.30	0.53	−0.74	1.68	0.66	**−5.53**	**3.06**	**0.07**	**0.80**	**0.32**	**0.01**	**9.78**	**2.53**	**<0.01**	−11.12	13.06	0.40	**−8.54**	**1.65**	**<0.01**
HMGD																					
FRST	**22.25**	**10.03**	**0.03**	1.21	0.92	0.19	**9.35**	**2.18**	**0.00**	**5.21**	**1.66**	**<0.01**	0.63	0.63	0.31	2.32	1.52	0.13	−0.29	2.23	0.90
FARM	**0.24**	**0.07**	**<0.01**	−0.03	0.03	0.31	**0.14**	**0.06**	**0.01**	**0.98**	**0.38**	**0.01**	0.02	0.02	0.48	**0.04**	**0.02**	**0.05**	**2.14**	**0.75**	**<0.01**
LIVT	0.14	0.30	0.64	0.02	0.02	0.21	0.68	0.43	0.11	**0.16**	**0.09**	**0.09**	0.28	0.53	0.60	0.01	0.34	0.97	−0.01	0.07	0.88
LIVT																					
FRST	**38.71**	**10.13**	**<0.01**	**8.78**	**3.53**	**0.01**	**−5.81**	**2.23**	**0.01**	**3.09**	**1.04**	**<0.01**	**−2.55**	**1.00**	**0.01**	−0.75	1.32	0.57	−0.15	1.60	0.93
FARM	0.05	0.05	0.36	0.10	0.13	0.41	**0.09**	**0.04**	**0.01**	**1.43**	**0.22**	**<0.01**	**0.03**	**0.01**	**0.02**	**0.11**	**0.03**	**<0.01**	**0.81**	**0.18**	**<0.01**

Predictors with an associated P-value lower than 0.1 are in bold.

of crop and livestock production (farm area, presence or not of a home garden, livestock ownership) were coarse: a more refined picture could be obtained by measuring actual production and diversity of key food groups by these different farm components. Similarly, the use of actual income data, rather than the use of a binary variable for integration to the cash economy, would be more powerful to test the income pathway.

In addition to these issues of data resolution, our analysis could have been improved with the inclusion of forest tenure and ethnicity data. For instance, we did not account for forest tenure in this analysis, and as such, forest cover does not necessarily equate with accessibility of forests. As noted above, the inability to account for ethnicity complicated interpretation of the results in some sites where this has a strong impact on dietary habits. For example, forest communities in the Indonesia landscape do not normally consume pulses, while those living in less forested areas have adopted tofu consumption, a dietary practice introduced from elsewhere in Indonesia.

CONCLUSIONS

While a growing number of studies have found fairly consistent relationships between forest cover and diet quality (Ickowitz et al., 2014; Galway et al., 2018; Rasolofoson et al., 2018), this study highlights the diversity of pathways that may be driving these relationships. The relative importance of each pathway varied between each of the study sites. We found evidence of a direct pathway to at least one of our four diet metrics in four landscapes (Bangladesh, Cameroon, Ethiopia, and Zambia), of an income pathway in none of the landscapes, and of an agroecological pathway in three landscapes (Bangladesh, Ethiopia, and Indonesia).

Although it appears to be the most important link between forests and diets, the sustainability of the direct pathway is threatened both by a return to more stringent conservation policies (Hutton et al., 2005) and by unsustainable harvesting of forest products, often fueled by demand from distant markets. This study also found evidence of forest supporting crop and livestock production in five landscapes, although this only led to improved diet quality (i.e., agroecological pathway) in three landscapes. These forest-production linkages have implications for the question of integration or segregation of food production and nature conservation, as encapsulated by the land sharing vs. land sparing debate. Although several studies have demonstrated that land sparing (i.e., segregation of food production and nature conservation) appears to offer the best outcome for tropical biodiversity (Phalan, 2018), this segregation is likely to represent a threat to local food production, as it would cut off smallholder farms from critical ecosystem services (critical as smallholders in the tropics tend to depend on ecosystem services more than external inputs).

These results highlight the intricacies of when and where different pathways link forests to better diet quality. In the context of rapid dietary and landscape changes, forests may be more important in some places than others, but we do not yet have enough evidence to determine where forests are most needed. Our results also suggest that the positive contributions of forests to rural livelihoods cannot be generalized and should not be idealized.

DATA AVAILABILITY STATEMENT

The datasets generated and analyzed for this study can be found in the Harvard Dataverse repository (doi: 10.7910/DVN/8SRAZT).

AUTHOR CONTRIBUTIONS

FB conceived the analysis, analyzed data, and wrote the first draft of the manuscript. ST and SG analyzed the remote sensing data used in the analysis. JG helped with the analysis. TS coordinated the project that provided the field data. FB, ST, BP, SG, and TS contributed to data interpretation and writing of the final manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2019.00097/full#supplementary-material

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Life on the Rainforest Edge: Food Security in the Agricultural-Forest Frontier of Cross River State, Nigeria

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A growing body of evidence demonstrates the importance of forests and wild animal-based foods for diets within tropical environments. However, deforestation and associated land-use changes can have competing effects on nutrition and food security as communities reorient from wild food use and subsistence-based agriculture to import/export markets. This research examines dietary differences and associated changes in food security during intermediate stages of deforestation and market integration in the agriculture-forest frontier of Cross River State, Nigeria. We used participant responses to mixed-methods interviews ($n = 528$) in six communities to measure individual dietary diversity, household food access, and short-term nutritional status, with specific attention to animal-based foods and the cultural and economic values attached to them, in two interior forest ($n = 177$) and four forest-edge ($n = 351$) communities. Multivariate analysis of dietary compositions revealed differences in food categories and types of meat consumed between forest environments. People in forest-edge communities reported consuming less bushmeat and dark green leafy vegetables, and more pulses, domestic meat, fish, eggs, dairy, other vegetables, sweets, condiments, and non-red palm oil compared to interior forest communities. Bushmeat was highly preferred and had more economic value than other animal-based foods, regardless of location. Forest-edge communities had fewer households involved in bushmeat related activities, and fewer hunters per household. However, traders in forest-edge communities sold a larger proportion of meat to people outside of the community than did traders in interior forest communities. Measures of nutrition and food security, but not wealth, improved in relation to dietary patterns in forest-edge communities compared to interior forest communities. Our results may reflect a "best of both worlds" scenario during the intermediate stages of deforestation and agricultural expansion near forested areas, where people have access to forest resources, increased ability to capitalize on forest goods, and access to market goods as they become integrated into market economies. Understanding the dietary consequences of environmental change is important, as food-related experiences may shape the trajectories of livelihood practices and landscape changes in tropical forests of biodiversity significance.

Keywords: agriculture, deforestation, bushmeat, conservation, diet, food security, West Africa

INTRODUCTION

Food provisioning is an important ecosystem service of forests, contributing to improved dietary diversity, nutrition, and food security in rural areas (Powell et al., 2011; Johnson et al., 2013; Vinceti et al., 2013; Ickowitz et al., 2014; Vira et al., 2015; Galway et al., 2018; Rasolofoson et al., 2018). Consumption of wild animals (colloquially known as "bushmeat") is considered particularly valuable, as it improves access to bioavailable nutrients that can be difficult to obtain from plants alone (Fa et al., 2003, 2015; Murphy and Allen, 2003; Sirén and Machoa, 2008; Cawthorn and Hoffman, 2015). Mounting evidence for nutritional benefits of forests suggests that forest conservation itself may offer benefits on par with nutrition-sensitive interventions (Ruel et al., 2013; Rasolofoson et al., 2018). For example, forest proximity causes children to have 25% greater dietary diversity (Rasolofoson et al., 2018), and removing access to wildlife is projected to induce a 29% increase in the prevalence of childhood anemia and a tripling of cases among those in the poorest households in Madagascar (Golden et al., 2011).

There are multiple interrelated pathways by which food systems may respond to tropical land use changes, including interactions between agricultural expansion, market integration, and conservation policies. Agricultural expansion is the leading cause of tropical deforestation, altering local ecologies and contributing to biodiversity losses (Geist and Lambin, 2002; van Vliet et al., 2012). Conservation policies aimed, in part, at reducing agricultural expansion and deforestation often restrict use of remnant forests thereby also limiting access to wild foods and new agricultural land (Ribot et al., 2006; Sandbrook et al., 2010). Limited access to wild foods can have negative consequences for nutrition and food security in local communities, especially in low income areas (Myers et al., 2013; van Noordwijk et al., 2014). Limited access to land from agricultural expansion and/or conservation policies further alters food systems by encouraging intensive agriculture when space is limited and forest clearing is prohibited (van Vliet et al., 2012). Land use intensification and monocropping can in turn create new agricultural challenges; for example from pests, weeds, and reduced soil quality (Geist and Lambin, 2002; van Vliet et al., 2012). Overall, declining diversity in agricultural production is associated with lower household and individual dietary diversity (Jones, 2017).

Land use change can have additional effects on food systems when deforestation results in reorientation to import/export markets. Markets can negatively affect dietary diversity (Reyes-García et al., 2019), as communities shift away from locally collected and produced foods toward processed foods high in fat, sugar, and salt (Kuhnlein and Receveur, 1996; Popkin, 2004; Kuhnlein et al., 2009; Piperata et al., 2011; Van Vliet et al., 2015; Reyes-García et al., 2019). Market access is also associated with decreased use of shifting cultivation strategies and increased reliance on intensive and commercial agriculture (van Vliet et al., 2012). However, market access and integration can also help redistribute food, increase dietary diversity, and shape food preferences (Bowles, 1998; Sibhatu et al., 2015; Clary et al., 2017; Koppmair et al., 2017; Ickowitz et al., 2019). Thus, with market integration, commercialization of agricultural, and forest products can provide new food and income opportunities that may improve nutritional outcomes and purchasing power. However, this may lead to trade-offs when income does not translate into improved nutrition (Herforth and Ahmed, 2015).

Bushmeat provides a clear example of the trade-off between nutrition and income. Bushmeat is a nutritionally significant component of local diets, providing an important source of protein (Fa et al., 2003), fat (Sirén and Machoa, 2008), and iron (Golden et al., 2011). There are demonstrated links between bushmeat consumption and improved nutritional status in rural hunting communities (Golden et al., 2011; Fa et al., 2015; Sarti et al., 2015). However, large profit margins incentivize trade in local, national, and international markets, thereby diverting nutritionally important resources outside of communities (Fa et al., 2002, 2006). Widespread exploitation and commercialization of bushmeat across West and Central Africa may therefore threaten food security as well as biodiversity (Fa et al., 2002, 2015; Ripple et al., 2016; Wilkie et al., 2016). For example, projected declines in availability of bushmeat protein over the next 50 years is expected to leave very few countries in the Congo Basin able to meet daily protein requirements (Fa et al., 2003).

Dietary transitions, and their associated health consequences, are primarily understood from studies of hunter-gatherer populations that provide a baseline for measuring the effects of market integration and increased reliance on agriculture (e.g., Reyes-García et al., 2019), and large panel studies that offer insights into the global trends and causal pathways by which forests impact nutrition (e.g., Rasolofoson et al., 2018). However, these approaches can systematically miss important variation at intermediate stages of deforestation and/ or market integration, when communities are lumped together using low stringency criteria, or when sites are ignored because they do not align within well-defined categories (e.g., forested vs. not forested; hunter-gatherer vs. farmer). Furthermore, forest communities with limited deforestation and market integration are often remote, making data collection resource intensive (Reyes-García et al., 2019). As a result, we know very little about the diets of people who live in marginal environments or who exist within the unexamined spaces of these gradients (i.e., semi-forested; hunter-farmers). Understanding these contexts is important, in that they reflect intermediate stages of dietary transitions, where people have access to forest, agricultural, and market foods, as well as the ability to capitalize on these resources via increased market vicinity. Furthermore, the food experiences in these intermediate stages contribute to the trajectory of dietary transitions within landscapes of change, and their consequent effects on health of humans and the environment.

In this study, we examine the effects of tropical deforestation and land use change on diets and food security in an agricultural-forest frontier in West Africa. Our research is focused in a highly relevant system within Cross River State in the South-South geopolitical zone of Nigeria, where expansion of subsistence and commercial agriculture and regional conservation efforts have altered the landscape that provides food and livelihoods. Cross River State contains the largest tract of contiguous forest left in Nigeria and is one of Africa's most important biodiversity reserves (Oates, 1999; Myers et al., 2000; Kamden-Toham et al., 2006). Diverse faunal assemblages within Cross River provide bushmeat to rural communities and urban markets throughout Nigeria and into Cameroon (Fa et al., 2014; Friant et al., 2015; Lameed et al., 2015; Abere et al., 2016). Communities in Cross River vary in their proximity and access to forests and their degree of market integration, in part, because of the long and complicated history of the formation of Cross River National Park and the more recent expansion of the agricultural frontier (Oates, 1999; Ite and Adams, 2000; Schoneveld, 2014). Here we examine how these landscape changes (i.e., the combined impacts of deforestation, agricultural expansion, and forest protection) affect diets and food security within this agriculture-forest frontier. We use a concept of food security that extends beyond caloric sufficiency and dietary staples, toward a more balanced view that reflects access to sufficient quantities of nutritious food for an active and healthy life (USDA, 1996; Ickowitz et al., 2014; Pingali, 2015). Using this framework, we examine how land use changes and market integration at the agriculture-forest frontier affect: (1) diets, (2) bushmeat consumption and trade, (3) food values, and (4) nutrition and food security outcomes. Finally, we consider the implications of our results for human and ecosystem health within landscapes of change.

The forests in Cross River are part of the Cross-Sanaga-Bioko coastal forest, which contains primary and secondary growth forest and unusually high species richness and diversity (Myers et al., 2000; Oates et al., 2004; WWF, 2016). The southern forests of Nigeria cover <2% of Nigeria's landmass, with deforestation in this region dating back to colonial rule in the 1800s and continuing beyond independence (1960s) at an estimated annual rate of 3.7% (FAO, 2010; Enuoh and Ogogo, 2018). Cross River National Park was established in 1991 with an initial plan to extend park boundaries to protect most nearby intact forest and bring rural development projects and guaranteed support for communities that would lose access to agricultural land and non-timber forest products (Ite, 1998; Oates, 1999; Ite and Adams, 2000). However, these plans were never fully implemented due to disputes over funds that were prioritized over conservation objectives, and the withdrawal of support from international donors in response to the execution of environmental activists in Nigeria at the time (Oates, 1999; Ite and Adams, 2000). As a result, Cross River National Park was never fully established, and limited funds have resulted in a support zone consisting of uncompensated and resentful communities on the periphery and interior of protected areas. Meanwhile, population growth and limited access to land contributed to early refusals to grant land for re-settlement of interior forest communities (Ewah, 2013). These communities now exist as designated enclaves within both divisions of the park, where they are allocated forest for farming and hunting. Communities outside of the designated park boundaries are classified as support zone communities. The South is one of Nigeria's largest producers of export crops, including cocoa, rubber, and palm oil, with rapid expansion of large new privatized areas of land allocated to "high-capacity" agricultural investors that are encroaching on both protected and indigenous lands (Schoneveld, 2014).

Local inhabitants of this region do not easily fall within the "hunter-gatherer"—"farmer" dichotomy, and are perhaps best characterized as hunter-agriculturalist societies that depend mainly on agriculture for staple food items and use a combination of wild and cultivated vegetables and animals (Rupp, 2003; Ewah, 2013; Friant et al., 2015; Lameed et al., 2015; Abere et al., 2016). Rural communities in this area depend on the forest for cooking fuel, farmland, and for non-timber forest products, including bushmeat. Forest protection prohibits, to some extent, agricultural expansion into protected areas. However, lax enforcement of laws has resulted in exploitation of forests for timber, construction of roads, agricultural land, and non-timber forest products. The ecological integrity of the forest is now severely threatened by a myriad of human activities, exacerbated by population explosion and high levels of poverty and unemployment (Mahmoud et al., 2017; Enuoh and Ogogo, 2018).

METHODS

Study Site and Participants

Our study included six out of 105 (est.) communities near the Oban ($\sim$3,000 km^2) and Okwangwo ($\sim$640 km^2) divisions of Cross River National Park (CRNP) in Nigeria (**Figure 1**). To increase the generalizability of our results, we included communities that represent the three predominate cultural groups living near the park: Boki, Ejagham, and Ayo (Chrisomalis, 2006). Communities were selected to ensure both divisions of the park were represented and to maximize sampling across cultural groups and local government areas. Within these criteria, we selected communities where we had previous research experience or the ability to establish contact with people who could facilitate our entry into potentially resentful communities.

We selected two communities designated as enclaves within the interior of CRNP ("interior forest") and four communities designated as the support zone on the periphery of CRNP ("forest-edge"). Interior and forest-edge communities differed in: (1) proximity and access to forests, (2) road access, and (3) access to markets for selling and purchasing food. Due to their location

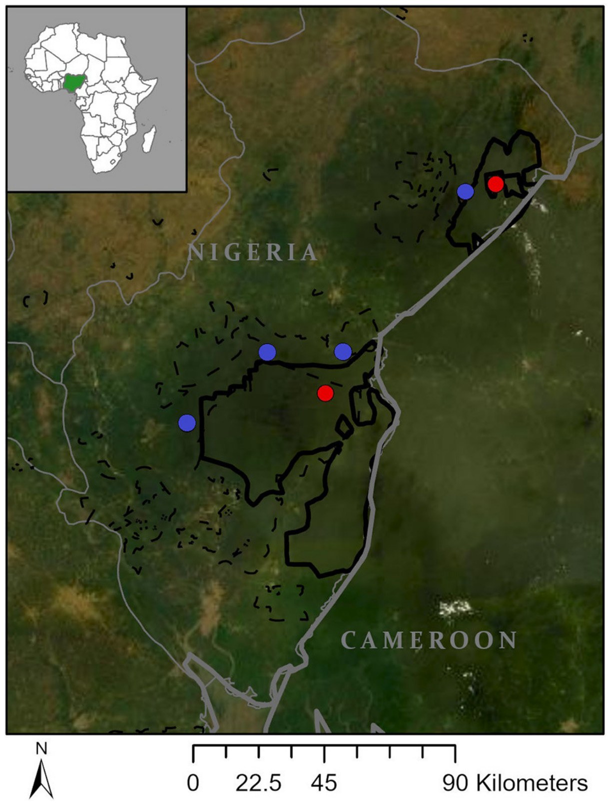

FIGURE 1 | Study communities. Map showing location of deep (red) and marginal (blue) forest communities relative to Cross River National Park (green) in Nigeria.

within CRNP, interior forest communities are surrounded by forest, lack motorable roads, and are typically accessed by foot or motorbike via forest trails and partially graded dirt roads that cut through protected areas representing a mosaic of forest and agricultural land. Depending on the mode of travel, it took inhabitants of interior forest communities between half and a full day to reach the nearest markets, and their loads were limited to what they could fit on the back of a motorbike on poor roads or what could be carried on their heads (∼20–50 kg). Forest-edge communities are typically accessed by motorbike, vehicles, or motorboat via grated dirt roads, paved roads, or rivers that connect communities to major roads. It took people between 30 min and 2 h to reach major roads and markets, and goods were transported via motorbikes (with heavier loads), vehicles, or boats. Due to their location on the periphery of CRNP, forest-edge communities had access to remnant community forest areas between communities and the park, but most of the surrounding landscapes were heavily deforested from expansion of farmlands, timber business, and/or private commercial agriculture industries (e.g., palm oil plantations).

Data Collection

We restricted data collection to the wet/lean season (June–August 2017) to limit effects of seasonal variation in food availability and road access across sites. We combined individual questionnaires and anthropometric measurements with key informant interviews and participant observations to obtain data on individual diets and nutritional status, household food security, and cultural and economic values attached to food items. All interviews were conducted in Nigerian Pidgin English, which is the *lingua franca* of the region, to limit differences in interpretation of questionnaires across cultures. However, translations to local languages were made *ad hoc* when specific words or phrases were not well-understood. Questionnaire instruments were translated into Nigerian Pidgin English, back translated, piloted, and adapted in a neighboring village where no study activities took place. During this pilot phase, we developed initial food lists from observations in households, farms, and local markets and shops. We then worked with key informants who added foods, information on edible parts, and food sources. They also provided locally relevant phrases and examples for evaluating food insecurity (e.g., lists of undesirable foods and local phrasing for "lack of resources") (Coates et al., 2007; Kennedy et al., 2011). Within each study community, we piloted the questionnaire, asking key informants to answer and then explain the meaning of each question to help ensure it was understood locally. However, because we did not undergo the full adaptation process in each community, we caution that biases could have been introduced where we missed more optimal phrases and locally relevant examples.

Within communities, we randomly selected households from a drawn village map. Households were defined as people who regularly shared food from the same pot. From the questionnaires, we obtained demographic, livelihood, and socioeconomic information, including household participation in the bushmeat trade and household food insecurity, alongside information on individual dietary diversity and meat consumption. Questionnaires were implemented with the head of household responsible for food production ($n = 323$), representing an average of 48% of households per village (range: 14–84%). This person was typically female ($n = 318$), unless there was no female present in the household ($n = 5$). We then randomly re-sampled ∼50% of those households to obtain dietary information from men within the same household ($n = 155$) for a total of 478 individuals (interior forest $n = 158$, forest-edge $n = 320$). To evaluate undernutrition, we recorded the mid-upper arm circumference of all respondents (Godoy et al., 2006; USAID et al., 2018). Questionnaire responses and anthropometric measures were recorded by one of four Nigerian research assistants who were accompanied by local translators who verbally translated into local dialects as needed. Answers to closed-ended questions were recorded using ODK® software on a tablet, and open-ended questions were transcribed in real time. All households were offered soap as an incentive gift for participation. We then purposively selected men and women involved in hunting, cooking, and trading in meat as key informants to obtain information on meat preferences and economic values attached to different types of meat (interior forest $n = 19$, forest-edge $n = 31$).

Household and Sociodemographic Information

We collected information to identify demographic and socioeconomic factors that may influence diets and food security status, including: age (*years*); marital status (*yes/no*); children (*number*); education (*primary school or less/ beyond primary school*); and primary occupations (*top 3; open*). We collected more detailed information from households that participated in hunting or trading bushmeat, including: hunters per households (*number*), household participation in trading meat (*yes/no*), destination of meat sold (*inside/outside of community*), and average proportion of meat sold within (vs. outside of) communities (*none [0%], little [5%], some [25%], half [50%], most [75%], all [100%]*). We created a wealth index by scoring household assets, including: house ownership, material of roof and walls, number of rooms, type of toilet, household items, and hired farm laborer (Malleson et al., 2008).

Individual Dietary Diversity

We recorded dietary diversity data for 478 participants using 24-h open recalls followed by a second round of probing for additional food items (Kennedy et al., 2011). We categorized food items into 15 food categories—10 main food categories ([1] grains, white roots and tubers, and plantains, [2] pulses (beans, peas, and lentils), [3] nuts and seeds, [4] dairy, [5] meat, poultry and fish, [6] eggs, [7] dark green leafy vegetables, [8] other vitamin-A fruits and vegetables, [9] other vegetables, and [10] other fruits) and five "other" categories ([1] insects and other small protein foods, [2] red palm oil, [3] other oils and fats, [4] sweets, [5] condiments, other beverages, and seasonings) (FAO and FHI 360, 2016) (**Table S1**). Large invertebrates (e.g., African giant snails and land crabs) were incorporated into the initial meat, fish and seafood category, whereas smaller invertebrates (e.g., small snails, shrimp, and crayfish) were incorporated into the insects and other small proteins category. We added an expanded 30-day recall for animal-based foods where meat, fish, and large invertebrates were disaggregated (**Table S2**). Within each category we further categorized food sources as either imported or produced within the community or collected from the forest. We calculated dietary diversity scores by first summing the 10 main food categories into a score ranging from 0 to 10. We calculated proportion of the respondents reporting consumption of food items from each group, comparing interior and forest-edge communities. We then calculated Minimum Dietary Diversity for Women of Reproductive Age (MDD-W) by sub-setting women of reproductive age (15–49; $n = 232$) and categorizing them as achieving minimum dietary diversity (score ≥ 5; more likely to have adequate micronutrient intakes) or not achieving minimum dietary diversity (score < 5) (FAO and FHI 360, 2016). Mid-upper arm circumference (MUAC), an indicator of short-term nutritional status, was measured to the nearest millimeter (mm) using MUAC tape (Frisancho, 2008). We used standard MUAC cutoffs to further categorize participants as overweight (MUAC $\geq$ 25 cm) or underweight (MUAC $\leq$ 24 cm) (Tang et al., 2013), however pregnancy status was not known for females.

Household Food Security

We ranked 323 households on a Household Food Insecurity Access Scale (HFIAS) based on the prevalence and frequency of experiences of food insecurity (Coates et al., 2007). In each household, we interviewed the individual most involved in food preparation and meals and asked them to respond on behalf of the household. Interview responses were used to quantify experiences of nine household food insecurity access-related conditions within three domains (i.e., anxiety, insufficient quality, and insufficient quantity and physical consequences). We ranked households on the food insecurity access scale by combing prevalence and frequency-of-occurrence to create a score ranging from 0 (secure) to 27 (insecure) (Coates et al., 2007).

Cultural Salience of Bushmeat

To measure the cultural salience of different meat items, we asked key informants to free list animals across multiple domains (e.g., taste preferences and economic value). Following free listing exercises, we used images of wild animals from Kingdon's Pocket Guide of African Mammals (Kingdon, 2005) and standard images of domestic animals and fish sourced from the internet, to ask participants to rank their listed animals.

Data Analysis

We used descriptive statistics to analyze sociodemographic, dietary, and nutritional characteristics of our study population. From dietary recall data, we categorized each food item into food categories (FAO and FHI 360, 2016) (**Table S1**) and calculated the percentage of diets that included at least one food item in each food category. We also calculated the percentage of diets that included food items from each category that were produced or imported and food items that were harvested from the wild (in either forest or farm). We used mixed-effects linear and logistic regression models, in which we incorporated village as a random effect to account for community clustering of non-independent samples, to compare our samples between interior forest and forest-edge communities. For models containing more than one predictor variable, we used backwards elimination of variables and retained only significant variables (at the alpha = 0.05 level) and first-order interactions among significant main effects in the final model. All analyses were performed in RGui 3.4.4 and statistical significance was determined at the alpha = 0.05 level.

Multivariate Analysis of Diet Composition

We examined the multivariate composition of diets, and bushmeat specifically, in deep and forest-edge communities via non-metric multidimensional scaling analysis (NMDS) with Jaccard dissimilarity matrices. We removed unidentified bushmeat and collapsed categories for animals that were not regularly differentiated (e.g., pangolin, monkey, and nocturnal primate species). We tested for differences in compositional dissimilarity (position of the group centroid) using Permutational Multivariate Analysis of Variance (PERMANOVA) and analysis of multivariate homogeneity of group dispersion (average distance of group members to the group centroid) (PERMADISP), both with 999 permutations. To identify the specific food items that characterized deep and

TABLE 1 | Sociodemographic characteristics of respondents.

	Interior (n = 158)	Edge (n = 320)	Total (N = 478)
Women: men (%)	63: 36	68: 32	66: 33
Average age	41.1 ± 14.23	41.6 ± 15.52	41.3 ± 15.09
Family size	5.22 ± 2.75	4.86 ± 2.67	4.97 ± 2.73
Education beyond primary school (%)	60.1	69.1	66.1
Occupation (%)			
Farmer	93.7	90.6	91.6
Harvest NTFPs	39.9	31.6	34.3
Trade goods	12.6	20.3	17.8
Wealth Index	7.41 ± 2.35 (n = 97)	7.76 ± 2.41 (n = 203)	7.65 ± 2.39 (n = 300)

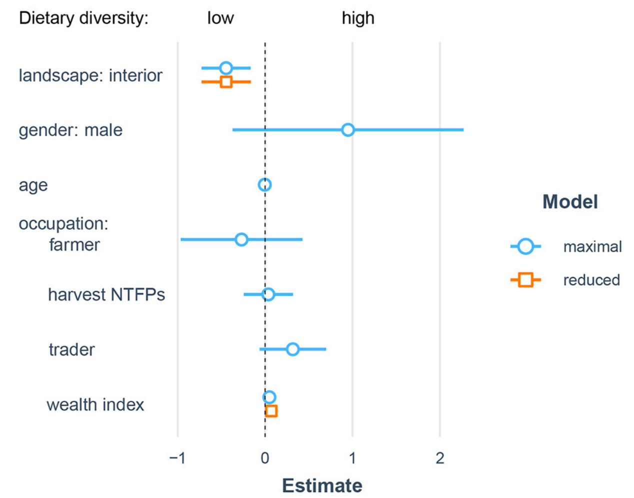

FIGURE 2 | Socioecological predictors of dietary diversity. Results from linear mixed model predicting individual dietary diversity from sociodemographic and landscape differences. Coefficient estimates from full models are shown in blue and coefficients from reduced models retiaing only significant predictors are shown in orange.

forest-edge communities, we used an indicator species analysis. Indicator values (IV) range from 0 to 1, with higher values for stronger indicators. Only food items with IV > 0.3 and $p < 0.05$ were considered good indicators (Dufrene and Legendre, 1997). We performed analyses using the metaMDS, adonis2, and betadisp functions within the *vegan* and *indval* function within *labdsv* package in RGui 3.4.4.

Cultural Domain Analysis

We calculated cultural salience (Smith's S) from ranked free lists produced during key informant interviews, where:

$$S = \sum \frac{inverted\ item\ rank/\#\ items}{\#\ of\ informants}$$

We constructed salience plots to visualize taste preferences and economic values of animals by relating the frequency that each animal was mentioned to the average rank assigned to it. We performed analysis using the *AnthroTools* package in RGui 3.4.4.

RESULTS

Demographics

The primary occupations of our respondents were farming, harvesting of non-timber forest products, and trading in goods (**Table 1**). We found no differences between deep and forest-edge communities with respect to demographics, livelihoods, or household size (**Table 1**).

Dietary Diversity

Dietary diversity was significantly related to village location (X = 9.7, df = 1, $p < 0.01$) and wealth (X = 6.4, df = 1, $p < 0.05$), with a marginally significant interactive effect (X = 3.8, df = 1, $p = 0.05$) such that individuals from wealthier households had marginally higher dietary diversity in forest-edge communities but lower dietary diversity in interior forest communities (**Figure 2**).

Overall, a larger proportion of individuals from interior forest communities reported consuming dark green leafy vegetables (**Table 2**). Individuals living in forest-edge communities reported consuming more pulses (i.e., beans), dairy, fish, eggs, other vegetables, other oils and fats (i.e., non-red palm oil), sweets, and condiments, other beverages, and seasoning (**Table 2**). Interior and forest-edge communities differed in where they sourced food items from each category. Specifically, individuals from interior forest communities reported consuming more meat, poultry, and fish (including skin) collected from the wild, and more cultivated vitamin A-rich fruits and vegetables compared to forest-edge communities (**Table 2**). The opposite trend was true for forest-edge communities, who had a larger proportion of individuals who consumed produced or imported meat, poultry, and fish (including flesh, internal organs, and skin), and more vitamin A-rich fruits and vegetables collected from the forest).

Comparison of Jaccard dissimilarity matrices, built from binary responses to 24-h dietary recalls assessing dietary diversity (n = 15 food categories), showed that individuals from interior forest communities had a different dietary composition than forest-edge communities (PERMANOVA: F = 12.1, df = 1, $p < 0.001$). Intragroup variability did not differ between sites (PERMADISP: F = 0.39, df = 1, $p = 0.52$). A non-metric multidimensional scaling plot shows a degree of dietary similarity (overlapping dietary compositions) but also dietary differences (different group centroids) between interior and forest-edge communities (**Figure 3A**). Dark green leafy vegetables were a significant indicator category characteristic of interior forest community diets (IV = 0.43, $p < 0.001$). Other vegetables (IV = 0.40, $p < 0.05$), fruits (IV = 0.54, $p < 0.01$), insects and other small proteins (IV = 0.51, $p < 0.001$), and condiments (IV = 0.50, $p < 0.05$) were all indicator categories of forest-edge diets (**Figure 3B**).

Comparison of Jaccard dissimilarity matrices, built from animal-source foods reported during 30-day dietary recalls, showed that individuals from interior and forest-edge communities consumed different compositions of meat (PERMANOVA: F = 9.33, df = 1, $p < 0.001$) and had different

TABLE 2 | Consumption of food items and the sources of those foods in diets of deep and forest-edge communities based on 24-h recall data.

	% of diets including food items		% food items produced or imported		% food item collected	
	Interior	Edge	Interior	Edge	Interior	Edge
Grains, white roots and tubers, and plantains	99.4	99.7	100	100	0	0
Pulses (beans, peas, and lentils)	10.1	24.4***	100	100	0	0
Nuts and seeds	82.3	80	64.6	89.1	73.1	57
Dairy	3.2	12.2**	100	100	0	0
Meat, poultry, and fish	87.3	92.8	43.4	73.4**	89.8*	64
Flesh meat	70.9	54.7	0.8	23.3*	100	83.4
Internal organs	14.5	9.4	4.3	27.6*	95.6	73.3
Skin	28.5	25	40	76***	64.4***	25
Fish	50.6	75***	60.0	78.7	47.5	29.1
Eggs	1.3	8.1*	100	100	0	0
Dark green leafy vegetables	77.8*	61.2	77.2	66.7	47.1	46
Other vitamin A-rich fruits and vegetables	10.1	6.6	75**	28.6	25	81.0**
Other vegetables	62.5	74.7*	83.8	88.3	17.2	13.8
Other fruit	43.7	62.2	100	100	0	0
Insects and other small protein	67.1	81.6	97.2	99.6	13.2	12.6
Red palm oil	99.4	96.6	100	100	0	0
Other oils and fats	10.7	24.1***	100	100	0	0
Sweets	13.3	25.6*	100	100	0	0
Condiments, other beverages, and seasonings	94.9	99.1*	100	100	5.3	2.2

**$p < 0.05$, **$p < 0.01$, ***$p < 0.001$ in linear and logistic mixed-effects regression models comparing deep and forest-edge communities with village incorporated as a random effect.*

intragroup variability (PERMDISP: F = 4.86, df = 1, $p < 0.05$). The non-metric multidimensional scaling plot shows a degree of similarity in meat consumed (overlapping dietary compositions) but also dietary differences (different group centroids) between deep and forest-edge communities, with the latter showing higher dispersion (**Figure 3C**). Together, these results show that the core composition of consumed meat was similar in interior and forest-edge communities, and that individuals from forest-edge communities consumed on average a higher diversity of animals. Monkeys (*Cercopithecus* sp.) (IV = 0.41, $p < 0.001$) and porcupine (*Atherurus africanus*) (IV = 0.43, $p < 0.05$) were significant indicator species of interior forest diets, whereas crayfish (IV = 0.52, $p < 0.001$), pigs (IV = 0.34, $p < 0.001$), and cows (IV = 0.33, $p < 0.001$) were indicators of forest-edge diets (**Figure 3D**).

Food and Nutrition Security

Households from interior forest communities exhibited significantly higher household food insecurity access scores, fewer women of reproductive age who achieved minimum dietary diversity scores, and lower average mean upper arm circumference (MUAC) in men (**Table 3**). However, differences in MUAC were not associated with significant differences in the proportion of adults who were categorized as over or underweight (as designated using standard MUAC cutoffs) in interior and forest-edge communities (**Table 3**).

Bushmeat Hunting and Trade

Interior forest communities had a significantly higher proportion of households with bushmeat hunters and/or traders and a higher number of hunters and/or trappers per hunting household compared to forest-edge communities (**Table 4**). Respondents from both locations reported selling meat that they hunted to people within and outside of the community. However, traders from interior forest communities reported selling a relatively larger proportion of meat to people within the communities (**Table 4**).

Cultural Salience of Animals as Food

Seventy-four percent of participants reported a preference for bushmeat, compared to 19% who preferred fish, and 7% who preferred domestic animal meat. Salience scores (Smith's S) for specific animals revealed preferences for similar species across sites. The top five preferred animals in each landscape type were: African brush-tailed porcupine (*Atherus africanus*) (interior forest: $S = 0.35$; forest-edge: $S = 0.20$), pangolin (*Manis* spp.) (interior forest: $S = 0.20$; forest-edge: $S = 0.13$), red river hog (*Potamochoerus porcus*) (interior forest: $S = 0.09$; forest-edge: $S = 0.09$), monkeys (*Cercopithecus* sp.) (interior forest: $S = 0.17$; forest-edge: $S = 0.07$), Ogilby's duiker (*Cephalophus ogilbyi*) (interior forest: $S = 0.04$), and blue duiker (*Cephalophus monticola*) (forest-edge $S = 0.04$) (**Figures 4A,B**).

Similarly, the economic salience of different animals was comparable across landscape types, with large bodied wild animals listed as most valuable: red river hog (interior forest: $S = 0.38$; marginal: $S = 0.26$), Ogilby duiker (interior forest $S = 0.13$; forest-edge $S = 0.08$), African buffalo (*Syncerus caffer*) (interior forest: $S = 0.10$; forest-edge: $S = 0.06$), African forest elephant (*Loxodonta cyclotis*) (interior forest: $S = 0.13$; forest-edge $S = 0.21$), African brush-tailed porcupine (interior forest: $S = 0.06$),

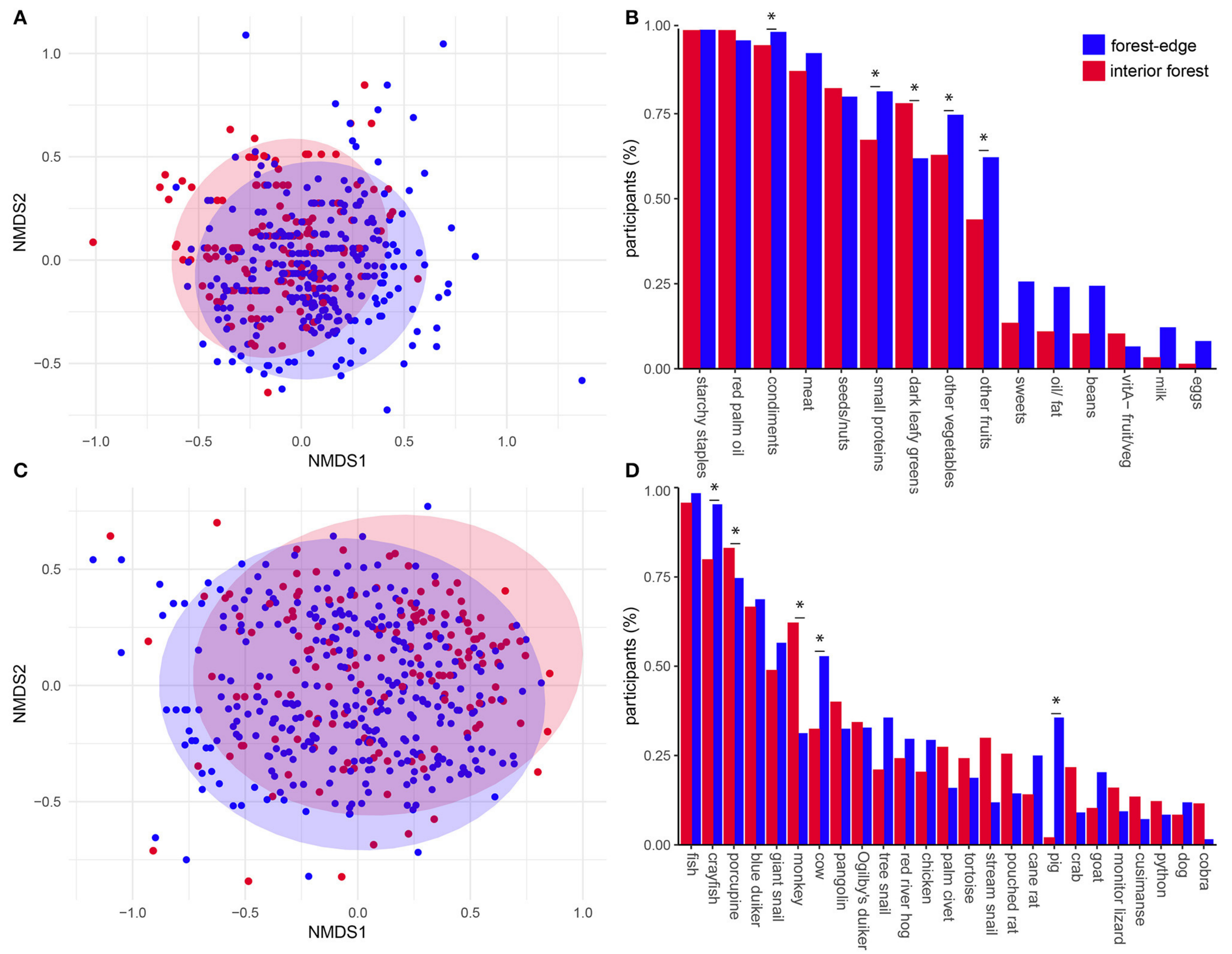

FIGURE 3 | Diet compositions of deep and forest-edge communities. Nonmetric multidimensional scaling (NMDS) plot of the first two axes of Jaccard distance matrices describing dietary composition from 24-h recall data (stress = 0.11, $k = 4$) **(A)** and bushmeat composition from 30-day recall data (stress = 0.16, $k = 3$) **(C)**. Plots are comparing dietary compositions of individuals in interior (red) and forest-edge (blue) forest communities. Barplots compare relative frequency of consumption of food items from each food category **(B)** and different animals **(D)** between locations (items with relative frequency < 0.05 not shown). Asterisks indicate indicator foods with IV > 0.3 and $p < 0.05$ of deep and forest-edge diets.

and drill monkey (*Mandrillus leucophaeus*) (forest-edge: $S = 0.07$) seen as most valuable (**Figures 4C,D**).

Domestic animals appeared only in salience plots as preferred foods in forest-edge communities, and included, chicken, dog, goat, and cow (**Figure 4B**). Similarly, more domestic animals appeared in economic salience plots of forest-edge communities (goat, cow, and pig). Goat appeared in plots derived from both locations, but it was listed more frequently and was assigned higher average rank in forest-edge communities.

DISCUSSION

Across the tropics, forests are being converted to land for subsistence and commercial agriculture, altering local food systems and diets in ways that are currently not well-understood. Our comparison of interior and forest-edge diets highlight the effects of tropical land use changes on local food systems, with implications for understanding the changes occurring at intermediate stages of ecological and dietary transitions at the agricultural-forest frontier. Our results show a high degree of dietary overlap coupled with dietary differences that are associated with better nutrition and food security in forest edges. We argue that nutritional benefits may accrue during intermediate phases of dietary transitions in the tropics—where people retain access to forest resources, obtain access to more agricultural and market goods, and gain the ability to commercialize their food resources. Understanding people's dietary experiences during the early and intermediate stages of deforestation and market integration will be critical, as these early experiences inform dietary and livelihood strategies that further shape ecological and nutritional transitions.

TABLE 3 | Food security and nutritional status, by forest proximity.

	Interior	Forest-edge
Household food insecurity access score (M ± SD)	13.50 ± 5.70 (n = 103)*	8.85 ± 5.84 (n = 220)
Achieved minimum dietary diversity (%)	57% (n = 73)	75% (n = 158)**
Male MUAC (M ± SD)	27.03 ± 2.54 (n = 58)	28.19 ± 3.14 (n = 102)*
Female MUAC (M ± SD)	26.81 ± 3.52 (n = 99)	27.60 ± 2.86 (n = 218)
Underweight (%)	14.0 (n = 158)	12.8 (n = 320)
Overweight (%)	7.6 (n = 158)	7.2 (n = 320)

*p < 0.05, **p < 0.01 in linear and logistic mixed-effects regression models comparing deep and forest-edge communities with village incorporated as a random effect.*

TABLE 4 | Participation in bushmeat hunting and trade, by forest proximity.

	Interior	Forest-edge
Household involvement in bushmeat trade (%)	66.0 (n = 103)*	44.5 (n = 220)
Bushmeat hunters per household (M ± SD)	2.41 ±1.64 (n = 49)**	1.67 ± 0.87 (n = 92)
Hunter households selling bushmeat (%)	93.9 (n = 46)	92.4 (n = 85)
Sell bushmeat inside communities (%)	76.1 (n = 42)	85.7 (n = 70)
Sell bushmeat outside communities (%)	38.8 (n = 42)	59.8 (n = 92)
Proportion of meat sold within communities (%)	41.2 (n = 32)**	23.4 (n = 58)

*p < 0.05, **p < 0.01 in linear and logistic mixed-effects regression models comparing deep and forest-edge communities with village incorporated as a random effect.*

Although forest foods contributed to diets across all sites, we observed fewer forest foods in diets of people living in areas with more deforestation and increased market access. In contrast to forest-edge communities, interior forest communities consumed more dark green leafy vegetables and bushmeat. These observed dietary changes are, to a degree, similar to what has been described during dietary transitions following integration into market economies. Similar to conservation zones with rapid commercial agricultural expansion in Laos (Broegaard et al., 2017), we found that more people in forest-edge communities consumed animal-based foods that were not sourced from the wild. We also observed integration of more processed foods, sweets, and fats, which is similar to dietary transitions described in contemporary hunter-gatherers (Popkin, 2004; Kuhnlein et al., 2009; Crittenden and Schnorr, 2017; Reyes-García et al., 2019). However, we found that interior and forest-edge zone communities were equally likely to consume animal-based foods overall (e.g., meat, protein, and fish), but that forest-edge diets included more beans, dairy, fish, eggs, and other vegetables. Small proteins were an indicator food of forest-edge communities, which can be best explained by high consumption of dried crustaceans, locally referred to as "crayfish," that are obtained from markets and imported into communities. These findings contrast with dietary transitions described in hunter-gatherer groups, which are characterized by decreased availability of nutritionally important foods (e.g., fruits, vegetables and animal foods) with integration into market economies (Popkin, 2004; Kuhnlein et al., 2009; Crittenden and Schnorr, 2017; Reyes-García et al., 2019). The differences between our study and "typical" hunter-gatherer transitions could be reflective of differences in livelihood strategies (e.g., hunter-agriculturalist) and/or the degree of market integration already present in interior forest communities, while also indicative of non-linear dietary responses to land use change and market integration.

Dietary differences between locations were associated with higher dietary diversity, increased measures of protein, energy, and micronutrient status (e.g., MUAC in men and MDDS-W), and improved food access (i.e., low HFIAS) in forest-edge communities. These results are contrary to previous studies, which found increased dietary diversity in isolated hunter-gatherers compared to close communities with increased market integration (Reyes-García et al., 2019), positive associations between forest use, tree cover, and dietary diversity (Powell et al., 2011), and negative effects of land use change on quality of nutrition in areas adjacent to conservation zones (Broegaard et al., 2017). However, our results are similar to other studies showing improved dietary diversity associated with market access (Sibhatu et al., 2015; Koppmair et al., 2017), and support the notion that market access may be more important for dietary diversity than forest proximity, at least in early and intermediate stages of deforestation. We also note however, that we did not measure differences in agricultural diversity between these sites, which is shown to have a positive effect on dietary diversity (Jones, 2017). Interestingly, while we found no systematic differences between sociodemographic composition of our study samples, we did find a marginal interactive effect of wealth on the relationship between dietary diversity, such that wealth appeared to only contribute to improved dietary diversity in forest-edge communities. This further highlights the importance of market access in the translation of wealth to improved nutrition.

Our results revealed some additional and unexpected trends. For example, we found that vitamin-A rich fruits and vegetables (e.g., bush mango [*Irvingiaceae*]) were wild-sourced more in forest-edge than interior forest communities. Although contrary to our expectations, this finding may reflect higher availability of bush mango in agroforest areas. Agroforest and fallow areas are known to be important for obtaining wild foods and may contribute to increased dietary diversity in forest-edge areas (Powell et al., 2011). Alternatively, this could indirectly reflect widespread trade of bush mango seeds, known locally as "ogbono" and used in preparing Nigerian soups. Bush mango is mass-harvested in agroforests and in protected and unprotected forest areas in this region, with people setting up forest camps for the primary purpose of harvesting bush mango. The bush mango fruit is typically discarded, but sometimes consumed opportunistically when people are processing the fruit for the seed. Thus, increased consumption of wild vitamin-A rich fruits in marginal communities could reflect increased

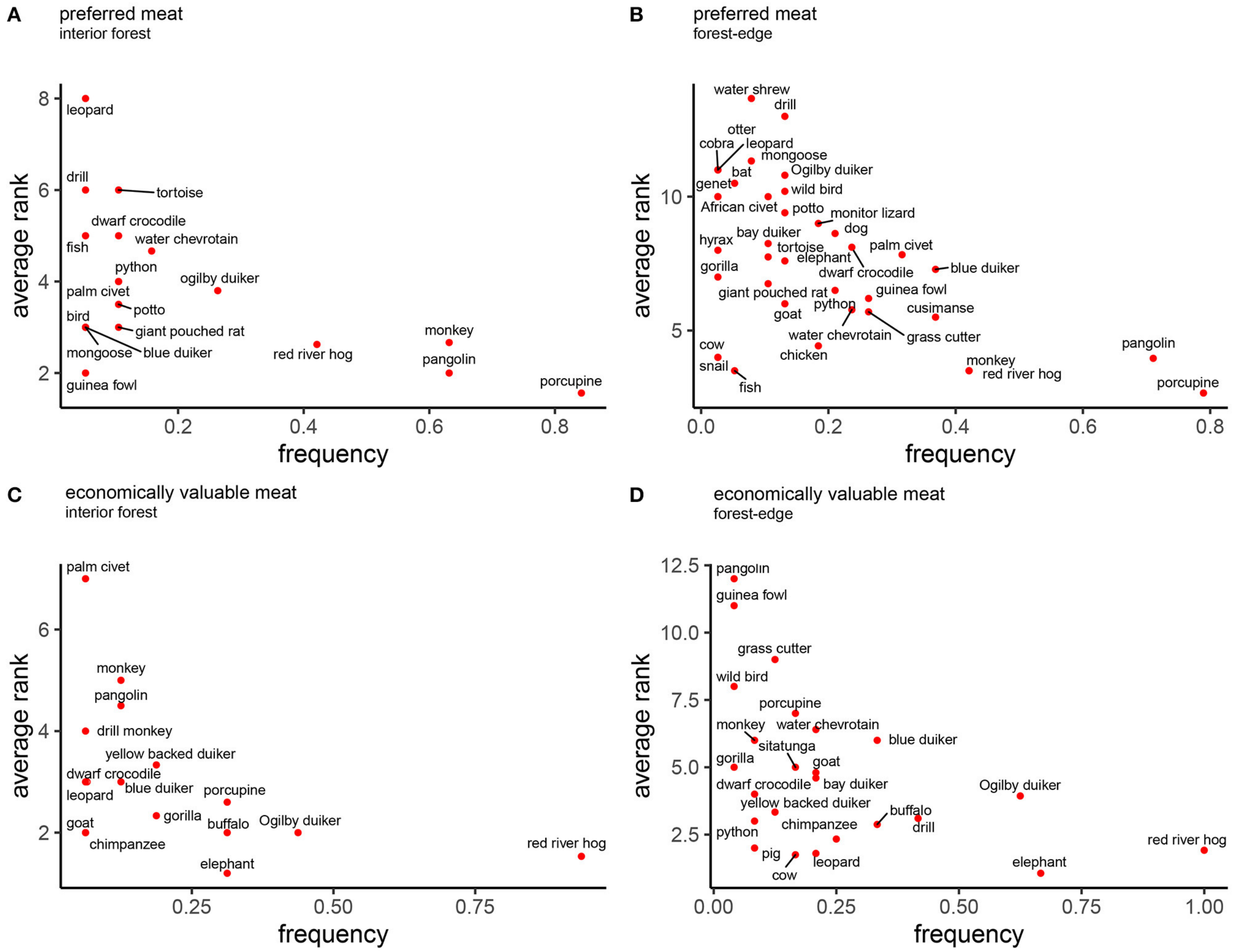

FIGURE 4 | Salience of animals as food and economic resource, by forest proximity. Salience plots of the frequency an animal was mentioned (x-axis) and the average rank assigned to each animal (y-axis) during free-listing exercises with key informants. Plots show preferred animals **(A,B)** and most economically important **(C,D)** animals in the lower right-hand corner, comparing deep **(A,C)** and marginal **(B,D)** forest communities.

handling and opportunistic consumption of bush mango in areas with better access to markets, demonstrating how commercial trade might affect diets, even in small and unexpected ways. Overall, the pathways by which forest-edge households achieve improved food access (e.g., direct subsistence from forest, agricultural, and market goods, or purchasing power gained from commercialization of these goods) is variable across food categories and systems.

Despite interior forest communities having more households that hunted and more hunters per household, forest-edge households sold a higher proportion of the meat they hunted to people outside of their communities. This switch toward income-driven hunting did not appear to result in nutrition-income trade-offs, likely due to the availability of alternatives. Interior and forest-edge communities had diets with similar proportions but different compositions of animal-based foods. Meat in interior forest diets was more likely to come from the wild than in forest-edge communities. Specific indicators of interior forest diets were porcupines and monkeys, whereas indicators of forest-edge diets were dried crustaceans, and domestic pig and cow meat/skin, which were imported into communities by traders. These findings align with previous studies showing that bushmeat consumption declines along the rural to urban gradient, being replaced by domestic and processed meat and fish (Van Vliet et al., 2015). Unlike those studies, however, dietary differences we documented were not associated with nutritional inadequacies in forest-edge communities (Sarti et al., 2015; Van Vliet et al., 2015), potentially because these communities still retained access to forests and bushmeat. However, hidden nutrition-related consequences could accrue via putative differences in micro and macro nutrient composition of wild animals compared to domestic animals and fish, though these are not well-understood (Cawthorn and Hoffman, 2015).

Differences in bushmeat consumption in interior forest communities may reflect differences in availability (i.e., animal biomass) and/or access (e.g., affordability). However, evidence from Central Africa indicates that mammalian biomass can actually be higher in marginal rainforest zones, despite higher biodiversity in interior forest zones (Fa et al., 2015). Market vicinity also influences rates of trade in bushmeat, with increased proximity related to higher extraction rates and concentration on large bodied species in the Amazon (Espinosa et al., 2014). If supply is limited, market proximity may reduce access to bushmeat within local communities, when profit margins for selling bushmeat are high. Reduced consumption of bushmeat in forest-edge communities could therefore reflect differences in availability due to ecological degradation associated with deforestation, or reduced access to the meat when hunters and traders prefer to sell outside of the community at higher profit margins.

Markets not only influence trade in goods, but also the values and taste preferences attached to those goods, which may accelerate dietary transitions or preserve the use of traditional foods (Bowles, 1998). Our results showed that the cultural salience of animals was similar across communities but differed across domains. Bushmeat was preferred and had more economic value than domestic animals and fish in both deep and forest-edge communities. Communities shared four out of five of the same preferred species (porcupine, pangolin, monkey, and red river hog) and economically valuable species (red river hog, Ogilby's duiker, African buffalo, and African forest elephant). While the importance of bushmeat likely has much to do with availability, during several interviews key informants referred to domestic animals as "dirty" compared to bushmeat which is "natural" and "sweet" (meaning it has good taste) as reason for their preference. Overall, bushmeat consumption in our study communities is shaped, in part, by preference for bushmeat over domestic species. This preference preserves the use of wild animals, even when other components of the diet differ.

Differences in consumption of bushmeat in forest-edge communities were mirrored by slight differences in value orientation toward domestic animals. Although domestic animals were not highly salient in either domain, they were listed as preferred species in forest-edge communities alone. Similarly, more domestic animals were listed as economically salient in forest-edge communities (e.g., goat, cow, and pig). Goat was listed in both interior and forest-edge communities but was listed more frequently and assigned higher rank in forest-edge communities. These data suggest that preferences can shift toward integration of domestic species as transitions progress. Importantly, rural diets are heavily intertwined with livelihood choices. Although bushmeat is highly preferred, hunting within this region, and in many of the same communities, is considered a low-merit livelihood described as full of suffering and stress, unpredictable, and something people turn to for lack of better alternatives (Friant et al., 2015). Thus changes in livelihood opportunities may further modify consumption practices away from bushmeat consumption (Nasi et al., 2011; Van Vliet et al., 2015). However, high demand for bushmeat by urban populations (Fa et al., 2006; Macdonald et al., 2012) show that even when domestic animals are integrated into daily diets, preferences for bushmeat are maintained, and economic incentives from urban demand will motivate people to continue to hunt.

Heavy regional involvement in the bushmeat trade is associated with wildlife declines and expected species extinctions that may decrease availability of this preferred and nutritionally rich resource (Fa et al., 2002, 2006; Ripple et al., 2016). Indeed, hunters report having to travel further distances and stay longer in the forest to obtain meat, and community members report reduced availability of wild fish due to the use of unsustainable fishing practices (e.g., use of poison and dynamite in streams). When faced with declining availability of meat, especially during lean seasons, interior forest communities have limited ability to supplement wild resources with domestic and imported alternatives. Forest-edge communities may therefore have a dietary advantage, in that they are able to switch between consumption of wild and domesticated meat. Thus, forest-edge communities may be better able to cope with declines in bushmeat by importing meat and using capital from traded goods. Meanwhile, when households lack funds, they may still fall back on forest resources for food in times of need. Indeed, results from Congolese agricultural communities indicate that wild foods play a small role in household consumption but a major role in household income with 90% of bushmeat and fish sold at the market and increased value of these resources during the lean season (de Merode et al., 2004).

Overall, increasing commercialization of forest resources, coupled with high rates of extraction and land conversion in this region is unsustainable (Fa et al., 2006; Schoneveld, 2014), and our data support the notion that ensuing ecological change may disproportionately affect different members of society (Myers et al., 2013). Our results imply that continued heavy extraction from communities for sale of bushmeat would more heavily impact the diets of interior forest communities that lack alternatives. Improved access to markets, when coupled with forest protection, could help enhance dietary diversity and preserve the use wild foods for rural communities. Inclusion of alternative animal-based foods, especially in interior forest communities, will be important for maintaining high quality diets in the face of increased deforestation, agricultural expansion, and improved conservation efforts. Although cultural preference for wild foods is often seen as a barrier to acceptance of new or alternative foods, our results indicate that food preferences may shift as alternatives are introduced and become more culturally salient. However, access to alternative meat sources in rural forested communities may have very little effect on hunting, given that in the presence of alternatives people tend to shift to income-driven hunting and supplement their diets with alternatives. We argue that forest protection and economic alternatives, alongside improved access to alternative animal-based, will be critical for protection of bio- and dietary diversity.

Our study offers an in-depth analysis of food systems in a region of Nigeria undergoing rampant and unregulated environmental change. However, our study has several limitations, including non-random sampling of a small number of communities ($n = 6$) and lack of associated ecological and landscape data, which together limit the generalizability of our results. Due to logistical constraints of accessing our remote study communities, and the potential that these communities would be uncooperative based on the complicated history of their relationship with the park, we strategically selected a small number of study sites that were representative of the area, but where we could feasibly carryout the study (i.e., connections with people who could favorably introduce us and our research and ability to stay in communities for up to a month). Despite the importance of Cross River as a unique biodiversity reserve in Africa, limited research effort in these areas has severely limited availability of data on deforestation and land-use changes, and thus prohibited a quantitative comparison of the ecological differences between our interior and forest-edge study communities. We have therefore evoked the complicated history of CRNP to aid in explaining site specific differences. Despite the myriad challenges faced during the formation of Cross River National Park, its existence has so far prevented interior forest areas from experiencing the large-scale land conversion that is typical of forest-edge communities. CRNP has also prevented the construction of access roads that link interior communities to major roads and markets. Despite these key differences, we cannot assume that food systems are homogenously impacted based on proximity to CRNP alone. For example, communities in the northern Okwangwo division of the park have been more heavily impacted by conservation policies due to the presence of the Cross-River Gorilla, whereas communities in the southern Oban division have been more heavily impacted by industrial agriculture. Thus, while our data describe the responses of food systems to differences in locations and landscapes, we cannot infer causal processes due to the multiple interacting pathways by which communities might be affected by and respond to tropical land-use change.

CONCLUSIONS

Diets at the agricultural-forest frontier of southern Nigeria are characterized by fewer forest-based resources, specifically nutrient-rich foods such as bushmeat and dark green leafy vegetables. Bushmeat was consumed less but traded more often by in forest-edge communities, illustrating potential nutrition-income tradeoffs. However, forest-edge communities appear to compensate for the reduction of forest foods in diets by incorporating alternative animal-based foods (e.g., fish and domestic animals) and other nutritionally important foods, including small proteins, beans, dairy, eggs, and other fruits and vegetables. These data also highlight the heterogeneity in the effect of tropical land use change in diets overtime, suggesting that in the intermittent stages of tropical deforestation, communities experience the best of two worlds—the agricultural and forest frontier. In our study sites, these dietary differences led to improved nutrition and dietary diversity in forest-edge communities. We explain these differences through trade-offs between market access, agricultural expansion and deforestation, and conservation policies. Understanding "micro-transitions" at intermediate stages of land use change will be necessary to provide a clearer picture of the trajectory of livelihood responses to ecological transitions and their associated consequences for human and ecosystem health.

DATA AVAILABILITY STATEMENT

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

ETHICS STATEMENT

Nigeria Health Research Ethics Committee (# NHREC/01/01/2007-18/05/2017), City University of New York Integrated Institutional Review Board (#2016-0352), and The Pennsylvania State University Institutional Review Board (#00011190) approved all research activities prior to the initiation of data acquisition. Study activities were also approved by Nigeria National Parks Service and community leaders of each study community. Methods were carried out in accordance with the relevant guidelines and regulations of these institutions and with oral informed consent from all subjects.

AUTHOR CONTRIBUTIONS

SF designed the study, contributed to data collection, cleaned and analyzed data, and drafted the manuscript. WA, AA, NI, and OO contributed to data collection and analysis. JR, TG, JJ, CA, and DO contributed to study design.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2019.00113/full#supplementary-material

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Forest Conservation: A Potential Nutrition-Sensitive Intervention in Low- and Middle-Income Countries

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Childhood undernutrition yearly kills 3.1 million children worldwide. For those who survive early life undernutrition, it can cause motor and cognitive development problems that translate into poor educational performance and limited work productivity later in life. It has been suggested that nutrition-specific interventions (e.g., micronutrient supplementation) that directly address the immediate determinants of undernutrition (e.g., nutrient intake) need to be complemented by nutrition-sensitive interventions that more broadly address the underlying determinants of undernutrition (e.g., food insecurity). Here, we argue that forest conservation represents a potentially important but overlooked nutrition-sensitive intervention. Forests can address a number of underlying determinants of undernutrition, including the supply of forest food products, income, habitat for pollinators, women's time allocation, diarrheal disease, and dietary diversity. We examine the effects of forests on stunting—a debilitating outcome of undernutrition—using a database of household surveys and environmental variables across 25 low- and middle-income countries. Our result indicates that exposure to forest significantly reduces child stunting (at least 7.11% points average reduction). The average magnitude of the reduction is at least near the median of the impacts of other known nutrition interventions. Forest conservation interventions typically cover large areas and are often implemented where people are vulnerable, and thus could be used to reach a large number of the world's undernourished communities that may have difficult access to traditional nutrition programs. Forest conservation is therefore a potentially effective nutrition-sensitive intervention. Efforts are needed to integrate specific nutrition goals and actions into forest conservation interventions in order to unleash their potential to deliver nutritional benefits.

Keywords: demographic and health surveys, ecosystem services, food security, height-for-age, malnutrition, planetary health, partial identification, stunting

INTRODUCTION

Childhood undernutrition is a global problem, a factor responsible for the death of 3.1 million children under the age of five annually, or roughly 45% of all child deaths (Black et al., 2013). Prevalence is higher in low and middle-income countries than elsewhere (Perez-Escamilla et al., 2018). Childhood undernutrition, particularly from conception to a child's second birthday, has been related to motor and cognitive development problems that have adverse effects later in life, such as poor school performance, limited learning, and work capacity, decreased economic productivity, and shorter adult stature [Almond and Currie, 2011; Currie and Vogl, 2013; United Nations Children's Fund (UNICEF), 2013]. In addition to the high prevalence and detrimental consequences of childhood undernutrition, the fight against it is only growing more difficult as growing human population, volatile food and oil prices, conflicts and governance crises, and the increasing human perturbation of Earth's natural systems (e.g., climate, land cover) all threaten the food system (Godfray et al., 2010).

Given these challenges, it has been suggested that nutrition-specific interventions—those addressing the immediate causes of undernutrition—(e.g., nutrient supplementation, food fortification) need to be complemented by nutrition-sensitive interventions that address the underlying determinants of undernutrition and incorporate specific nutrition goals and actions (e.g., agriculture, social safety nets; Ruel and Alderman, 2013). Underlying determinants of nutrition include household income, food security, and access to services affecting nutritional status (i.e., anthropometry, micronutrient status). Nutrition-sensitive interventions often are implemented at large scale with intention to reach vulnerable populations. They therefore can also serve as vehicles to improve both the coverage and targeting of delivery of nutrition-specific interventions (Ruel and Alderman, 2013). A recent systematic review conducted by Hossain et al. (2017) indicates that greater effectiveness has been observed when programs combine nutrition-specific and nutrition-sensitive interventions. Investments in development and implementation of nutrition-sensitive interventions have increased in the latest decade (Ruel and Alderman, 2013).

Here, we investigate whether forest conservation represents a potentially important but overlooked nutrition-sensitive intervention. We first describe the underlying determinants of undernutrition that can be addressed by forests. We then use a unique multi-country database to examine effects of forests on nutritional status, particularly stunting (i.e., low height-for-age), which is a common manifestation of long-term childhood undernutrition (Hossain et al., 2017). Finally, we discuss the features of forest conservation interventions that make them potentially effective nutrition-sensitive interventions and suggest ways to increase their nutrition sensitivity.

UNDERLYING DETERMINANTS OF UNDERNUTRITION ADDRESSED BY FORESTS

A number of studies examine effects of forests on underlying determinants of undernutrition, forming intermediate outcomes along the pathways between forests and nutritional status (**Figure 1**). Forests supply ecosystem services important to nutrition. Among these services, forest foods are collected by a large number of rural forest households in low- and middle-income countries. A study covering 24 countries indicates that over 55% of rural households with moderate-to-good access to forest resources collect forest food products (e.g., diverse species of animals, plants, and mushrooms) for subsistence (Hickey et al., 2016). For the top forest dependent communities across these countries, forest food products provide nearly 15% of the recommended quantities of fruits and vegetables, and 106% for meat and fish (Rowland et al., 2017). Fungo et al. (2016) report that forest foods contribute 93% of daily vitamin A intake of women in rural forest-dependent communities in Cameroon.

Forest food and non-food products (e.g., timber and non-timber forest products) form a significant portion of the income (in-kind and cash) of rural forest households across low- and middle-income countries. A synthesis of 51 case studies suggests that, on average, forest products compose 22% of forest household total income (Vedeld et al., 2007), a percentage similar to that found by a more recent study across 24 low- and middle-income countries (Angelsen et al., 2014). In addition to being used for subsistence, forest products are also sold for cash income (Angelsen et al., 2014), which can be invested in household nutrition through food purchase, protection against or treatment for diseases (e.g., diarrhea, measles) that affect nutritional status.

Another nutrition-relevant service that forests provide is habitat for pollinators. Seventy-five percent of leading food crops, accounting for 35% of the world's crop production, depend to varying extents on pollinators (Klein et al., 2007). Pollination is also crucial for the provision of essential micronutrients. For example, 98, 70, and 55% of the available vitamin C, vitamin A, and folate, respectively, in the world's leading crops are produced by pollinated plants (Eilers et al., 2011). In addition to pollinators' roles in subsistence agriculture, they substantially contribute to the cash income of millions of rural and poor people [Intergovernmental Science-Policy Platform

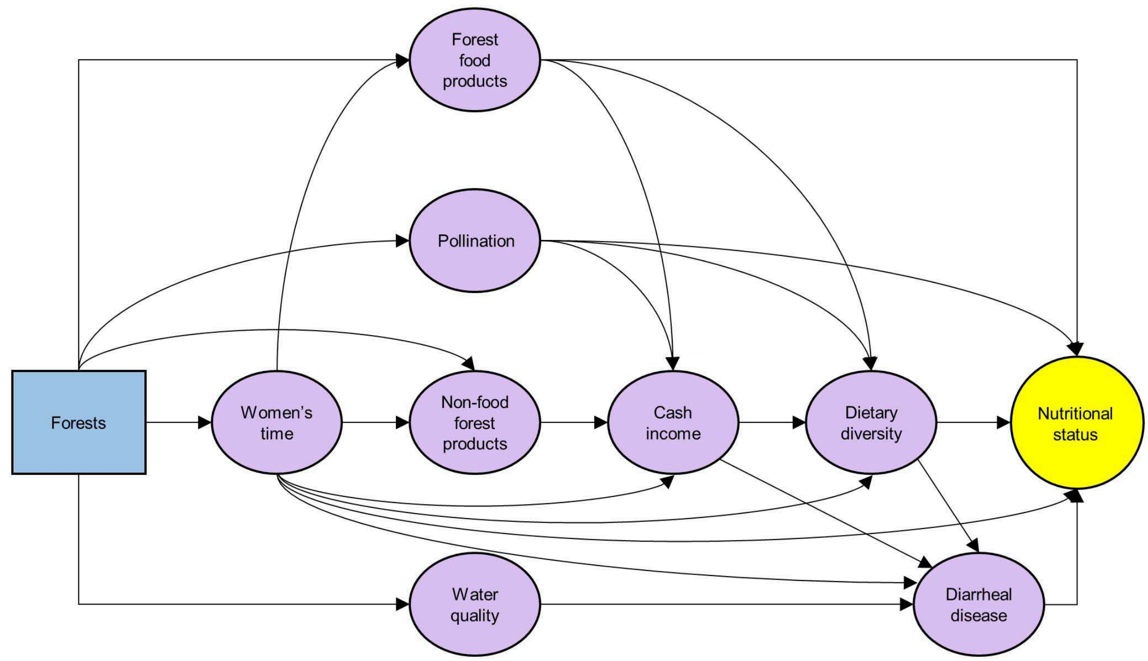

FIGURE 1 | Mechanistic pathways linking forests and nutritional status (purple oval boxes: underlying determinants of nutrition addressed by forests).

on Biodiversity and Ecosystem Services (IPBES), 2017]. For example, many of the world's leading export crop products from rural low- and middle-income countries are pollinator-dependent (e.g., coffee and cocoa) [Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), 2017]. The additional cash income arising from pollination service can be used to improve household nutrition.

Women's empowerment is a key underlying determinant of childhood nutrition that could be addressed by many nutrition-sensitive interventions (Ruel and Alderman, 2013), including forest conservation. Particularly, in many low- and middle-income countries, women are the primary collectors of non-timber forest products (e.g., forest food, firewood, fodder; Sunderland et al., 2014). Reduced access to these products due to deforestation and forest degradation increases time and energy women spend collecting them, shifting their time and energy away from food preparation, more careful child feeding behaviors, income generation, and health care (Agarwal, 2009; Johnson et al., 2013). For example, Wan et al. (2011) reported that in India, women used to walk 1–2 km every day to gather sufficient firewood for cooking. Eight years later, after deforestation, they needed to walk 8–10 km for the same activity. Such a shift in the use of time and energy by women can negatively affect the nutrition of household members (Ruel and Alderman, 2013).

Forests are also linked to reduced risk of diarrheal disease (Pattanayak and Wendland, 2007; Johnson et al., 2013; Herrera et al., 2017), which is a strong underlying determinant of stunting in children (Checkley et al., 2008). For example, a study across 35 low- and middle-income countries indicates that, in rural areas, a 30% increase in upstream tree cover is associated with 4% reduction in the probability of downstream incidence of diarrheal disease (Herrera et al., 2017). The reduced diarrheal disease could be at least partly due to the improvement of drinking water quality by forests. Forests have been shown to remove pathogens and sediments from water (Ensign and Mallin, 2001; Cunha et al., 2016). Water filtration by forests is likely to be particularly valuable for the 663 million people, living primarily in low- and middle-income countries, who use unimproved drinking water sources [World Health Organization (WHO), 2017].

Dietary diversity is another underlying determinant of undernutrition affected by forests (Ickowitz et al., 2014; Galway et al., 2018; Rasolofoson et al., 2018; Rasmussen et al., in press). In a study across 27 low- and middle-income countries, Rasolofoson et al. (2018) estimate that exposure to forest leads to at least 25% greater dietary diversity in children exposed to forest than non-exposed children. Rasmussen et al. (in press) indicate, in a study across five African countries, that forest configuration across landscapes, not just forest coverage, influences dietary diversity. High dietary diversity correlates significantly with better nutritional status in several low- and middle-income countries (Arimond and Ruel, 2004; Steyn et al., 2006). Forests could therefore improve childhood nutritional status through their effects on dietary diversity. Increased dietary diversity can also affect nutritional status via reducing risk for diarrheal disease. More diverse diets are more likely to provide adequate levels of micronutrients (Moursi et al., 2008; Zhao et al., 2017), which can shield children against infectious diseases. In particular, through its role in the immune system, vitamin A—the consumption of which is positively affected by exposure to forest (Johnson et al., 2013; Rasolofoson et al., 2018)—decreases susceptibility to diarrheal disease (Semba, 1999; Villamor and Fawzi, 2005) and thus lowers the probability of stunting (Checkley et al., 2008).

The weight of evidence therefore tilts toward forests addressing underlying determinants of nutritional status. However, effects on underlying determinants do not necessarily translate into effects on actual measures of nutritional status (Ruel and Alderman, 2013). Empirical evidence about effects of forests on nutritional status is therefore needed, but unfortunately such evidence is rare (e.g., Golden et al., 2011; Johnson et al., 2013). To strengthen the evidence about effects of forests on nutritional status, we examine effects of exposure to forest on prevalence of child stunting across 25 low and middle-income countries in Africa, South America, and Southeast Asia (**Figure 2**). We also explore effects of forests in view of the impacts of different nutrition interventions on stunting, in order to shed light on the potential of forest conservation as a nutrition-sensitive intervention.

MATERIALS AND METHODS

Stunting

Height-for-age represents the linear growth achieved at the age of measurement. Prevalence of child stunting is the percentage of children whose height-for-age values fall below two standard deviations (-2 Z-scores) from the median height-for-age of a reference population [World Health Organization (WHO), 2006]. In 2011, stunting affected 165 million children across the globe (Black et al., 2013). In a synergistic association with infectious diseases (diarrhea, pneumonia, measles), stunting is responsible for a third of child deaths due to undernutrition, making it one the deadliest manifestations of undernutrition, particularly in South Asia and sub-Saharan Africa (Black et al., 2013).

Our data come from the Demographic and Health Surveys (DHS) program of the [United States Agency for International Development (USAID), 2018]. The DHS program has collected demographic and health information across more than 90 developing countries. We used the DHS stunting information that is based on the Centers for Disease Control and Prevention (CDC) reference population [United States Agency for International Development (USAID), 2013]. Our dataset comprises 59,378 children under the age of five across 25 low- and middle-income countries surveyed in different years between 2006 and 2013 (**Supplementary Material Table S1**).

Exposure to Forest

We defined exposure to forest following and using the same data as Rasolofoson et al. (2018). Our forest data are from the MODIS Vegetation Continuous Field products at 250 m spatial resolution (DiMiceli et al., 2011). Forests are areas with at least 40% tree coverage [United Nations Environmental Programme (UNEP), Food and Agriculture Organization of the United Nations (FAO), and United Nations Forum on Forests Secretariat (UNFF), 2008]. The georeferenced communities surveyed by the DHS program (referred to as "clusters" in DHS documents) were integrated with the spatial forest data. Each child was assigned to the forest cover of the year when they were surveyed or to the 2010 forest cover when the survey took place in or after 2010 as the MODIS Vegetation Continuous Field products ended in 2010.

We defined children exposed to forest as those living in communities within 3 km of the nearest forest edge and with at least 30% of the land within 5 km buffer around the community

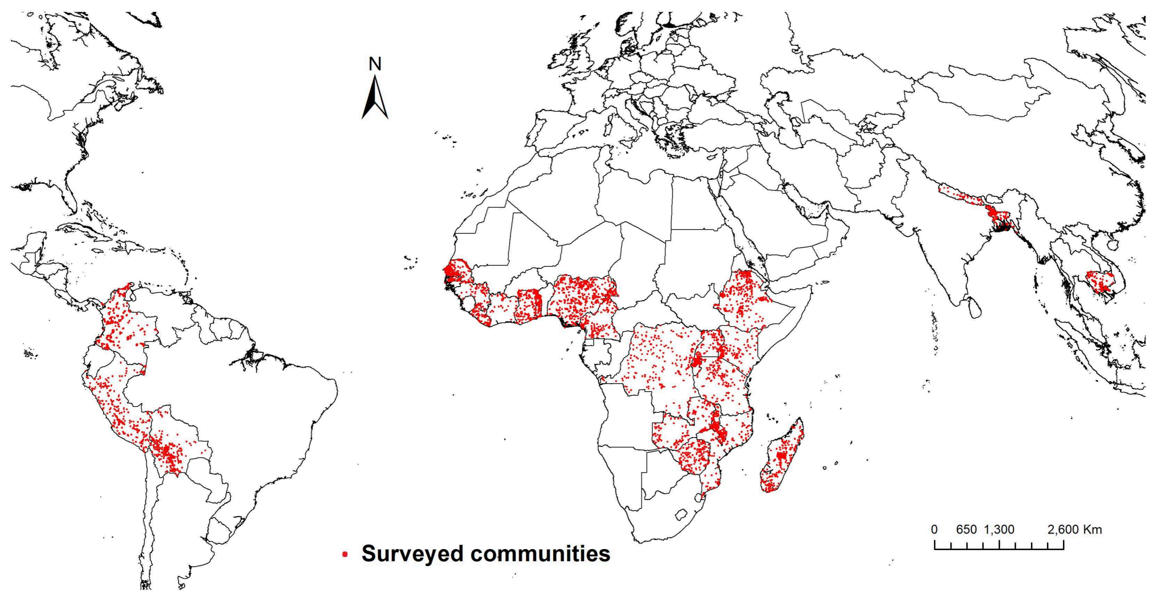

FIGURE 2 | Communities included in the estimation of effects of forest on child stunting. Data are from the Demographic and Health Surveys (DHS) program of the [United States Agency for International Development (USAID), 2018].

centers covered by forests. We defined children not exposed to forests as those living farther than 8 km from of the nearest forest edge. The criteria for the definitions of exposure and non-exposure to forests are based on the average distance forest people in low- and middle-income countries walk to come to the closest forest to collect forest products, foraging distance of pollinators (a mechanism through which forests affect nutrition), and the uncertainty associated with the locations of communities in our data (see full details and data description in Materials and Methods of Rasolofoson et al., 2018). In fact, the locations of communities in DHS were randomly displaced up to 5 km to protect anonymity of survey respondents. This is the reason why we used the 5-km buffer around the community centers in our definition of exposure to forest. Moreover, this displacement also means that communities located between 3 and 8 km from forest edges could actually be within 3 km of forest edges and thus their children could be exposed to forest according to our definition of exposure to forest. This is why we defined children not exposed to forest as those living further than 8 km from forest edge and we excluded children of communities located between 3 and 8 km from forest edges. We identified 13,927 children exposed and 45,451 children non-exposed to forest.

Identification of the Effect of Forests on Prevalence of Stunting: Partial Identification

The effect of exposure to forest on stunting prevalence is the difference between the prevalence of stunting for children exposed to forest and the counterfactual prevalence of stunting had these same children not been exposed to forest. The former is the observed percentage of stunted children among those exposed to forest. The latter, i.e., the counterfactual, is not observed. We must thus assume that percentage of stunted children among a comparison group not exposed to forest represents the counterfactual. The credibility of our effect estimate depends on the plausibility of the assumptions invoked to identify the counterfactual. A precise point estimate of effect (e.g., regression coefficients) often requires non-transparent and strong identifying assumptions about the counterfactual and thus, is of limited credibility (Manski, 2011; Ferraro and Hanauer, 2014; McConnachie et al., 2016). We used the partial identification approach (Manski, 2003), which considers observed data on the characteristics of children exposed and non-exposed to forest and invokes weak, but plausible, identifying assumptions to generate ranges—delimited by lower and upper bounds—within which the counterfactual and thus, the estimate of effects of exposure to forest on stunting can occur.

Without making any assumptions, we know that the counterfactual prevalence of stunting for the children exposed to forest, had they not been exposed to forest, would be greater than 0% (no stunted child) and smaller than 100% (all children stunted). The difference between the prevalence of stunting for children exposed to forest and these two extreme counterfactual values, respectively, give the upper and lower "no-assumption" bounds of the effect of exposure to forests.

We then invoked the monotone treatment selection (MTS) assumption (Manski and Pepper, 2000). MTS posits that either positive or negative selection bias is plausible (McConnachie et al., 2016). Positive selection bias occurs when children exposed to forest, had they not been exposed to forest, would have stunting prevalence (counterfactual) greater than that for children not exposed to forest. Negative selection bias occurs when the counterfactual stunting prevalence for children exposed to forest is smaller than the stunting prevalence for children not exposed to forest. For our study, positive selection is plausible. Forests are often located in marginal lands with low agricultural potential, far from infrastructure (e.g., roads, markets), and with high poverty (Sunderlin et al., 2005). These forest related characteristics are not favorable for nutrition (Rasolofoson et al., 2018). Children exposed to forest are therefore likely to have characteristics less favorable for nutrition than children not exposed to forest (Rasolofoson et al., 2018)—as generally confirmed in our data (**Table 1**). It is thus plausible to assume that the counterfactual stunting prevalence for children exposed to forest, had they not been exposed to forests, would be greater than the stunting prevalence for children not exposed to forest. Therefore, we moved the lower bound of the counterfactual from 0% (no assumption) to the stunting prevalence for children not exposed to forest. In turn, the upper bound of the range of the effect estimate becomes the difference between the prevalence of stunting for children exposed to forest and that for children not exposed to forest.

To test the statistical significance of the upper and lower bounds of the effect estimate (i.e., the differences between the prevalence of stunting for children exposed to forest and the upper or lower bounds of the counterfactual), we used linear regressions with stunting statuses (stunted or not stunted) of children as dependent variable and forest exposure (exposed or not exposed to forest) as independent variable. These regressions are equivalent to using independent t-tests to compare the prevalence of stunting for children exposed to forest and the upper or lower bounds of the counterfactual (Pandis, 2016). Comparing means of a binary variable (stunted or not stunted) between two groups (exposed or not exposed to forest) with t-test is similar to comparing proportions (percentages) with proportion z-test when the sample size is large (Park, 2009), as in the case of our analyses. We clustered the standard errors at the community level. We computed bootstrap confidence intervals. We did the analyses with the R "clusterSEs" package (Esarey, 2017).

Effects of Forests and Impacts of Different Nutrition Interventions on Child Stunting

To put the result of the partial identification approach into perspective, we plotted the conservative bound of the estimate of effects of exposure to forest on stunting with the estimated impacts of different interventions on stunting investigated in studies systematically reviewed in Hossain et al. (2017). Hossain et al. (2017) estimated the impacts of the interventions they reviewed as the average annual rate of reduction (AARR).

TABLE 1 | Means and standard deviations (in parentheses) of characteristics of children exposed vs. non-exposed to forest.

Variable*	Children exposed to forest	Children non-exposed to forest	Means difference	95% confidence interval
Stunting prevalence (%)	30.25 (45.94)	37.36 (48.38)	−7.11	[−8.57, −5.66]
Poverty rate (% in two lowest wealth quintiles)	74.32 (56.31)	67.50 (53.16)	6.82	[4.16, 9.48]
Age of household head (year)	39.13 (12.69)	40.03 (12.75)	−0.90	[−1.30, −0.51]
Number of children under the age of 5 in a household	2.02 (0.97)	2.27 (1.30)	−0.25	[−0.29, −0.20]
Size of a household	6.42 (2.80)	7.20 (4.00)	−0.78	[−0.92, −0.64]
Education of mother (years)	4.69 (3.67)	3.00 (3.93)	1.69	[1.51, 1.88]
Distance to a road (km)	13.63 (28.48)	2.95 (3.51)	10.68	[9.18, 12.18]
Slope (degree)	2.45 (3.27)	1.46 (2.07)	0.99	[0.77, 1.20]
Population size (individual)	6,539 (17,305)	15,113 (29,037)	−8,574	[−10,167, −6,980]
Distance to a market (km)	43.51 (41.15)	33.39 (27.51)	10.12	[7.45, 12.79]
Land suitable for agriculture (%)	38.95 (48.77)	42.06 (49.37)	−3.11	[−6.90, 0.69]
Community GDP (US$ billion PPP)	1.43 (3.63)	1.56 (2.18)	−0.13	[−0.31, 0.04]
Areas with low livestock density (%)	74.01 (43.86)	23.55 (42.43)	50.47	[47.33, 53.61]
Areas with medium livestock density (%)	18.15 (38.54)	47.83 (49.95)	−29.69	[−32.56, −26.82]
Areas with high livestock density (%)	7.84 (26.88)	28.62 (45.20)	−20.78	[−23.19, −18.38]

*Detailed descriptions and sources of all the variables are in **Supplementary Material Table S2**.*

We perused the reviewed studies. We extracted the raw numbers of stunting prevalence from each reviewed study. We then calculated impacts as the total changes in stunting prevalence (in percent points) brought by the interventions (**Supplementary Material Table S3**). Nevertheless, these studies are not directly comparable to ours for a variety of reasons. They investigate bundles of nutrition (specific or sensitive) interventions instead of one at a time, and study designs and scales differ from ours. Our plot therefore needs to be interpreted with caution.

RESULTS

Partial Identification

The percentage of stunted children among those exposed to forest is 30.25% (SD: 45.94%). The no-assumption lower bound of the effect of exposure to forest on stunting is therefore 30.25–100 = −69.75% points (95% CI [−70.86%; −68.64%]). The no-assumption upper bound is 30.55–0 = 30.25% points (95% CI [29.10%; 31.40%]). Without invoking any assumption then, we can identify the estimate of effects of exposure to forest on stunting to be within the range of [−69.75%; 30.25%].

The positive selection bias posited by the monotone treatment selection assumption implies that counterfactual stunting prevalence for children exposed to forest, had they not been exposed to forests, would be greater than 37.36% (SD: 48.38%), which is the stunting prevalence for children not exposed to forest. Therefore, we can move the upper bound (conservative) estimate of effects of exposure to forest on stunting to 30.25–37.36 = −7.11% points (95% CI [−8.57%; −5.66%]), thus narrowing our range of effect estimate to [−69.75%; −7.11%].

Effects of Forests and Impacts of Different Nutrition Interventions on Child Stunting

Figure 3 plots our conservative (upper bound) estimate of effects of exposure to forest on stunting (−7.11% points) and the estimated impacts of different interventions on stunting reviewed in Hossain et al. (2017). The interventions reviewed in Hossain et al. (2017) include combinations of nutrition education, growth monitoring and promotion, micronutrient supplementation, immunization, health and family planning, access to health facilities, women's empowerment, social safety net, poverty, and food security alleviation, food fortification, integrated management of childhood illness, infant and young child feeding, water, sanitation and hygiene, deworming, child psychological stimulation, community kitchen and garden, telemedicine, feeding practices, and diarrhea and malaria prevention and treatment (**Supplementary Material Table S3**). The impacts of different combinations of these interventions span from a reduction of 30% points to an increase of 6.3% points in stunting prevalence (see **Supplementary Material Table S3**). Our conservative estimate of forest effects falls near the median impact of these other interventions (−10.05% point) (**Figure 3**).

DISCUSSION

Forests address a number of underlying determinants of undernutrition, including the supply of forest food products, income, habitat for pollinators, women's time allocation, prevalence of diarrheal disease, and dietary diversity. As a likely result of these mechanisms, our analysis across 25 low- and middle-income countries suggests that, on average, exposure

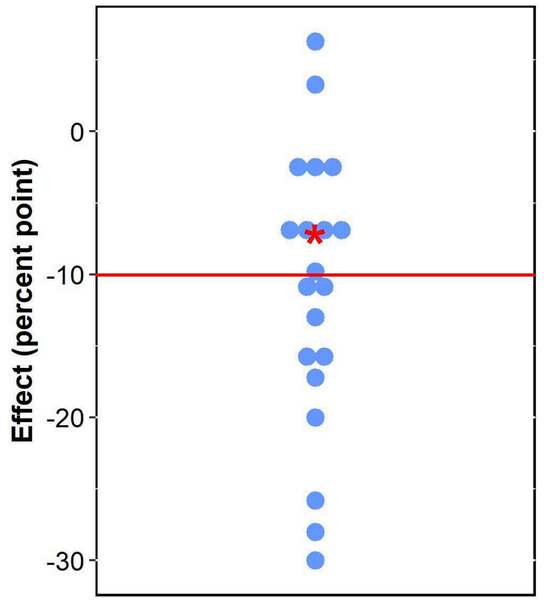

FIGURE 3 | Effects of forests and nutrition interventions on child stunting (red star: conservative effect of forests; blue circles: effects of interventions reviewed in Hossain et al., 2017; red line: median of blue circles).

to forest leads to lower child stunting prevalence compared to non-exposure. Further, the average magnitude of the effect of exposure to forest on child stunting prevalence is at least near the median of the estimated impacts of different nutrition (specific and sensitive) interventions.

Our study confirms the increasingly recognized beneficial effects of forests on quality of human diet (Ickowitz et al., 2014; Galway et al., 2018; Rasolofoson et al., 2018; Rasmussen et al., in press). We moved beyond effects on diet to actual measure of nutritional status (stunting). While we did not identify a precise point estimate of effects of forests on stunting, based on a plausible assumption, we were able to indicate that, on average, forests reduce the prevalence of child stunting and that this average reduction is at least comparable to the impacts of other known nutrition interventions. Our results thus suggest that forest conservation can be a promising nutrition-sensitive intervention.

Different levels of restrictions on use of forest resources by different types of forest conservation interventions can block, to a various degree, some of the mechanisms through which forests affect nutritional status. These restrictions, for example, include limited access to forest food products and non-food products important for income (Poudyal et al., 2018) and therefore may negatively affect nutritional status. On the other hand, forest conservation interventions can also generate benefits through improved ecosystem services, tourism and infrastructure development (Andam et al., 2010). These benefits could lead to improvement in the nutritional status of affected communities (Naidoo et al., 2019). Therefore, the net impact of forest conservation interventions on nutritional status is an empirical question.

A number of studies capture promising actions to enhance the nutrition sensitivity of forest conservation interventions (e.g., volume 13, special issue 3 in International Forest Review; Vira et al., 2015). Some studies suggest that multifunctional landscapes that integrate diversity of agricultural production systems and forests deliver both nutritional and conservation benefits by maintaining key ecosystem services (Sunderland, 2011; Vira et al., 2015). Similar to the cases of other nutrition-sensitive interventions (Ruel and Alderman, 2013), other studies indicate that actions promoting gender equity can increase the nutrition benefits of forest conservation interventions (Sunderland, 2011; Wan et al., 2011). Jamnadass et al. (2011) advocate that actions improving yield, quality, and market access for forest food products can enhance nutrition in rural communities by supplying ample nutritious food products of good quality and raising income. Education, particularly nutrition education, which is shown to enhance the impact of different nutrition-sensitive interventions on nutritional status (Berti et al., 2004; Leroy et al., 2009; Girard et al., 2012), also has great potential to improve the effect of forests on nutrition (Vira et al., 2015; Rasolofoson et al., 2018).

To further determine the merit of recognizing forest conservation among nutrition-sensitive interventions, it helps to examine the key features specified in their definition. One key feature of nutrition-sensitive interventions is that they are often implemented at large scale and can effectively target disadvantaged populations with high rates of undernutrition. Nutrition-sensitive interventions can therefore serve as delivery platforms for nutrition-specific interventions (e.g., nutrition behavior-change communications, food fortification) in efforts to increase their scale, coverage and effectiveness (Ruel and Alderman, 2013). Another key feature is that nutrition-sensitive interventions incorporate specific nutrition goals and actions to achieve these goals (Ruel and Alderman, 2013).

One of the most widespread measures to conserve forests is the designation and management of protected areas [Millennium Ecosystem Assessment (MEA), 2005]. Protected areas currently cover 14.7% of the globe's land area (Jones et al., 2018). Forest conservation can also be advanced through community forest management. Local communities, to a various extent, manage 15.5% of the world's forests [Rights and Resources Initiative (RRI), 2014]. Protected forests and community managed forests are often located in lands with higher elevations, steeper slopes, greater distances to roads and cities, less suitable for agriculture, and high poverty rates (though protected areas may be more remote and less developed relative to community forest areas; Sunderlin et al., 2005; Joppa and Pfaff, 2009; Rasolofoson et al., 2015). Hence, people living in or around protected areas or community managed forests often lack access to sufficient agricultural products, markets, and health services, and thus are likely to have high rates of undernutrition. Using protected areas or community forest management, in different ways, as delivery platforms for nutrition-specific interventions will therefore ensure that these interventions reach large numbers of the world's undernourished communities.

Combination of nutrition-sensitive and nutrition-specific interventions is one of the elements of success of nutrition programs (Hossain et al., 2017; Perez-Escamilla et al., 2018). Therefore, using forest conservation interventions as delivery platforms for nutrition-specific interventions may better deliver nutrition benefits than either of them alone. Examples of such combination could include addition of nutrition behavior-change communications, micronutrient supplementation, food fortification, or disease prevention programs to forest conservation initiatives. Where local communities are involved in forest conservation (e.g., community forest management), the experience and external support they receive in managing their forests can develop social (e.g., community associations, network), human (e.g., skills, expertise), and institutional (e.g., community rules and regulations) assets that constitute a good foundation upon which nutrition interventions can build to reach their goals (Pailler et al., 2015). These community assets are important, given that nutrition interventions are more likely to be successful where there are community-based delivery platforms accompanied with active community engagement (Hossain et al., 2017).

Addition of explicit nutrition goals and actions to nutrition-sensitive interventions help boost their potential to deliver on nutrition outcomes (Ruel and Alderman, 2013). International funding for forest conservation increasingly links conservation and poverty alleviation goals (Miller, 2014). Forest conservation projects therefore increasingly include activities aiming to compensate local communities for benefits forgone due to restrictions on access to forest resources and to improve their livelihoods in order to win their support for conservation (Tabor et al., 2017). Nevertheless, forest conservation initiatives rarely consider health issues (Wan et al., 2011)—including the integration of nutrition goals and actions. Adding nutrition-specific interventions and nutrition goals to forest conservation interventions may be challenging. However, cases of collaboration between health and conservation experts have promoted positive health and conservation outcomes (Wan et al., 2011).

In conclusion, given that forests address a number of underlying determinants of undernutrition, lead to lower child stunting prevalence, and that forest conservation interventions cover large areas and are often implemented where people are vulnerable, policy makers, and public health practitioners might consider forest conservation as a potential nutrition-sensitive intervention. Such interventions might be particularly useful in contexts where implementation of standard nutrition interventions is challenging and where forest conservation interventions might be feasible. This suggests that public health and conservation practitioners should work together to identify, design, and implement projects that help achieve both forest conservation and nutritional goals. Nutrition benefits of forest conservation would not only be of interest to those trying to improve public health, but also those concerned with biodiversity conservation. Co-benefits for nutrition could help to incentivize local communities to participate in conservation, a key factor in determining the success of conservation interventions (Wan et al., 2011). There is unlikely to be a single bundle of nutrition interventions that is effective across all contexts (Hossain et al., 2017) and future research should test different combinations of forest and nutrition interventions in various contexts. Nonetheless, our growing understanding of the potential nutritional benefits of forest conservation is promising.

DATA AVAILABILITY STATEMENT

The primary data on stunting, household, and individual socio-economic characteristics analyzed in this study were obtained from the Demographic and Health Surveys (DHS) Program of the United States Agency for International Development. Information about each DHS survey used in the analysis is included in the **Supplementary Material**. Requests to access these datasets should be directly submitted at https://dhsprogram.com/Data/. The spatial data (forests, road, market, agriculture suitability, livestock density, slope, population, GDP) are available from the corresponding author upon request. Complete metadata is available at https://dx.doi.org/10.6084/m9.figshare.7264658.

AUTHOR CONTRIBUTIONS

All authors contributed to the design of the study and writing of the manuscript. RR organized the database, performed the statistical analysis, and wrote the first draft of the manuscript. All authors read and approved the submitted version.

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REVIEW
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Forest Conservation, Rights, and Diets: Untangling the Issues

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Recent research has highlighted the contributions of forests and tree-based systems to both dietary diversity and nutrition as well as agricultural production in the form of tree-based ecosystem services. Wild foods provide a significant nutritional contribution to the diets of rural dwellers, the majority of whom would be classified as some of the world's poorest. Yet, despite the important human-forest interactions and relative degrees of forest dependency, access to much of the global forest estate is increasingly regulated under the guise of biodiversity conservation. How this restricted access plays out when the "right to food" is a deeply enshrined human right has been deeply contested, particularly with regard to land annexation. This paper outlines the critical issues related to dietary diversity and nutrition in the context of the availability of wild foods juxtaposed with the growing call for the annexation of land for conservation. We suggest that a more integrated and equitable approach to land management that embraces both biodiversity conservation and broader food security and nutrition goals can provide multiple benefits, while mitigating local conflicts. As such, a rights-based approach to conservation and an embracing of broader landscape perspectives are possible strategies to achieve these seemingly conflicting agendas.

Keywords: rights, dietary diversity, biodiversity conservation, food security, nutrition

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INTRODUCTION

For the majority of our history we humans have sustained ourselves by foraging edible plants and hunting animals encountered in grasslands, forests and other wild habitats (Smith et al., 1983). Indeed, our evolutionary development is almost entirely based on a complex system of hunting and gathering that was able to provide a varied and nutritious diet (Crittenden and Schnorr, 2016). Even today, remaining bands of hunter gatherers exhibit greater dietary diversity, nutrition and health than the majority of their more sedentary counterparts (Dounias and Froment, 2006; Reyes-Garcia et al., 2019).

Around 12,000 years ago, agriculture simultaneously emerged in various parts of the world, representing a food system that is very much dominant today. This "agricultural revolution" (Gordon et al., 2016), resulted in the settlement of former forager communities to focus on the production of a small variety of crops and livestock. Sedentary agriculture increased the overall volume of food, yet ultimately led to a more simplified diet. It also resulted in large swathes of wild habitats being permanently transformed into agricultural landscapes, unprecedented population growth and the emergence of cities and society as we now know it (Harari, 2016).

However, despite the overwhelming dominance of contemporary agriculture, people remain dependent on forests in myriad ways (Agrawal et al., 2013). Forests and the resources within them provide a wide array of goods and services for those in their proximity and also to wider society (HLPE, 2017a). Of the estimated 1.6 billion people said to be dependent on forests in some way (FAO, 2014a) many derive much of their dietary diversity and, ultimately, nutrition and broader health, from wild foods (Ickowitz et al., 2014; Powell et al., 2015).

Contemporary agriculture currently focuses on the production of large quantities of a limited number of crops; i.e., the main thrust of global food security remains an emphasis on the production of calories (Vandermeer et al., 2018; Ickowitz et al., 2019). Monocultures of grains and other high-intensity crops require land and much of the historical and contemporary expansion of agriculture has come at the expense of natural habitats, notably forests (Gibbs et al., 2010) with a significant proportion of this forest conversion, primarily for commodity crops, being permanent (Curtis et al., 2018).

We now grow more food than ever before and, in terms of overall production, sufficient food is available to feed the current and projected future populations (Holt-Giménez et al., 2012; HLPE, 2017b). Inequalities in markets, income and distribution, however, mean that vast numbers of the global population are malnourished in a number of ways. This can include undernutrition, being micronutrient deficient, or overweight, and/or obese. The "feast or famine" (Darnton-Hill and Coyne, 1998) dichotomy that characterizes the global food system suggests that we need a dramatic re-thinking of how we feed the global population (HLPE, 2017b; Vandermeer et al., 2018).

In addition to uncertain dietary outcomes, contemporary agriculture has also resulted in significant environmental damage, arguably to the limits of planetary boundaries (Campbell et al., 2017). Aside from being a major driver of deforestation, agriculture accounts for an estimated one third of greenhouse gas emissions through the intensive reliance on fossil fuels (Springmann et al., 2018) and soil erosion from agricultural tillage currently exceeds soil formation (Amundson et al., 2015). Up to 70% of the world's freshwater is appropriated to nourish crops and livestock (Tanentzap et al., 2015) and the loss of ecosystem services through habitat conversion is affecting the resilience of many agricultural systems (Reed et al., 2017). Thus, the global food system is currently characterized by less than adequate nutrition outcomes, compounded by an environment that is being increasingly degraded to support it.

In recent years considerable evidence has emerged that forests and other wild habitats continue to contribute to the dietary diversity and overall nutrition of hundreds of millions of people, particularly those affected by chronic poverty (Ickowitz et al., 2014; HLPE, 2017a). Further evidence also suggests that more complex biodiverse environments are linked with better nutrition outcomes (Sunderland, 2011; Dawson et al., 2019). This is especially important for rural populations with limited market access or who are suffering the effects of poverty and are thus not able to purchase sufficient food to nourish themselves or their families. Wild foods provide people with much needed dietary diversity that includes essential micronutrients (Powell et al., 2015). However, despite high level policy recommendations that better access to wild foods be facilitated, especially those within forests (see Vira et al., 2015; HLPE, 2017a), there is a conflicting movement to protect and isolate vast areas of the global forest estate for the conservation of biodiversity (Sylvester et al., 2016; Perfecto et al., 2019). The International Union for Conservation of Nature (IUCN) estimates that around 15% of the world's land area and 7% of the world's oceans have been designated as protected areas (PAs), many of which restrict access for local communities (IUCN, 2019). This is slightly <10 and 17% for marine and terrestrial protection outlined in the commitments of the Aichi Targets for 2020. PAs and the global hunger statistics may, at first glance, seem like two unrelated global challenges, or even competing interests, but they are in fact inextricably interlinked.

Combating malnutrition is a critical development objective due to the long-term and far-reaching health and socioeconomic implications of malnutrition such as, compromised cognitive development in children (Cawthorn and Hoffman, 2015), childhood stunting (Fa et al., 2015; Nielsen et al., 2018) and increased susceptibility to non-communicable diseases (Popkin, 2001; Vinceti et al., 2013; Savage et al., 2019). Biodiversity conservation, is likewise an important global objective, due to the rapid and ongoing depletion of species and concomitant habitat destruction occurring worldwide, especially in the face of a changing climate (Morales-Hidalgo et al., 2015). In common with malnutrition, biodiversity loss has far-reaching impacts, which negatively impacts both humans and nature. While interest in PA-driven conservation as an effective means to safeguard biodiversity continues to grow, so has the parallel global movement to eradicate hunger and malnutrition. Can these two objectives be achieved in concert?

HOW DO FORESTS PLAY A ROLE IN DIETS AND NUTRITION?

Contribution of Wild Foods to Diets

An important contribution of forests to food security is in the form of the direct provisioning of wild foods such as edible plants, nuts, seeds and wild meat, or bushmeat (Rasmussen et al., 2017). Research has demonstrated that many rural populations that live in or around forested areas rely, to varying degrees, on the harvesting of wild foods to help meet their dietary needs (Broegaard et al., 2016; Rowland et al., 2017). Positive relationships between tree cover and dietary diversity have been identified in Malawi (Johnson et al., 2013; Hall et al., 2019) and also in the vast and diverse archipelago of Indonesia (Ickowitz et al., 2016). However, it is a series of multi-country meta-analyses that provide the most compelling evidence for the positive linkages between forests and diets. Ickowitz et al. (2014), for example, found a positive relationship between tree cover and dietary diversity among the diets of children in 21 African countries. Rasolofson et al. (2018) found the same relationship in their 27-country analysis on the same continent. A global comparative analysis found that 77% of rural households surveyed engaged in wild food collection, highlighting that such harvesting is an integral part of many livelihood strategies, particularly in developing countries (Hickey et al., 2016). Looking at the issue from a different perspective, Galway et al. (2018)

noted that deforestation and the loss of forest cover around dwellings and agricultural fields resulted in poorer dietary outcomes for children in sub-Saharan Africa.

The harvesting of wild foods can contribute to food security by allowing rural dwellers to access these nutritious foods when they may otherwise not have other sources of sustenance (Boedecker et al., 2014). Access to wild foods is also an important part of achieving overall food security as it can help mitigate hardships brought on by internal and external shocks such as droughts, war, illness, and/or failing crops (Pouliot and Treue, 2013; Clements et al., 2014). The collection of wild foods can also bring resilience to traditional agricultural systems by providing a safety net in case of crop failures, pests infestations or crop raiding by animals, a common occurrence in and around PAs (Nyahongo et al., 2009; Pouliot and Treue, 2013; Schulte-Herbrüggen et al., 2013; Wunder et al., 2014; Cawthorn and Hoffman, 2015).

While an agricultural system can provide a family with a few staple food crops and help fulfill the daily caloric requirements of an individual, it doesn't always adequately provide a diverse and nutritious diet when compared to that possible when supplemented with locally available wild foods (Fischer et al., 2017; Nakamura and Hanazaki, 2017). Studies have shown that increased agricultural production has, in some cases, actually led to lower quality diets that are comprised of calorie rich food which lack important micronutrients such as iron, zinc and vitamin B12 (Cawthorn and Hoffman, 2015; Powell et al., 2015; Ickowitz et al., 2016). Thus, harvesting wild foods can provide dietary diversity and help combat micronutrient deficiencies, also known as "hidden hunger" (Ickowitz et al., 2014; Fa et al., 2015; Nielsen et al., 2018). Micronutrient deficiency is an important aspect of malnutrition that can have dire consequences in vulnerable sectors of the population such as young children and can lead to childhood stunting, which has life-long consequences (Temsah et al., 2018).

Although wild foods do not necessarily contribute a large percentage of calories to the diets of rural households, they have been found, in several studies, to contribute to a greater proportion of essential vitamins and minerals (Powell et al., 2015; Asprilla-Perea and Díaz-Puente, 2019). In an assessment of the contribution of natural resources to the nutritional status of the local population in a PA in Gabon, Blaney et al. (2009), found that the consumption of natural resources by children aged 5 to 9, was the best predictor for nutritional status. While natural foods only contributed to 12% of the energy requirements of villagers of the Gamba Complex of Gabon, they contributed an estimated 82% of protein, 36% of Vitamin A and 20% of iron requirements (Blaney et al., 2009).

Hunting for bushmeat has long been a controversial issue due to concerns over the conservation impacts of wildlife depletion but bushmeat hunting is also important in helping rural households to achieve food security (Fa et al., 2009; Nyahongo et al., 2009; Rentsch and Damon, 2013; Golden et al., 2014; Cawthorn and Hoffman, 2015; Reuter et al., 2016; Nielsen et al., 2018). In the Abun region of West Papua, Indonesia, hunting has proved to be an important factor in fighting food insecurity, as wildmeat accounted for 49% of the diets of respondents (Pattiselanno and Lubis, 2014). Bushmeat hunting around the world remains an important source of protein and, more importantly micronutrients, for many rural households and can provide vulnerable populations such as children with important micronutrients (Golden et al., 2011; Van Vliet et al., 2015).

Health

Despite "western" divisions between food, medicine and health, natural resources continue to be an important contributor to health and well-being for many communities (Heywood, 2011). Access to wild foods is therefore important for human health, since nutrition and health are inherently linked. The impact of a loss of medicinal plants and nutritious diets can be seen in many indigenous communities that have undergone nutrition transitions. For example, Indigenous communities in Canada (Binnema and Niemi, 2006; Damman et al., 2008), Argentina (Damman et al., 2008), Sri Lanka (Weerasekara et al., 2018), the Eastern Mediterranean (Heywood, 2011) and Borneo (Dounias et al., 2007) have all undergone nutritional transitions away from their traditional diets. This dietary shift toward a narrower range of foods that are higher in fat, sugar, salt and refined carbohydrates has led to documented increases in the prevalence of non-communicable diseases such as cardiovascular disease and diabetes in the affected populations (Popkin, 2001; Albala et al., 2002; Kuhnlein et al., 2004; Damman et al., 2008; Lourenço et al., 2008; Savage et al., 2019). This has been due to both an increase in a nutritionally poor diets that makes individuals more susceptible to disease and illness, as well as a decrease in access to traditional medicinal plants. In Madagascar, Golden et al. (2011) found that a reduction in wild meat consumption, either by restricted access or wildlife depletion, could lead to a predicted 29% increase in children with anemia and a tripling of anemia in children in the poorest households (Golden et al., 2011). Likewise, in Cameroon, Tata et al. (2019) found incidences of anemia in women was far less prevalent where people had access to leafy forest vegetables. Thus, the benefits of wild foods go beyond the mere consumptive.

CONSERVATION, RIGHTS, AND ACCESS

Biodiversity conservation, as we relate to it, is a relatively modern construct. The creation of designated PAs is ultimately rooted in the western perspective of nature as untouched, uninhabited and unaltered (Neumann, 2002). This notion of pristine nature is, however, fails to recognize how people have been altering their landscapes for centuries and these altered landscapes have thus been classified as "natural" and "wild" (Shafer, 2015; Massé, 2016; Anaya and Espírito-Santo, 2018). Consequently, the dominant approach to conservation throughout the twentieth century was through the establishment of PAs from which people were essentially excluded. This model of conservation came to dominate twentieth century thinking, drawing primarily from the well-known North American networks of National Parks (Adams, 2004; Hutton et al., 2005).

However, there is often an inherent asymmetry in the costs and benefits of biodiversity conservation through protectionism, particularly in developing countries. While the multiple benefits of biodiversity conservation accrue at the national and global levels, the costs of PAs are often borne by local communities (Arjunan et al., 2006) particularly in terms of loss of access to resources. In this regard, many conservation initiatives around the world have had a long history of decoupling food security from biodiversity conservation by failing to understand the important role that natural resources play in the healthy and nutritious diets of rural populations (Powell et al., 2015; Sylvester et al., 2016) and conversely neglecting the important stewardship role that indigenous people play in natural resource management (Garnett et al., 2018). PAs can contribute to food insecurity through a variety of pathways such as a loss of direct access to the harvesting of wild foods (Nakamura and Hanazaki, 2017), loss of livestock due to predation by wildlife (Banerjee, 2012; Givá and Raitio, 2017), loss of access to water bodies used for irrigation or drinking water (Adhikari et al., 2009; N'Danikou et al., 2017), loss of fuelwood for cooking (Banerjee, 2012), loss of traditional knowledge (Turner and Turner, 2008; Desmet, 2016), and loss of access to markets and increased food prices due to tourism (Rosendo et al., 2011; Bennett and Dearden, 2014).

The proximity of local populations living in and around PAs has caused tensions in many parts of the world, which has, in some cases, resulted in conflict (West et al., 2006). The number of documented abuses of power and human rights due to the establishment, management and policing of PAs has been so prolific that the seriousness of the problem was recognized at an international level as early as 1982 at the Third World Park Congress (WPC), but a new agreement highlighting the problem was not reached until the 5th WPC in 2003 (IUCN, 2005). At this event, the Durban Accord was established to represent a shift in thinking that recognized the need to involve indigenous communities and address their needs in the context of PA establishment and management (Adams and Hutton, 2007). This also led to the launching in 2009 of the Conservation Initiative on Human rights, established by the largest conservation organizations to integrate and protect human rights in the design and implementation of conservation.

However, despite these movements toward respecting basic rights in contested landscapes, mounting evidence of human rights violations with regards to food access in PAs, land annexation for conservation has continued to grow along with enforcement to restrict access for harvesting wild resources (Sylvester et al., 2016; Newing and Perram, 2019). A case in point are the recent accusations leveled at the World Wildlife Fund who are being accused of significant human rights abuses in terms of over-zealous enforcement and restricting access to lands formerly utilized as a source of forest products[1].

Traditional preservationist approaches to conserving our natural heritage have made way in the past 30 years to a more participatory and people-centered approach. Indeed, in the past decade or so, the majority of newly-created PAs fall within the lower end of the IUCN categories that allow for some level of subsistence use and management by local communities. Naidoo et al. (2019), in a comprehensive global analysis of the economic impacts of PA's show that where access and rights are respected, human well-being can be positively impacted by conservation implementation. Thus, sustainable use of selected resources, particularly wild foods, would suggest that there is greater scope for the integration of human use, and management, in many PAs.

However, while the conservation community has made increasing strives in recent decades to move away from what is described as "fortress conservation" approaches (Brockington, 2002) and toward community-based natural resource management that take into account local concerns and livelihoods, there remain persistent concerns with regard to a general disregard to effectively implementing "rights-based approaches" to conservation (Campese et al., 2009—see also **Box 1**). This is particularly concerning given recent calls to increase the area for biodiversity conservation to 50% of terrestrial land, or "half-Earth" (Wilson, 2016). The feasibility of such a proposition has been questioned given current human needs (Buscher et al., 2017), particularly for global food security (Mehrabi et al., 2018). However, the debate around such a proposition characterizes the dichotomy between the twin imperative of conserving global biodiversity while achieving a just, equitable and healthy food system. Despite the right to food[2] being enshrined in the human rights commitments of many nations around the world (see **Box 2**), these rights seem to not be factored in to the debate surrounding the need to achieve commitments toward biodiversity conservation.

RECONCILING RIGHTS AND ACCESS TO ENSURE DIETARY DIVERSITY

While biodiversity conservation is an important goal in a time when climate change and biodiversity loss are both real threats to human societies, clearly this must only take place when the underlying power relations that displace, restrict, enforce and result in significant social inequities, are addressed (Newing and Perram, 2019). Despite the majority of recently establish PAs falling within IUCN categories that allow multiple use, rights, tenure and access remain issues of contention within the traditional biodiversity conservation approach (Mollett and Kepe, 2018). As nation states attempt to achieve commitments to increase the area of land committed to conservation due to their global commitments, how can this be reconciled with achieving, or maintaining, food security and nutrition goals?

As stated previously, many PAs around the world have resulted in the loss of land rights and food access for local populations which has in turn negatively impacted the diets and nutrition of nearby communities. In order to prevent some of these impacts it is important to understand how management strategies can lead to food insecurity. Enlisting new initiatives to alleviate food insecurity and biodiversity loss will thus require the involvement of multiple disciplines to contribute innovative ways forward (Brockington et al., 2006; Timko and Satterfield, 2008).

[1] https://www.buzzfeednews.com/article/tomwarren/wwf-world-wide-fund-nature-parks-torture-death.

[2] "*The right to adequate food is realized when every man, woman and child, alone or in community with others, have physical and economic access at all times to adequate food or means for its procurement*." CESCR General Comment No. 12: The Right to Adequate Food (Art. 11).

BOX 1 | Principles of the conservation initiative on human rights.

RESPECT HUMAN RIGHTS Respect internationally proclaimed human rights and make sure that we do not contribute to infringements of human rights while pursuing our mission.

PROMOTE HUMAN RIGHTS WITHIN CONSERVATION PROGRAMS Support and promote the protection and realization of human rights within the scope of our conservation programs.

PROTECT THE VULNERABLE Make special efforts to avoid harm to those who are vulnerable to infringements of their rights and to support the protection and fulfillment of their rights within the scope of our conservation programs.

ENCOURAGE GOOD GOVERNANCE Support the improvement of governance systems that can secure the rights of indigenous peoples and local communities in the context of our work on conservation and sustainable natural resource use, including elements such as legal, policy and institutional frameworks, and procedures for equitable participation and accountability.

Source: www.thecihr.org

BOX 2 | The right to food: Selected policies and legislative framework related to food security.

1941: U.S. President Franklin D. Roosevelt includes right to food one of the freedoms: "*The freedom from want*."

1948: Universal Declaration of Human Rights recognizes the right to food as part of the right to an adequate standard of living.

1966: The International Covenant on Economic, Social and Cultural Rights, reiterates the Universal Declaration of Human Rights with regard to be free from hunger.

1974: At the inaugural World Food Conference, 135 participating countries issued the Universal Declaration on the Eradication of Hunger and malnutrition which declared that "*[e]very man, woman and child has the inalienable right to be free from hunger and malnutrition in order to develop fully and maintain their physical and mental faculties*" (UN General Assembly, 1975, art. 1).

1975: The IUCN passed the Kinshasa Resolution on the protection of the "traditional ways of life" and called on governments to halt the displacement and relocation of people due to PA establishment (Adams and Hutton, 2007).

1996: The World Food Summit resulted in highlighting food security as a new global development goal. During this summit, food security was defined as "*exists[ing] when all people, at all times, have physical and economic access to sufficient, safe and nutritious food that meets their dietary needs and food preferences for an active and healthy life*" (World Food Summit, 1996).

2003: Durban Action Plan, outcome 5 "*The rights of indigenous peoples, including mobile indigenous peoples, and local communities are secured in relation to natural resources and biodiversity conservation*". http://danadeclaration.org/pdf/durbanactioneng.pdf

2004: The Convention on Biological Diversity called for the recognition of "*the economic and socio-cultural costs and impacts arising from the establishment and maintenance of protected areas, particularly for indigenous and local communities, and (an adjustment of) policies to ensure that such costs and impacts—including the cost of livelihood opportunities forgone—are equitably compensated*". COP 8 Decision VIII/23: https://www.cbd.int/decision/cop/?id=11037

2007: Establishment of the United Nations Declaration on the Rights of Indigenous Peoples (UNDRIP) affirmed the rights, survival, dignity and well-being of Indigenous people as well as safeguard the individual and collective rights of Indigenous people that may not be addressed by other human rights charters.

2009: Adoption of the Optional Protocol to the International Covenant on Economic, Social and Cultural Rights, making the right to food justiciable at the international level.

2012: UN Zero Hunger Challenge, which calls for sustainable food systems, an end to rural poverty, adaptation of all food systems to eliminate loss or waste of food, increase access to adequate food and healthy diets for all people all year round and finally, for an end to malnutrition in all its forms. https://www.un.org/zerohunger

2015: Sustainable Development Goals: SDG2, Target 1: "*End the global hunger crisis and ensure all people, especially the poor, have access to sufficient and nutritious food*". https://www.mdgmonitor.org/sdg2-end-hunger-achieve-food-security-and-improved-nutrition-and-promote-sustainable-agriculture

A move toward an increased recognition of synergies, rather than the trade-offs, between food security and biodiversity conservation presents an opportunity for the emergence of new conservation frameworks that build on rights-based approaches, food sovereignty principles, and participatory-conservation to rethink how PA enactment and enforcement is approached (Perfecto et al., 2019). As Broegaard et al. (2016) point out, within a landscape, nutritional outcomes are determined as much by access to resources that comprise rural diets as much as food production. Yet the inalienable right to be free from hunger is still denied for many rural populations that live within or adjacent to PA's, where there remains a strong emphasis on enforcement and restricted access.

While biodiversity conservation remains embedded in the paradigm of PAs, it is known that much of the world's biodiversity actually occurs in areas not under formal protection, but often in complex multi-functional landscape mosaics (Cox and Underwood, 2011; Gray et al., 2016). Such landscapes are often characterized by remnants of trees, either retained, planted or regenerated, intermixed with small-scale agricultural production systems. Indeed, it is estimated that between 35% (Ricciardi et al., 2018) and 70–80% (FAO, 2014b) of the world's food is actually grown by these smallholders who often manage such systems for a whole suite of products and ecosystem service benefits such as pollination, climate regulation, nutrient cycling etc. (Padoch and Sunderland, 2014; Baudron et al., 2019). There is increasing evidence of the myriad ways that forests and trees sustain agriculture when there is integration, rather than segregation, of function at the landscape scale (Reed et al., 2017). Such complex systems are also more resilient to both economic and environmental shocks (Wunder et al., 2014). This matrix provides a suite of agricultural products but also facilitates access to wild foods and other resources.

Interestingly, in recent years, research has begun to question the very premise of conservation in terms of the pristine nature of many of our wild places. It is currently emerging that vast tracts of the forested areas we have assumed to be thought of as pristine nature are in fact artifacts of millennia of human use and intervention. These include large areas of the Amazon (Levis et al., 2017; Maezumi et al., 2018) and Congo Basins (Osilisly et al., 2013; Lupo et al., 2015). In both forest blocks, extensive historical evidence has been found showing large settlements, anthropogenic burning, previous plant domestication and distribution, mining, agroforestry and crop production were prevalent.

This, of course, does not suggest that such forest formations are not worth conserving. Clearly, because of their considerable, and often unique, levels of biodiversity, as well as their carbon value, essential for climate change mitigation, they are. However, it should be recognized that, in many instances, their current manifestation is due to human influence. Given that indigenous peoples currently manage or have tenure rights over at least a quarter of the world's land surface which intersects with around 40% of terrestrial PAs and ecologically intact landscapes (Garnett et al., 2018), surely there is scope for these "gatekeepers" (Mackelworth and Carić, 2009) to be increasingly integrated into the establishment and management of conservation initiatives whereby both biodiversity conservation and livelihood goals are achieved in concert.

For example, in a meta-analysis of 55 PA's in developing countries, Andrade and Rhodes (2012) found the variable that most influenced the level of compliance with PA policies was the level of involvement of local communities in decision-making processes. Such evidence therefore gives further credence to the call for rights-based approaches which recognizes and respects the rights of local communities. Chhatre and Agrawal (2009) likewise found that higher levels of involvement and decision-making power of local communities led to more favorable conservation outcomes. In addition, the findings of Naidoo et al. (2019) that lower levels of protection within PA's, respecting usufruct rights of local communities, notably access to wild food resources, can lead to positive livelihood and conservation outcomes. A further meta-analysis of 165 PAs found that those that were associated with a positive socioeconomic outcome were more likely to also report positive conservation outcomes and thus demonstrated that conservation and food security goals are not antagonistic (Oldekop et al., 2016). Therein lies an opportunity to rethink how PAs are enacted and managed in order to support both biodiversity conservation and food security.

Thus, rights-based approaches to conservation will be one key instrument in moving toward more salient conservation policies that integrate the fundamental "right to food" by helping to identify rights-holders and duty-bearers to better inform PA management (Young et al., 2004; He and Cliquet, 2014; Newing and Perram, 2019). Adopting such an approach to conservation will present its own set of challenges, such as funding, lack of expertise and/or government capacity and competing rights, but it is a necessary step forward that can help to increase both conservation and food security (He and Cliquet, 2014; Kraak, 2018). In some cases, a rights-based approach will require the dissemination of power within PAs in favor of more egalitarian, bottom-up approaches such as community-based conservation projects and livelihoods-based conservation (Campese et al., 2009).

Rights-based approaches can also increase the resilience of both humans and nature by supporting both social and environmental justice through collaboration and shared responsibility (Walsh-Dilley et al., 2016). Using a rights-based approach to empower local communities to make their own management decisions around harvesting, logging and other resources practices can actually increase conservation outcomes as an increase in rights and responsibilities decreases unsustainable harvesting practices (Nielsen et al., 2018).

In a recent synthesis report on "Sustainable Forestry for Food Security and Nutrition" commissioned by the Committee of World Food Security (CFS) a series of recommendations were proposed and adopted by the CFS (HLPE, 2017a). This potentially represents the greatest leverage to include forests and trees onto the global food security agenda, even as a significant proportion of the forest estate is being increasingly allocated for conservation. We now have a much deeper understanding of the mechanisms as to how sustainable forest management contributes to food security and nutrition, but this contribution could be increased significantly through priority actions to:

1. Provide secure land and forest tenure and equitable access to resources.
2. Recognize and integrate the contribution of forests to food security and nutrition in forest policies.
3. Improve the alignment of food security and nutrition policies across the agriculture, forestry, livestock, fisheries, energy, mining and other relevant sectors.
4. Increase access by small forest and farmholders and their organizations to business skills, training, credit, technology, extension services and insurance.
5. Integrate gender equality in the formation, implementation and evaluation of relevant forest policies, and in investment strategies.
6. Strengthen the collection and timely dissemination of data relevant to policy-making on the contribution of forests and trees to food security and nutrition (Source: HLPE, 2017a).

It is evidently clear that a more holistic approach to conservation, forests management and food security can contribute to more successful outcomes for each sector, rather than the current siloed and detached focus on them in their singularity. Managing landscapes in an integrated manner for such multiple benefits is but one way forward (Sayer et al., 2013; Reed et al., 2016), and it has been suggested that various and diverse landscape configurations can provide multiple benefits for both conservation and agriculture (Rasmussen et al., 2019). Of course, global food security cannot be achieved by such an approach alone, but with the current emphasis on calories and monocultures the broader recognition of natural systems in the provision of diverse and nutritious diets is very timely. Integrating and respecting rights into our global conservation network is also long overdue, perhaps the implementation of a more "convivial conservation" as outlined by Büscher and Fletcher

(2019: 283), a "post-capitalist approach to conservation and promotes radical equity, structural transformation and environmental justice. . . .to create a more equal and sustainable world."

IN CONCLUSION

As the contribution of forests and tree-based systems continues to be recognized so does the opportunity to reconcile conservation in PAs with the rights to food in these spaces. With the increasingly growing demand to conserve more land and seascapes and reach the goals set out by global treaties it is now more important than ever to move forward with more inclusive management programs that do not jeopardize human livelihoods. As the amount of land that is set aside for the creation or expansion of PAs continues to grow so does the opportunity to recognize and rework broken management schemes that do not accurately reflect the social costs of conservation, the burden of which is most heavily felt by the poor and disfranchised parts of the population. While the recognition of rights-based approaches to conservation and rights to food will help alleviate food insecurity and malnutrition, it is but one strategy to ensure a more sustainable and equitable future.

Likewise, increasingly loud calls for a more ecologically friendly agriculture suggest there is developing interest in promoting long-term sustainability in the agricultural sector over production alone (DeClerck et al., 2011; Campanhola and Pandey, 2019; Ickowitz et al., 2019). Extensive evidence is emerging that breaking down the barriers between agriculture and forest conservation at the landscape scale could have significant potential both conserve biodiversity and ensure a more sustainable agricultural production; indeed, taking the "whole earth" approach advocated by (Büscher and Fletcher, 2019). Such an approach could also have significant long-term impacts on the nutrition and health of millions, if not billions of people (Gordon et al., 2016; Campanhola and Pandey, 2019) while ensuring the rights to access to such healthy and nutritious food in wild, and often protected, habitats are increasingly uncontested.

AUTHOR CONTRIBUTIONS

TS conceived of and secured funding for the larger body of work that provided the background evidence for this research. He designed the basic structure of paper and revised the first draft post-peer review. WV assisted with a broader literature review and drafted an early version of the paper.

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Deforestation and Household- and Individual-Level Double Burden of Malnutrition in Sub-saharan Africa

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Introduction: Although forests and forest-based ecosystems have been shown to influence health and sustainable diets, there is limited evidence on how deforestation affects the current nutrition transition and the double burden of malnutrition. We examined the relationship between deforestation and the individual- and household-level double burden of malnutrition in 15 countries in Sub-Saharan Africa.

Materials and methods: We combined data from geolocated Demographic and Health Surveys and the Global Forest Change dataset. We defined household-level double burden of malnutrition as the co-occurrence of an overweight woman of childbearing age (WCBA) and a stunted pre-school child (PSC) within the same household. We defined individual-level double burden in two ways: (1) as the co-occurrence of overweight and anemia within an individual WCBA, and (2) as the co-occurrence of overweight and stunting within a PSC. We used logistic regression analysis to examine the association between forest cover loss and these three measures after adjusting for potential confounders. We also assessed the mechanisms linking forest cover loss and nutritional status, such as livestock ownership and access to clean water.

Results: In our sample, the prevalence rates of the three measures of the double burden were: overweight and anemic WCBA: 8.4%, overweight WCBA and stunted PSC: 6.9%, overweight and stunted PSC: 2.7%. After adjusting for the confounders as well as country fixed effects and the month of the survey, forest cover loss was marginally associated with a higher odds of an overweight WCBA and stunted PSC [odds ratio (95% CI): 4.80 (0.82, 28.25)]. We found no association between forest cover loss and odds of an overweight and stunted PSC [odds ratio (95% CI): 2.47 (0.80, 7.60)] or the odds of an anemic and overweight WCBA [odds ratio (95% CI): 0.71 (0.15, 3.32)].

Discussion: Deforestation does not seem to be an important driver of the double burden of malnutrition in SSA. However, deforestation influences several intermediate factors which, in turn, may influence the double burden. The overall weak association between forest cover loss and double burden measures mask significant heterogeneity across regions within SSA. Future research should unpack the mechanisms behind these regional differences.

Keywords: deforestation, double burden, Sub-Saharan Africa, malnutrition, food systems

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INTRODUCTION

Deforestation is occurring globally at an alarming rate (Hosonuma et al., 2012; Hansen et al., 2013), particularly in the tropics (Lindquist et al., 2012; Hansen et al., 2013), raising concerns about potential impacts on biodiversity, ecosystems services, and human health. Deforestation has been shown to be associated with a number of conditions, such as acute respiratory infection (Pienkowski et al., 2017), malaria (Bauch et al., 2015; Austin et al., 2017; Berazneva and Byker, 2017), and diarrheal disease (Johnson et al., 2013; Berazneva and Byker, 2017).

Simultaneously, many low- and middle-income countries are experiencing a rapid rise in the prevalence of the double burden of malnutrition (Dieffenbach and Stein, 2012; Oddo et al., 2012; Roemling and Qaim, 2013; Wojcicki, 2014; Berazneva and Byker, 2017), wherein obesity and diet-related chronic disease commonly co-occur with conditions of undernutrition (e.g., child growth stunting, micronutrient deficiencies, and associated anemia). An emerging body of literature has examined this double burden as well as its determinants (Lee et al., 2010, 2012; Oddo et al., 2012; Bassete et al., 2014; Aitsi-Selmi, 2015; Kosaka and Umezaki, 2017). The most commonly assessed determinants include socio-economic determinants, such as urban/rural residence, income, and maternal/household-head education level (Kosaka and Umezaki, 2017). Empirical evidence on the role of environmental determinants, such as forest cover loss, is lacking. Given the pace, urgency, and scale of global environmental changes, there is a growing need to address this knowledge gap.

Forests and forest-based ecosystem services have also been shown to be important drivers of healthy and sustainable diets (Dounias and Froment, 2011; Golden et al., 2011; Vinceti et al., 2013; Brown et al., 2014; Ickowitz et al., 2014, 2016; Powell et al., 2015; Vira et al., 2015; Pienkowski et al., 2017; Rowland et al., 2017; Galway et al., 2018; Rasolofoson et al., 2018), suggesting that deforestation has the potential to affect the double burden of malnutrition. To our knowledge, two prior studies (Rasolofoson et al., 2018 and Galway et al., 2018) have provided frameworks for understanding the potential mechanisms linking forests to diets. Broadly speaking, the mechanisms include changes in the availability of forest foods, pollination, the availability of non-forest products, mother's time for food preparation and child care activities, and agricultural techniques (Rasolofoson et al., 2018). In Galway et al. (2018), we provide a finer breakdown of these mechanisms with supporting evidence from the literature.

The mechanisms through which forest cover loss could affect the double burden of malnutrition are likely similar, with changes in diet as an important intermediate factor. Therefore, in **Figure 1**, we adapt the framework from Galway et al. (2018) with the double burden of malnutrition as the outcome. The shaded areas of the framework are taken from Galway et al. (2018), while the darker boxes and arrows represent additional factors linking forest loss to the nutritional double burden (In **Figure 1**, the factors we evaluate in the current study are highlighted in bold). Two of the additional factors need elaboration. First, changes in the amount of time spent gathering firewood or changes in agricultural practices can affect women's calorie expenditure, which in turn can affect anthropometric status, in particular weight. Indeed, existing evidence shows that women tend to get less physical activity than men in an urban environment (Shrimpton and Rokx, 2012), which may lead to an increased risk of overweight. Second, individuals' access to clean water and sanitation may change—either because they move in response to deforestation or their existing water source gets contaminated. Such changes in access to water and sanitation can affect children by making them more vulnerable to illnesses, such as diarrheal

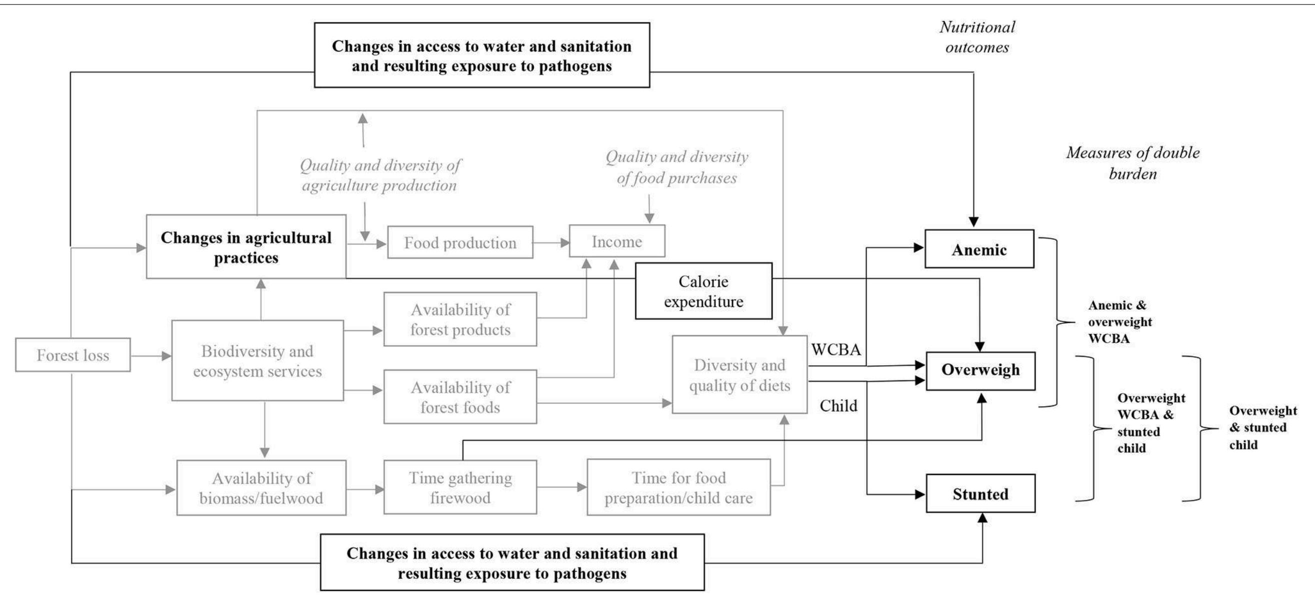

FIGURE 1 | Conceptual framework illustrating hypothesized links between forest loss and double burden of malnutrition*. The shaded part of the framework is adapted from Galway et al. (2018) with the authors' permission. The outcomes we evaluate in the current study are highlighted in bold.

diseases and infections, which may lead to an increased risk of anemia. If the exposure to pathogens is persistent, children may also be at higher risk of stunting.

Deforestation may also affect different population segments differently, even within the same household. For example, mothers might be impacted differently than young children. If the primary reason for deforestation is to make space for cultivation of crops, mothers, now having to work in the field, may end up having less time for breastfeeding and other caregiving activities, potentially worsening children's nutritional status. Their own nutritional status may improve or worsen, depending on how their work burden changes. By altering diets, deforestation may also affect the multiple measures of malnutrition differently even for the same child. For example, if more iron-rich crops are grown, the likelihood of being anemic may fall, but if mother's caregiving is adversely affected—for example, through reduced frequency and duration of breastfeeding—a child may be at increased risk of stunting or wasting.

The overall effect of deforestation on the nutritional double burden, both at the household level and at the individual level remains an unanswered and important empirical question. Against this backdrop, the goal of this study was to examine the association between deforestation and the household- and individual-level double burden of malnutrition. We conducted this study in the context of Sub-Saharan Africa (SSA), where the rate of deforestation is twice the world average (UNEP, 2008; FAO, 2010; Gibson et al., 2011; D'Annunzio et al., 2015; Austin et al., 2017). The nutritional double burden is also prevalent throughout SSA and is certain to increase (Zeba et al., 2012; Wojcicki, 2014; Jones et al., 2016).

MATERIALS AND METHODS

Data

We used geolocated data collected by the USAID-funded Demographic and Health Survey (DHS) program. The DHS program collects nationally representative health, fertility and nutrition data from more than 90 countries using a two-stage cluster sampling procedure (Measure DHS/ICF International, 2012). We compiled individual and children's recode DHS data files from all countries in SSA for which the standard DHS survey was conducted between 2012 and 2016 and for which geolocated data were available. In total, datasets from 15 countries met these criteria and were included in our analyses (**Table 1**). We included all non-pregnant women of childbearing age (WCBA) from 15 to 49 years of age who were interviewed in the DHS as well as pre-school children (PSC) aged 12–59 months. In our analysis of PSC, we included all children from a mother except in few instances where two children from the same mother were coded as having the same birth index.

We obtained data on forest cover loss (a proxy of deforestation) from the publicly available Global Forest Change dataset developed (Hansen et al., 2013). The Global Forest Change dataset, developed by Hansen and colleagues, measures forest cover loss worldwide (excluding Antarctica and the Arctic) at a spatial resolution of ∼30 m (Hansen et al., 2013). Hansen et al. (2013) define forest loss as stand-replacement disturbance

TABLE 1 | Demographic and Health Survey data sets used.

Country	Year	*n* (children)	*n* (mothers[a])	*n* (children and mothers[b])	*n* (clusters)
Democratic Republic of the Congo	2013–14	5,710	4,136	5,651	490
Gabon	2012	2,389	2,128	2,374	305
Zambia	2013–14	8,910	–	8,881	709
Kenya	2014	6,960	–	6,909	1,246
Rwanda	2014–15	2,758	2,534	2,753	448
Benin	2011–12	6,089	2,299	6,031	730
Cote D'Ivoire	2011–12	2,287	2,009	2,243	317
Ghana	2014	2,052	1,787	2,047	346
Guinea	2012	2,328	2,136	2,316	287
Liberia	2013	2,355	–	2,345	314
Mali	2012–13	3,513	2,789	3,468	400
Nigeria	2013	18,585	–	18,461	887
Senegal	2012–13	4,352	–	–	199
Sierra Leone	2013	3,186	3,346	3,143	417
Togo	2013–14	2,467	2,121	2,460	302
Total		73,941	25,285	69,082	6,615

[a] *Countries where hemoglobin levels were not collected as part of DHS were not included in the calculation of the double burden of malnutrition among mothers. These countries include: Zambia, Kenya, Liberia, Nigeria, and Senegal.*

[b] *Maternal BMI was not available for Senegal. Therefore, we excluded Senegal from the analysis for the calculation of double burden of malnutrition at the household level.*

or the complete removal of tree cover canopy at the pixel scale while trees are defined as vegetation taller than 5 m in height. Following our previous study (Galway et al., 2018), we used version 1.0 of the dataset which measured forest cover loss between 2000 and 2012 using a time-series analysis of more than 600,000 multispectral satellite images from Landsat 7. We used version 1.0 rather than the updated version of the dataset to ensure that our measurement of forest lost preceded the DHS survey data collection. We also used data on percent tree cover for the year 2000 from the Global Forest Change dataset (Hansen et al., 2013). Information on road location was obtained from the Global Roads Open Access Data Set (Center for International Earth Science Information Network, 2013). Finally, we used the Global Aridity Index from the Consultative Group on International Agricultural Research (CGIAR, 2009) to measure climate across DHS clusters. We combined data from the DHS and the Global Forest Change dataset at the cluster-level using a geographic information system and ArcGIS software (ESRI, 2011).

Outcome Variables

We defined overweight or obesity in WCBA using standard BMI (in kg/m^2) cutoffs ($\geq$25). Anemia in WCBA was defined as a hemoglobin concentration <120 g/L, the recommended cutoff for non-pregnant women aged $\geq$15 years. We considered children with a HAZ <2 SD below the mean according to the WHO Child Growth Standards to be stunted and those with WHZ >2 SD above the mean to be overweight.

We used three measures of double burden: two at the individual level and one at the household level. We defined individual-level double burden in two ways: (1) as the co-occurrence of overweight and anemia within an individual WCBA, and (2) as the co-occurrence of overweight and stunting within a PSC. We defined household-level double burden of malnutrition as the co-occurrence of an overweight WCBA and a stunted PSC within the same household.

In an effort to understand potential mechanisms linking deforestation to the nutritional double burden, we also examined the relationship between forest loss and five intermediate outcomes: (i) ownership of livestock, (ii) ownership of agricultural land, (iii) access to improved water, (iv) access to improved sanitation, and (v) occurrence of diarrhea among the PSCs.

As discussed in the results section below, for the overall sample, we found only a weak association between forest cover loss and the co-occurrence of overweight and stunting within a PSC and no association between forest cover loss and the two other measures of double burden. This contradicted with the negative associations we found between forest cover loss and many of the intermediate outcomes. Therefore, we also examined the association between forest cover loss and individual components of nutritional status; in these analyses, the outcome variables were the components used to construct the double-burden measures, specifically occurrences of: (i) overweight WCBA, (ii) anemic WCBA, (iii) overweight PSC, and (iv) stunted PSC.

Independent Variable

The primary independent variable we examined was deforestation measured as forest cover loss between 2000 and 2012. Though the construction of this variable has been previously described (Galway et al., 2018), briefly, the Global Forest Change dataset defines forest cover loss as stand-replacement disturbance or a change from a forest to non-forest state during the 2000–2012 period in a 30 m by 30 m grid cell. In the dataset, each 30 m by 30 m pixel is coded as "1" for forest cover loss or "0" for no loss of forest cover. The Global Forest Change data are downloadable as tiff panels; we downloaded those panels covering the spatial extent of our 15 study countries in SSA (Hansen et al., 2013). The georeferenced DHS cluster locations are randomly displaced in order to protect the confidentiality of the survey respondents (Warren et al., 2016). The large majority (99%) of the locations are displaced by 0–5 km, with a remaining 1% of rural clusters displaced to a maximum of 10 km (Measure DHS/ICF International, 2012). Using ArcGIS software (ESRI, 2011), we aggregated the original 30 m pixels to 5 km pixels to account for the displacement. We then spatially joined the aggregated pixels to the georeferenced DHS data to extract the percentage of forest cover loss in the 5 km area surrounding each DHS cluster.

Covariates

We included several covariates in analyses to adjust for potential confounding of the relationship between forest loss and the double burden of malnutrition. We selected these covariates based on previous evidence of the determinants of nutritional status (Guldan et al., 1993; Variyam et al., 1999; Vereecken et al., 2004; Black et al., 2013). The child-level covariates included sex, episode of diarrhea during the 2 weeks preceding the survey, and age in months. Other covariates included the highest attained education level of the child's mother, household access to an improved source of water and sanitation (WHO/UNICEF Joint Monitoring Programme for Water Supply and Sanitation, 2012), and a measure of household wealth available from the DHS. DHS creates the wealth index from several items related to household assets (e.g., radio, refrigerator), housing characteristics (e.g., type of flooring), and utilities and infrastructure (e.g., number of persons sleeping per room) (Rutstein and Johnson, 2004). Households are divided into quintiles based on the index. In a subset of the analyses, we also include household's ownership of agricultural land and ownership of livestock as covariates. We show results from both sets of regressions (i.e., those with and without controlling for agricultural land and livestock ownership) in an attempt to shed light on changes in the double burden measures originating from changes in deforestation independent of agricultural land use (to the extent that changes in land use are reflected in the ownership of agricultural land and livestock).

At the cluster level, we adjusted for whether the cluster was urban or rural based on DHS definitions and distance of the DHS cluster to the nearest road. We calculated the Euclidian distance of the DHS cluster centroid to the nearest road using road location data from the Global Roads Open Access Data Set (Center for International Earth Science Information Network, 2013). Climatic differences may in part explain geographic variation in mother and child nutrition outcomes (Dulal et al., 2017). To adjust for such confounding, we used the Global Aridity Index (CGIAR, 2009), aggregated to the 5 km level and spatially linked to each cluster. We also adjusted for baseline forest cover using percent tree cover in year 2000 (Hansen et al., 2013). As with the forest cover loss data, we aggregated the original 30 m pixels to 5 km pixels and spatially joined the data to each DHS cluster. In an attempt to account for nutritional differences that may have been driven by factors specific to a country (e.g., country's gross domestic product and government's policies on nutrition and health) or DHS cluster, and by seasonal variation in food availability (Abizari et al., 2017), we adjusted for country fixed effects, cluster-level random effects, and the month of the DHS survey in our analyses.

Statistical Analysis

We calculated means and proportions for the outcome and independent variables, as well as for key household- and child-level characteristics. Given the binary nature of the outcome variables, we used logistic regression analysis to examine the association between forest cover loss and the measures of the double burden of malnutrition, after adjusting for potential individual-, household- and community-level confounders. Specifically, we estimated the following equation:

$$Y_{ijk} = \alpha + \beta_1 Forest\ loss_k + \delta\ X_{ijk} + \ \eta\ + \ \theta + \ \omega + \ \varepsilon \qquad (1)$$

where Y_{ijk} is the relevant double burden of malnutrition measure for either child, mother, or child-mother pair i in household j living in DHS cluster k. Forest cover loss, the independent variable, varied by the DHS cluster. We reported odds ratios. In the equation, the odds ratio β_1 reflects the association between forest cover loss and the dependent variable. **X** represents child-, household- and cluster-level characteristics mentioned above that potentially influence the double burden of nutrition. η represents the country of the child, θ represents the cluster random effect, while ω represents the month of the survey.

In all models, we clustered the standard errors at the level of the DHS enumeration cluster to account for arbitrary correlation between observations within a cluster. Recall that the variation in forest cover loss is at the DHS cluster level. Clustering standard errors at the level of DHS sampling units also accounts for intra-household correlations among those households with multiple children or mothers in the sample.

The statistical significance of associations is reported at the $P < 0.1$, $P < 0.05$, and $P < 0.01$ levels. Given current debates on the arbitrary nature of these cutoffs (Wasserstein et al., 2019), we report 95% confidence intervals for all major findings. The analyses were carried out using the Stata statistical software package version 15 (StataCorp, 2017) and the ArcGIS software (ESRI, 2011). We checked the robustness of the main results by estimating the relationship between key variables in a step-wise manner, controlling for different set of covariates in each step.

RESULTS

Descriptive Data

In our analytical sample of 25,285 WCBA for whom both measures of the nutritional double burden (overweight and anemia) were available, 73,941 PSC, and 69,082 mother-child pairs, the prevalence rates of the three measures of the double burden were: overweight and anemic WCBA: 8.4%; overweight and stunted PSC: 2.7%; overweight WCBA and stunted PSC: 6.9% **(Table 2)**. The average forest cover loss between 2000 and 2012, the independent variable, was ∼2%.

The average age of the child was 35 months and there were equal proportions of boys and girls in the sample. The vast majority of mothers in the sample (75%) had primary level education or lower. Among the mothers, the average age was 30 years, nearly 43% were anemic and 22% were overweight.

The average number of members in a household was 7.4. Approximately 61% households had access to an improved source of drinking water and 43% had access to improved sanitation. Two-thirds of the children lived in areas classified as rural. The average forest cover in 2000, the aridity index and the distance to the nearest road were 20%, 8, and 16 km, respectively.

Main Results

In **Table 3**, we report odds ratios from estimating equation (1), separately for the three dependent variables. In each case, we first show results from estimating the equation controlling for all covariates except household's ownership of agricultural land and livestock. We then show odds ratios from estimating the equation with all covariates, including ownership of agricultural land and livestock. As mentioned earlier, we present results in this manner in an attempt to shed light on changes in the double burden measures originating from changes in deforestation independent of agricultural land use.

In regressions that do not control for the ownership of agricultural land and livestock, forest cover loss was marginally associated with overweight and stunted PSC [odds ratio (95% CI): 4.74 (0.80, 27.88)], but not with overweight and anemic WCBA [odds ratio (95% CI): 0.71 (0.15, 3.35)] or with overweight WCBA and stunted child [odds ratio (95% CI): 2.53 (0.82, 7.81)]. The odds ratios remain largely unchanged even when we control for the ownership of agricultural land and livestock.

Among the covariates, in both set of models, primary education among WCBA was associated with higher odds of concurrent anemia and overweight relative to no education. However, there was marginal or no difference between women with no education and those with higher education. Higher education among women were associated with lower odds of overweight mother and stunted child, while secondary education was associated with overweight and stunted child. Household wealth was strongly positively associated with two of the three measures, but not with overweight and stunted child. Other factors strongly associated with concurrent anemia and overweight among WCBA included women's age, urban location, access to improved water, access to improved sanitation, and aridity index. Factors strongly associated with overweight mother and stunted child included women's age, urban location,

TABLE 2 | Descriptive characteristics of the analytic sample.

Variables	*n*	Mean (SD) or %
Dependent variables		
Overweight and anemic WCBA, %	25,285	8.38
Overweight and stunted PSC, %	73,941	2.67
Overweight WCBA and stunted PSC, %	69,082	6.89
Principal independent variable		
Forest cover loss (2000–2012)[a], %	73,941	2.05
Child-level covariates		
Child age, months	73,941	34.74 (13.87)
Child sex, %	73,941	
Female		49.76
Male		50.24
Highest attained education of mother, %	73,941	
None		42.99
Primary		32.32
Secondary		21.47
Post-secondary		3.22
Woman-level covariates		
Age in years	25,285	29.52 (7.38)
Anemic (hemoglobin concentration<120 g/L), %	25,285	42.92
Overweight (BMI≥25), %	25,285	21.61
Household-level covariates		
Wealth quintiles, %	73,941	
Lowest		29.14
Low		18.52
Middle		17.85
High		18.09
Highest		16.40
Household size	73,941	7.38 (4.26)
Household access to improved water source, %	73,941	60.65
Household access to improved sanitation, %	73,941	43.79
Ownership of agricultural land, %[d]	73,941	65.80
Ownership of livestock, %	73,941	57.68
Cluster-level covariates		
Cluster location, %		
Urban	73,941	33.32
Rural		66.68
Forest cover (2000)[b], %	73,941	20.23 (21.95)
Aridity index[c]	73,941	7.97 (4.72)
Distance of cluster to nearest road, km	73,941	15.84 (16.60)

Values are proportions or means. The first measure of double burden is at the mother-level, the second measure is at the child-level, and the third measure is for the mother-child pair (see text). [a]Forest cover loss is measured as the mean of 30 m by 30 m grid cell data (i.e., 0's and 1's) at a 5 km resolution, multiplied by 100. [b]Forest cover (2000) is based on forest cover data (Hansen et al., 2013) indicating percentage of forest cover per 30 m grid cell, aggregated to 5 km resolution. [c]Global Aridity Index reflects mean annual precipitation and evapotranspiration per cluster based on CGIAR Global Aridity Index dataset; a higher number indicates higher humidity. [d]For Liberia, the survey asked "if any member of the household farmed agricultural land" instead of what was asked in the remaining countries: "does your household own any agricultural land?".

household size, access to improved sanitation, child's age and gender, forest cover in 2000, and distance to the nearest road. Finally, factors strongly associated with overweight and stunted child included mother's age (negative association), access to improved sanitation, child's age, and forest cover in 2000 (negative and marginally significant association). In sum, different sets of covariates influenced the double burden measures depending on the measure we examined, and the only covariate that influenced all three measures in the same direction was improved sanitation. Ownership of agricultural land and livestock were associated with lower odds of concurrent anemia and overweight among WCBA, while only the ownership of livestock was associated with lower odds of overweight mother and stunted child. We found no association between the ownership of agricultural land or livestock and overweight and stunted PSC.

Although the association between forest loss and overweight and stunted PSC were only marginally significant (i.e., significant only at the 10% significance level), the estimated odds ratio was robust to controlling for different set of covariates, as shown in **Appendix Table A1**.

Potential Mechanisms, and Results by Region

Discussion in this subsection proceeds in the following manner. We first discuss the relationships between forest cover loss and intermediate outcomes (e.g., ownership of livestock), and between forest cover loss and the individual components of our double burden measures (e.g., anemic WCBA) for the overall sample. We then discuss the relationships by region.

For the overall sample, forest cover loss was strongly associated with lower odds of owning livestock, but not associated with ownership of agricultural land (**Table 4A**). It was strongly associated with lower odds of having access to improved source of water and marginally associated with improved sanitation. It was strongly associated with higher odds of the occurrence of diarrhea among the PSC.

In terms of the individual components of the double burden measures, forest cover loss was associated with higher odds of anemic WCBA and not associated with any of the remaining three components (**Table 5A**).

Table 5A also provides some insight into our main finding reported in **Table 3** and suggests that the strong association between forest cover loss and only one of the three double-burden measures is due to approach in which we construct the double-burden measures. In **Table 5**, among the three measures of double burden, the associations with forest cover and individual components are in the same direction only for the components related to PSC (stunted and overweight child). For the remaining two measures the associations are in the opposite direction. For example, take anemic WCBA and overweight WCBA. Forest cover loss is positively associated with the odds of anemic WCBA but negatively associated (although statistically not significant) with overweight women. Likewise, forest cover loss is positively associated with stunted PSC but negatively associated with overweight WCBA.

The wide confidence intervals reported in this table preclude an analysis at a lower geographic level (e.g., country), which would be more meaningful for designing policies. The small

TABLE 3 | Odds ratio from a logistic regression of measures of double burden on forest cover loss.

	Overweight women with anemia		Overweight mother and stunted child		Overweight and stunted child	
	(1)	(2)	(3)	(4)	(5)	(6)
Forest cover loss, 2000–2012	0.711 [0.151, 3.350]	0.713 [0.153, 3.325]	2.529 [0.818, 7.814]	2.466 [0.801, 7.596]	4.736* [0.804, 27.88]	4.801* [0.816, 28.25]
Education of mother (none)						
Primary	1.215*** [1.059, 1.393]	1.200*** [1.047, 1.376]	0.976 [0.893, 1.067]	0.976 [0.893, 1.066]	0.938 [0.818, 1.075]	0.939 [0.818, 1.076]
Secondary	1.134* [0.977, 1.316]	1.107 [0.954, 1.285]	0.921 [0.831, 1.022]	0.919 [0.829, 1.019]	0.799*** [0.677, 0.944]	0.802*** [0.679, 0.947]
Higher	1.119 [0.811, 1.543]	1.103 [0.801, 1.520]	0.681*** [0.560, 0.829]	0.678*** [0.557, 0.825]	0.728 [0.496, 1.069]	0.730 [0.497, 1.071]
Wealth quintiles (lowest)						
Low	1.382*** [1.164, 1.642]	1.377*** [1.160, 1.635]	1.165*** [1.051, 1.291]	1.176*** [1.060, 1.304]	1.161** [1.005, 1.342]	1.163** [1.006, 1.343]
Middle	1.557*** [1.315, 1.843]	1.542*** [1.303, 1.825]	1.280*** [1.156, 1.418]	1.288*** [1.163, 1.427]	0.934 [0.801, 1.088]	0.935 [0.802, 1.090]
High	2.048*** [1.722, 2.435]	1.982*** [1.666, 2.358]	1.349*** [1.210, 1.505]	1.347*** [1.207, 1.502]	0.997 [0.843, 1.178]	1.002 [0.848, 1.185]
Highest	1.988*** [1.645, 2.402]	1.894*** [1.566, 2.290]	1.484*** [1.311, 1.680]	1.471*** [1.300, 1.666]	0.835* [0.685, 1.018]	0.842* [0.690, 1.026]
Women's age in years	1.050*** [1.044, 1.058]	1.051*** [1.044, 1.058]	1.034*** [1.030, 1.039]	1.035*** [1.030, 1.040]	0.991** [0.983, 0.998]	0.991** [0.983, 0.998]
Location (rural)						
Urban	1.463*** [1.268, 1.689]	1.293*** [1.115, 1.500]	1.252*** [1.139, 1.377]	1.192*** [1.081, 1.315]	0.975 [0.840, 1.132]	0.989 [0.848, 1.153]
Household size	0.996 [0.983, 1.010]	1.005 [0.992, 1.019]	1.013** [1.003, 1.023]	1.016*** [1.006, 1.027]	0.999 [0.983, 1.014]	0.998 [0.982, 1.014]
Improved water source (no)	1.178*** [1.042, 1.333]	1.163** [1.028, 1.315]	1.017 [0.943, 1.097]	1.011 [0.938, 1.091]	0.986 [0.878, 1.108]	0.987 [0.878, 1.109]
Improved sanitation (no)	1.271*** [1.127, 1.435]	1.224*** [1.084, 1.381]	1.165*** [1.078, 1.258]	1.156*** [1.070, 1.248]	1.253*** [1.109, 1.415]	1.258*** [1.113, 1.421]
Forest cover (2000), %	1.002 [0.998, 1.005]	1.002 [0.999, 1.006]	1.004** [1.001, 1.006]	1.004*** [1.001, 1.007]	0.996* [0.991, 1.000]	0.995* [0.991, 1.000]
Aridity Index	1.032*** [1.012, 1.051]	1.024** [1.005, 1.043]	1.004 [0.992, 1.017]	1.001 [0.988, 1.013]	0.983 [0.963, 1.004]	0.984 [0.963, 1.005]
Distance of cluster to nearest road, km	0.998 [0.994, 1.002]	0.998 [0.994, 1.002]	0.995*** [0.992, 0.998]	0.995*** [0.992, 0.998]	1.000 [0.995, 1.005]	1.000 [0.995, 1.005]
Ownership of agricultural land (no)		0.727*** [0.643, 0.822]		0.939 [0.867, 1.018]		1.076 [0.947, 1.223]
Ownership of livestock (no)		0.882** [0.784, 0.993]		0.892*** [0.827, 0.962]		0.972 [0.865, 1.093]
Child's age, months			1.050*** [1.036, 1.064]	1.050*** [1.036, 1.064]	1.032*** [1.012, 1.053]	1.032*** [1.012, 1.053]
Child's age squared			0.999*** [0.999, 1.000]	0.999*** [0.999, 1.000]	1.000*** [0.999, 1.000]	1.000*** [0.999, 1.000]
Child sex (male)			1.137*** [1.069, 1.210]	1.137*** [1.069, 1.209]	0.959 [0.872, 1.054]	0.959 [0.872, 1.054]
Child had diarrhea recently			1.048 [0.953, 1.153]	1.049 [0.953, 1.154]	0.896 [0.763, 1.053]	0.896 [0.763, 1.053]
Chi-squared	1059.92***	1098.26***	705.34***	720.79***	1261.79***	1263.82***
N	25,285	25,285	69,082	69,082	73,941	73,941

**p* < 0.10, ***p* < 0.05, ****p* < 0.01. *Each column represents results from a separate equation. All models were adjusted for the covariates shown as well as country fixed effects, cluster-level random effects, and the month of the survey. Standard errors were clustered at the level of DHS enumeration clusters. The reference categories for all covariates are shown in parentheses next to the name of each covariate. In columns (2), (4) and (6), we control for household ownership of agricultural land and ownership of livestock, in addition to the covariates in columns (1), (3) and (5), respectively.*

TABLE 4 | Odds ratio from logistic regressions of intermediate factors on forest cover loss.

	Ownership of livestock	Ownership of agricultural land	Improved water source	Improved sanitation	Child had diarrhea recently
A: Overall					
Forest cover loss, 2000–2012	0.005*** [0.001, 0.019]	1.333 [0.230, 7.731]	0.023*** [0.003, 0.193]	0.175* [0.025, 1.228]	5.052*** [2.256, 11.310]
Chi-squared	57.92***	0.10	12.14***	3.07*	15.51***
N	73,941	73,941	73,941	73,941	73,941
B: West Africa					
Forest cover loss, 2000–2012	0.000861*** [0.000112, 0.00659]	0.0251*** [0.00195, 0.323]	10.41 [0.582, 186.2]	2.518 [0.131, 48.27]	11.02*** [3.053, 39.75]
Chi-squared	46.18***	7.99***	2.54	0.38	13.43***
N	47,214	47,214	47,214	47,214	47,214
C: Central Africa					
Forest cover loss, 2000–2012	5.212 [0.660, 41.18]	16.88* [0.881, 323.5]	0.00105*** [0.0000162, 0.068]	3.483 [0.145, 83.59]	0.296** [0.102, 0.856]
Chi-squared	2.45	3.52*	10.41***	0.59	5.04**
N	17,009	17,009	17,009	17,009	17,009

*$^*p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$. This table shows odds ratios from estimating a logistic regression of intermediate factors linking forest cover loss to nutritional status on forest cover loss. All models include cluster-level random effects. **(A)** Shows results for the entire sample used in the main analysis. **(B,C)** Show results separately for West and Central African regions, respectively.*

TABLE 5 | Odds ratio from logistic regressions of individual components of double burden measures on forest cover loss.

	Anemic women	Overweight women	Stunted child	Overweight child
A: Overall				
Forest cover loss, 2000–2012	3.288** [1.321, 8.184]	0.865 [0.288, 2.600]	1.554 [0.788, 3.064]	2.102 [0.481, 9.189]
Chi-squared	1112.10***	2316.99***	3662.12***	1408.99***
N	25,285	25,285	73,941	73,941
B: West Africa				
Forest cover loss, 2000–2012	1.317 [0.355, 4.879]	0.547 [0.110, 2.725]	2.004 [0.707, 5.682]	5.468 [0.603, 49.58]
Chi-squared	197.77***	1478.82***	2328.02***	1149.08***
N	16,487	16,487	47,214	47,214
C: Central Africa				
Forest cover loss, 2000–2012	4.636** [1.294, 16.62]	1.850 [0.360, 9.518]	1.097 [0.440, 2.733]	1.713 [0.231, 12.71]
Chi-squared	194.18***	677.07***	865.88***	106.36***
N	6,263	6,263	17,009	17,009

*$^*p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$. The table shows odds ratios from a logistic regression of individual components of the double burden measures (shown in the top row) on forest cover loss. Each odds ratio is from a separate regression (that is, the table contains odds ratios from 12 different regressions). All models were adjusted for the covariates used in the main analysis, including country fixed effects, cluster-level random effects, and the month of the survey. Standard errors were clustered at the level of DHS enumeration clusters. The 95% confidence intervals are in brackets below the odds ratios. **(A)** Includes West, Central, and East African regions. **(A)** Shows results for the entire sample used in the main analysis. **(B,C)** Show results separately for West and Central African regions, respectively.*

sample size is a concern particularly for eastern Africa (which includes only two countries: Kenya and Rwanda). Therefore, in assessing the relationship between forest cover and intermediate outcomes as well as the individual components of the double burden at the regional level, we focus on western and central Africa.

The overall results mask significant heterogeneity across geographic regions within SSA. As shown in **Table 6**, the significant association between forest cover loss and overweight and stunted PSC are driven by the association in West Africa, while no such relationship exists in Central and East Africa.

There are also significant regional differences in the association of forest cover with the intermediate factors as well as the individual measures of nutritional status. With respect to the intermediate factors, forest cover loss is associated with lower odds of owning livestock and lower odds of owning agricultural land in West Africa (**Table 4B**). It is associated with higher odds of the incidence of diarrhea among the PSCs in the region. Unlike in the overall sample, there is no association between forest cover loss and access to improved water or sanitation. In contrast, in the central region, forest cover loss is associated with *higher* odds of owning agricultural land and *lower* odds of

TABLE 6 | Odds ratio from a logistic regression of measures of double burden on forest cover loss, by region.

	Overweight woman with anemia	Overweight mother and stunted child	Overweight and stunted child
West Africa			
Forest cover loss, 2000–2012	0.31 [0.03, 2.96]	3.21 [0.60, 17.07]	16.80** [1.31, 214.2]
Chi-squared	549.00***	489.64***	1031.23***
N	16,487	42,514	47,214
Central Africa			
Forest cover loss, 2000–2012	1.98 [0.24, 15.94]	2.96 [0.58, 15.08]	2.32 [0.19, 27.78]
Chi-squared	466.25***	230.07***	62.58***
N	6,263	16,906	17,009
East Africa			
Forest cover loss, 2000–2012	0.00 [0.00, 21291700000]	0.66 [0.01, 44.11]	0.00005 [0.00, 186.10]
Chi-squared	42.13***	134.86***	115.63***
N	2,534	9,662	9,718

**p < 0.1; **p < 0.05; ***p < 0.01. This table presents odds ratios from logistic regressions of the three measures of double burden of malnutrition on forest cover loss, separately for each region. West Africa includes Benin, Cote d'Ivoire, Ghana, Guinea, Liberia, Mali, Nigeria, Senegal, Sierra Leone, and Togo. Central Africa includes the Democratic Republic of Congo, Gabon, and Zambia. East Africa includes Kenya and Rwanda. In models with overweight and anemic WCBA, we control for the women's age and education level, household wealth, household access to improved water and sanitation, household ownership of agricultural land, household ownership of livestock, DHS cluster location (i.e., urban or rural), baseline forest cover in 2000, the CGIAR Global Aridity Index, distance of cluster to nearest road, country fixed effects, cluster-level random effects, and the month of the survey. In models with overweight mother and stunted child or overweight and stunted child, we control for all the variables used in first column plus the child's age, child age squared, child gender, and whether the child had diarrhea during 2 weeks prior to survey. Standard errors were clustered at the level of DHS enumeration clusters.*

the incidence of diarrhea (**Table 4C**). The association between forest cover loss and access to improved water is in the same direction as that of the overall sample. The key message from **Table 4** is that the mechanisms through which forest cover loss influences nutritional double burden differ between the regions. With respect to the individual components of the double burden measures, however, the relationships at the regional level are in agreement with those for the entire sample (**Tables 5B,C**).

DISCUSSION

The double burden of malnutrition is linked to the ongoing epidemiologic transition, whereby non-communicable conditions, such as cardiovascular diseases and diabetes, are replacing communicable conditions, such as malaria, as the primary causes of morbidity and mortality in low- and middle-income countries (Shrimpton and Rokx, 2012). An emerging body of literature has examined the socio-economic determinants (e.g., education, urbanicity, and income) of the double burden within the same household (Kosaka and Umezaki, 2017). Existing literature has also shown that environmental changes such as deforestation are linked to changes in diet, a key input to an individual's nutritional status (Rasmussen et al., 2017; Reed et al., 2017), as well as measures of malnutrition, such as underweightness (Pienkowski et al., 2018). However, remarkably little has been written on the possible linkages between deforestation and the nutritional double burden. Generally speaking, the ecological determinants of the double burden are poorly understood. This is an important omission because the double burden of malnutrition may be an important mechanism linking environmental changes to diets and non-communicable conditions (Frumkin and Haines, 2019). More importantly, deforestation may have a temporal dimension, through which different segments of the population might be affected differently and the same individual might be affected differently based on the outcome measured. In the current study, however, we found only a marginal association between the measure of the double burden pertaining to the child (overweight and stunted PSC) and forest cover loss and no association between forest cover loss and the remaining two measures. These findings suggest that the effect of deforestation observed in other health outcomes may either not extend or extend only marginally to the nutritional double burden, although the lack of association seems in part due to the way the double burden measures are constructed.

We must interpret our findings with a number of caveats. First, as we have pointed out previously (Galway et al., 2018), the Global Forest Change dataset defines trees as vegetation taller than five meters, and therefore underestimates true forest loss. Second, the forest loss data used does not take into account reforestation that may have taken place during the period. If the current double burden of malnutrition reflects the net effect of deforestation and reforestation, our estimates are underestimates of the true association.

Third, although we controlled for a range of potential confounders in our analysis, we cannot interpret the observed associations as causal. Our data are cross-sectional and as such we are unable to employ panel data methods available for establishing causal relationships. There is limited possibility of reverse causality (i.e., double burden of malnutrition triggering deforestation), in part because, by our study design, the timing of forest cover loss (2000–2012) precedes that of the DHS surveys (after 2013). Nonetheless, there could be omitted variables—such as food availability and access to market—for which we could not control given the data. Finally, we are unable to comment fully on how land use patterns, income, and lifestyle (e.g., amount of movement, time spent on fetching water and firewood or collecting fodder for livestock) may have changed as a result of deforestation and how those changes may influence the nutritional double burden.

Notwithstanding these limitations, the association between forest cover loss and the measure having child-level indicators—although it is only marginal—is worrying from a policy perspective, given the long-lasting nature of early-life nutritional deficiencies. A large body of research has shown that poor nutritional status in childhood has lasting effects into adulthood. For example, early-life nutrition is an important determinant of one's long-term productivity, earnings, and health (Alderman et al., 2006; Dewey and Begum, 2011; Currie and Vogl, 2013). Our findings contribute to the ongoing dialogue about the need to prevent deforestation and conserve biodiversity at a range of spatial scales for multiple goals, including limiting the potential adverse impacts on child malnutrition and children's long-term wellbeing.

More generally, we have previously shown that deforestation is associated with the diversity and quality of children's diet, and proposed various mechanisms for the association (Galway et al., 2018). The analysis presented here—using the same set of countries and periods as the previous study—showed that the association of forest cover loss may extend, albeit marginally, to children's anthropometric measures of the double burden of malnutrition, specifically the likelihood of simultaneously being overweight and stunted. One possible mechanism for this association is that reduced consumption of legumes and nuts, flesh foods, and other fruits and vegetables resulting from deforestation—as we established in the previous study—deleteriously affects child linear growth. Simultaneously, deforestation may raise the risk of overweight if calorie intake is increased by consuming energy-dense processed foods. When we examined the association between components of double burden measures—stunting and overweight—we find that both of these measures are positively associated with forest cover loss although the associations are not statistically significant.

Given the limitations of our data, we are unable to examine why deforestation is not associated with the prevalence of anemic and overweight WCBA or overweight WCBA and stunted PSC and is only marginally associated with the prevalence of overweight and stunted PSC. Additional data—including on, for example, the drivers of deforestation and usages of the deforested areas—will be needed to examine such questions. Likewise, because the prevalence of overweight also depends on lifestyle and habits (see Figure 4 in WHO, 2017), information on how deforestation influences these factors will be central to understanding the temporal dimension, if one exists. One can hypothesize, for example, that when forests are cleared for construction of building, it increases the amount of time women spend on finding fodder for cattle or firewood, thus expending more calories and reducing the chances of gaining excess weight (thus reducing the chances of being overweight and anemic). For the children, on the other hand, cleared land and development of a local marketplace may mean easier access to processed foods that contribute to unhealthy weight gains (thus increasing the chances of being overweight). Similarly, one can hypothesize that, other environmental factors that deforestation affects, such as access to clean drinking water (Mapulanga and Naito, 2019), may affect the nutritional status of young children, but not adults. These examples are only illustrative of the complex ways that deforestation may influence individual behavior and contribute to the double burden of malnutrition.

Additional research will be required to confirm the findings in our study, establish mechanisms, and to uncover any potential temporal dimensions (specifically, to examine different effects of forest cover loss among different population segments). For effective policy design, the mechanisms will need to be examined at the regional and country levels, as the mechanism may vary at those levels—as our analysis at the level of the regions within SSA suggests. For example, it is not clear why forest cover loss is negatively associated with the ownership of agricultural land in the West Africa region but positively associated in the Central Africa region. Some of the differences in potential mechanisms may be due to social and economic factors, such as education and wealth as the regions vary widely in these dimensions, but this needs to be investigated further.

For effective policy design, it is also important to understand the relative importance of different mechanisms shown in **Figure 1**. Among the various mechanisms, as mentioned before, several studies have hypothesized and assessed the association between forest cover loss and diets. This focus on diets is not surprising given SSA's reliance on forests for food. For example, 60 percent of the households in 11 African countries have been shown to collect wild food from forests (Hickey et al., 2016). Likewise, using data from 37 communities in 24 countries, including five in Africa, Rowland et al. (2017) find that more than half of the households in their sample collected forest food for consumption. However, the research linking forest cover loss to diets is far from conclusive, and the reliance on forests for food varies across regions, cultures, and population sub-groups. The strength of other mechanisms, therefore, likely varies across these dimensions as well.

CONCLUSION

Deforestation does not seem to be an important driver of the double burden of malnutrition in SSA. We found no association between deforestation and measures of the double burden pertaining to the same WCBA or mother-child pair within the household. The association we found between deforestation and the measure based on the same child is worrying from a policy perspective, even though additional research will be required to confirm it. If this association is robust, the effect of deforestation may have a temporal aspect, which also warrants further research. More generally, there is a need to better understand the potential mechanisms linking forest cover loss to health and nutrition, their relative contributions, and differences across geographic regions and countries.

DATA AVAILABILITY STATEMENT

Data used in this study were obtained from Demographic and Health Surveys (DHS) and the Global Forest Change dataset. The DHS data are available after a simple registration process (https://dhsprogram.com/data/new-user-registration.cfm). The Global Forest Change data are available from http://earthenginepartners.appspot.com/science-2013-global-forest.

ETHICS STATEMENT

Ethical approval was not required for this study as it used publicly-available de-identified data.

AUTHOR CONTRIBUTIONS

YA, LG, and AJ conceptualized the study. YA and LG prepared the dataset. YA and SN conducted the statistical analysis. All authors contributed in the preparation of the final manuscript and approved it for publication.

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APPENDIX

TABLE A1 | Odds ratio from step-wise logistic regressions of overweight and stunted child on forest cover loss.

	(1)	(2)	(3)	(4)
Forest cover loss, 2000–2012	4.801* [0.816, 28.25]	4.970* [0.846, 29.20]	5.108* [0.868, 30.07]	5.544* [0.945, 32.53]
Child sex (male)	0.959 [0.872, 1.054]			
Child's age, months	1.032*** [1.012, 1.053]			
Child's age squared	1.000*** [0.999, 1.000]			
Child had diarrhea recently (no)	0.896 [0.763, 1.053]			
Education of mother (none)				
Primary	0.939 [0.818, 1.076]	0.937 [0.817, 1.074]		
Secondary	0.802*** [0.679, 0.947]	0.801*** [0.678, 0.946]		
Higher	0.730 [0.497, 1.071]	0.730 [0.498, 1.071]		
Women's age in years	0.991** [0.983, 0.998]	0.991** [0.984, 0.999]		
Wealth quintiles (lowest)				
Low	1.163** [1.006, 1.343]	1.161** [1.005, 1.342]	1.158** [1.002, 1.337]	
Middle	0.935 [0.802, 1.090]	0.936 [0.803, 1.091]	0.928 [0.796, 1.081]	
High	1.002 [0.848, 1.185]	1.003 [0.848, 1.185]	0.977 [0.828, 1.153]	
Highest	0.842* [0.690, 1.026]	0.841* [0.690, 1.026]	0.787** [0.649, 0.955]	
Household size	0.998 [0.982, 1.014]	0.998 [0.982, 1.014]	0.996 [0.980, 1.011]	
Improved water source (no)	0.987 [0.878, 1.109]	0.988 [0.879, 1.110]	0.980 [0.872, 1.101]	
Improved sanitation (no)	1.258*** [1.113, 1.421]	1.258*** [1.113, 1.421]	1.244*** [1.101, 1.405]	
Ownership of agricultural land (no)	1.076 [0.947, 1.223]	1.076 [0.947, 1.222]	1.081 [0.951, 1.228]	
Ownership of livestock (no)	0.972 [0.865, 1.093]	0.974 [0.867, 1.095]	0.974 [0.866, 1.094]	
Forest cover (2000),	0.995* [0.991, 1.000]	0.995* [0.991, 1.000]	0.995** [0.990, 1.000]	0.995* [0.991, 1.000]
Aridity Index	0.984 [0.963, 1.005]	0.985 [0.964, 1.006]	0.979* [0.959, 1.000]	0.976** [0.956, 0.997]
Distance of cluster to nearest road, km	1.000 [0.995, 1.005]	1.000 [0.995, 1.005]	1.000 [0.995, 1.005]	1.000 [0.996, 1.005]
Location (rural)	0.989	0.989	0.966	0.927
Urban	[0.848, 1.153]	[0.849, 1.154]	[0.829, 1.125]	[0.811, 1.059]
Chi-squared	1263.82***	1254.50***	1241.00***	1213.40***
N	73,941	73,941	73,941	73,941

$^{*}p < 0.1$; $^{**}p < 0.05$; $^{***}p < 0.01$.
All models were adjusted for the covariates shown as well as country fixed effects, cluster random effects and the month of the survey. Standard errors were clustered at the level of DHS enumeration clusters. The reference categories for all covariates are shown in parentheses next to the name of each covariate.

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Deconstructing Diets: The Role of Wealth, Farming System, and Landscape Context in Shaping Rural Diets in Ethiopia

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Dietary diversification is central to improving dietary quality and nutrition for food security. Several factors have been linked to higher diversity diets, including household wealth, market access, on-farm crop diversity, and regional forest cover. How these factors combine in landscapes to shape diets, however, is not well-understood. We take the Ethiopian context as a case study of how wealth, farming system type, and landscape context interact to explain household dietary profiles. Using cluster analysis on nationally representative data on household food consumption, we identify three distinct dietary profiles across rural Ethiopia: (1) A low diversity diet, (2) A diverse diet particularly rich in fruit and vegetables, and (3) A diverse diet also rich in oils, fats, and sugars. We find that the low diversity diet was strongly associated with households in the bottom and middle wealth classes that were mostly involved in cereal-based farming, although not exclusively. In contrast, the diverse diet high in fruit and vegetables was primarily composed of households with coffee-agroforestry farming systems, and did not appear to be limited to any particular wealth class, although it was positively associated with forest cover. Households with a diverse diet profile also rich in oils, fats and sugars were stratified across multiple different farming types, situated closer to roads, and primarily came from the middle and top wealth classes. Finally, while forest cover was strongly associated with a dietary profile rich in fruits and vegetable and the pursuit of coffee-agroforestry farming, the forest cover in cereal-based systems was still significantly positively associated with the consumption of dark green leafy vegetables and fruits. This suggests that even small amounts of forest cover can contribute to healthy diets. These results, which illuminate how wealth, farming system type, and landscape context shape dietary profiles, have important implications for the design of effective food security policies in Ethiopia.

Keywords: Ethiopia, dietary diversity, forest cover, poverty, agriculture, cluster analysis

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INTRODUCTION

Globally, more than two billion people suffer from micronutrient deficiencies caused by poor diets (Haddad et al., 2015), which can impair childhood development and adult productivity (Lim et al., 2012; Black et al., 2013). The challenge is particularly severe in Africa where poor investments in agriculture have led to large yield gaps (Tittonell and Giller, 2013), limited processing and storage facilities, low income levels, and inadequate consumer understanding of micronutrient deficiencies (Barrett and Bevis, 2015). To date, most large-scale food security policies and funding efforts have placed heavy emphasis on meeting basic dietary energy intake by increasing the production and availability of staple crops (World Health Organization, 2005; Forouzanfar et al., 2016), with less attention to the nutritional constituents of diets (Ickowitz et al., 2019). While such efforts have reduced the proportion of hungry people globally (FAO, 2018), the current global agricultural system does not provide the foods necessary for nutritionally adequate diets (Ickowitz et al., 2019; Willett et al., 2019). Reasons include policy interventions that trigger a shift away from diversified agricultural production and consumption to cash crop monocultures, which can negatively impact the nutritional quality of diets (Siegel et al., 2014; Powell et al., 2015; Qaim et al., 2016; Qaim and Sibhatu, 2018). Moreover, agricultural expansion and conventional intensification are often associated with deforestation (Angelsen and Kaimowitz, 2001; Ordway et al., 2017; Curtis et al., 2018), which can reduce dietary diversity by decreasing the availability of wild foods (Rowland et al., 2016; Galway et al., 2018) and other forest products that can be sold to enable the purchase of diverse foods (Hickey et al., 2016).

Nutritionally, “better” diets include the consumption of multiple different types of foods, as they are more likely to meet human macro- and micro-nutrient requirements (Hall et al., 2009; Lachat et al., 2018). In response, food security programs are now placing focus on dietary diversification as a key strategy to improve dietary quality and nutrition (e.g., Dube et al., 2018; Ochieng et al., 2018; Schreinemachers et al., 2018). However, the factors that lead households to consume a diverse diet are not well-understood, even though such understanding is critical for designing effective food security and nutrition policies. Higher diversity diets have been linked to a number of household characteristics including the level of education, age and gender of the head of the household, as well as household size and household wealth (Cockx et al., 2018). Dietary diversity has also been associated with market access (Sibhatu et al., 2015; Qaim et al., 2016), on-farm production—whereby farm diversification is assumed to lead to the consumption of more diverse diets (Jones et al., 2014; Sibhatu and Qaim, 2018)—as well as regional forest cover (Ickowitz et al., 2014). Recent studies have demonstrated that tree cover in landscapes positively correlates with dietary diversity as well as the consumption of nutritionally important food groups in African countries (Ickowitz et al., 2014; Rasmussen et al., 2019). However, how these factors (household characteristics, market access, on farm production, and landscape context) combine to shape diets is not well-understood, especially the mechanisms whereby forests support more diverse diets.

Previous studies assessing how diets are influenced by farm production diversification have mostly relied on simple measures of crop diversity, such as counts of the total number of crop and livestock species on a farm (Sibhatu et al., 2015). Yet, the identity and relative proportion of each species may also be critically important to understanding their relation to diets. Smallholders' diets benefit differently from on-farm production of food crops (which can be directly eaten) vs. both food and non-food cash crops (which generate income that can be used to purchase food) (Jones, 2016). Crops also vary in their nutritional characteristics; for example, dark green leafy vegetables are an important source of iron, calcium and fiber, while red peppers, carrots, and pumpkins are critical for vitamin A. Similarly, cows, sheep, and goats provide dairy, whereas chickens, ducks, and other fowl provide eggs in addition to meat protein. Thus, farming systems that grow a diversity of similar crops (e.g., wheat, teff, maize, sorghum) will contribute less to dietary quality than farming systems with a diverse array of different fruits, vegetables, grains, etc. Assessing the linkages between on-farm production and households' dietary diversity thus requires a more nuanced characterization of on-farm crop and livestock diversity.

Many low- and middle-income countries are currently in the midst of a nutrition transition where traditional diverse diets (rich in e.g., vegetables and fruits) are being replaced with poorer quality diets excessive in fats and oils, and sugar (Abrahams et al., 2011; Steyn and Mchiza, 2014; Cockx et al., 2018). Improving our understanding of the factors associated with different dietary profiles will enable the design of efficient multi-pronged strategies that aim to achieve food security and nutrition goals. Moreover, a better understanding of how forests impact diets can shape agricultural policies that are better integrated with forest conservation and restoration targets (Sunderland et al., 2019).

The objectives of this study are to examine how wealth, farming system type, and landscape context interact and influence the diets of rural households across Ethiopia. Specifically we aim to: (i) identify the predominant dietary profiles of households across rural Ethiopia; (ii) develop farm typologies based on household agricultural production data to move beyond simple species counts; (iii) compare how households' wealth status and farming system type relate to their dietary profile; and (iv) identify whether the landscape context (forest cover and market access) is associated with the consumption of different food groups and food items.

METHODS

Constructing Dietary Profiles and Measuring Dietary Quality

We used publically available data from the World Bank's Living Standard Measurement Survey (LSMS) (http://microdata.worldbank.org/index.php/catalog/lsms) conducted in Ethiopia in 2015–2016 to build dietary profiles. The LSMS is a nationally representative household survey that collects a wide array of livelihood data, including details on farm-level crop production and livestock holdings, asset ownership and food consumption records. We focused our analysis on the diets of rural households and extracted data for the 2633 households surveyed in rural areas. Our final sample ($n = 2367$ rural households) excluded households for which farm production, food consumption, geo-location data, or asset data were missing, as complete information on each household was required in our analysis.

We used reported household food consumption (binary variable of consumption/no consumption over the past 7 days of 69 different food items (see **Figure 1**) from the LSMS to construct dietary profiles across rural households in Ethiopia. The LSMS dataset offers advantages over other similar sources of data (e.g., DHS, see Ickowitz et al., 2014), including: large sample sizes, extensive data on a diverse set of non-diet variables, and a disaggregated record of many individual foods, which permits the calculation of different dietary diversity scores (FAO and FHI 360, 2016). It also permits the construction of dietary profiles based on food items rather than aggregated food groups, allowing for an enhanced understanding of which food items contribute frequently to each food group in diets.

Following Alvarez et al. (2018), we used a principal component analysis (PCA) and hierarchical clustering analysis (HCA) on the list of consumed food items (presence/absence) reported in the 7-day recall by each household to construct a typology of dietary profiles (*sensu* Hu, 2002). We used a PCA to reduce the full list of reported food items ($n = 69$, see **Figure 1**) consumed by each household into a smaller number ($n = 5$) of synthetic but orthogonal variables, i.e., principal components (PC). In our analysis the first PC explained 50% of the total variance and the second an additional 5% (for a factor map of food items contributing most to the first two PC see **Figure S1**). We then applied Agglomerative Hierarchical Clustering to the household's PC scores using Ward's minimum-variance method to identify diet clusters, which we hereafter refer to as "*dietary profiles*." Ward's method was chosen as it builds clusters by minimizing within-cluster variation through iterative comparison amongst clusters using the calculated sum of squares between the two clusters, summed over all variables (Hair et al., 2010). The appropriate number of clusters (i.e., dietary profiles) was defined using the dendrogram shape. All statistical analyses were executed in R (version 3.1.0, ade4 package, Dray and Dufour, 2007).

We then characterized each of the identified dietary profiles in terms of the percentage of households with that dietary profile consuming specific food items and/or food groups. We calculated a mean dietary diversity score for households within each dietary profile. Dietary diversity is defined as the number of food groups consumed over a fixed period (generally ranging from 24 h to 7 days). At individual level, dietary diversity is a proxy for micronutrient adequacy of the diet (Arimond et al., 2010) which is one aspect of diet quality. Household diets are highly correlated with individual diets, yet household-level diet diversity does not account for intra-household distribution and can not be used

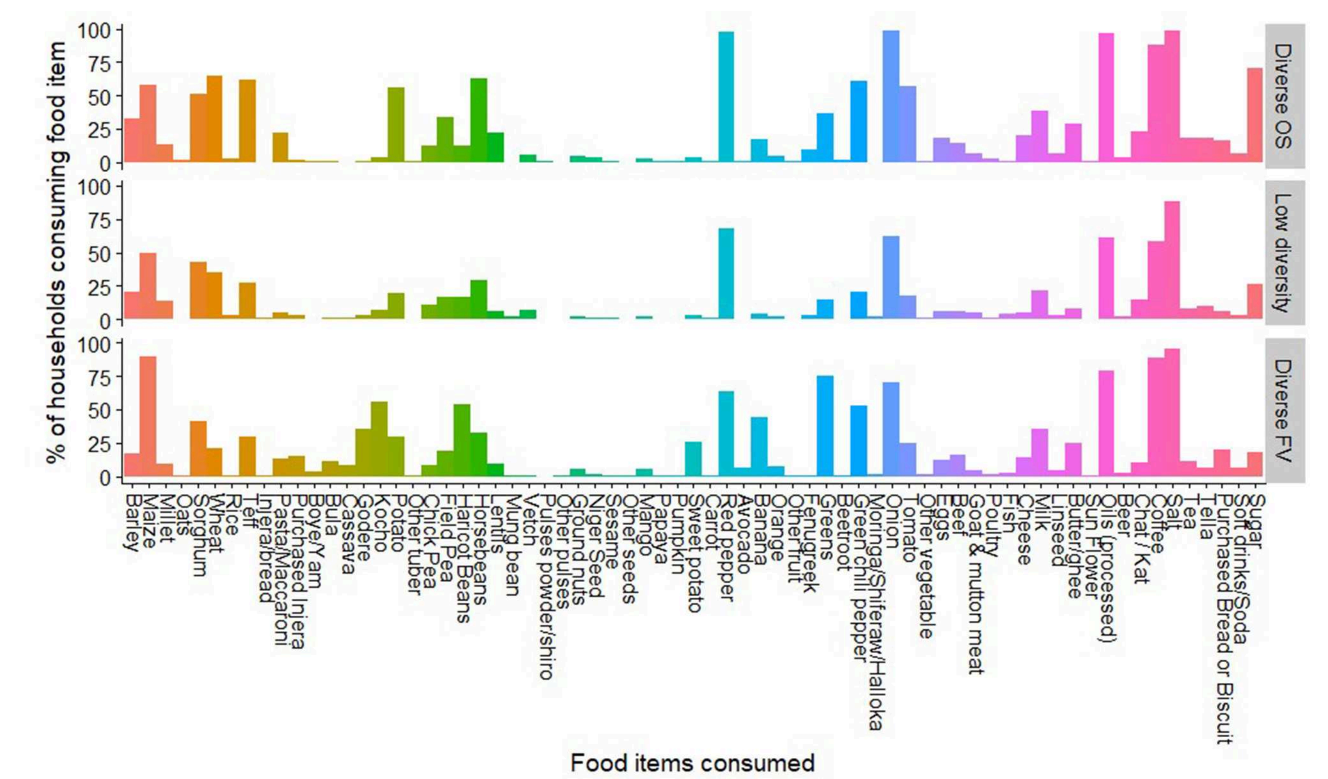

FIGURE 1 | Percentage of households reporting the consumption of various food items within the past 7 days for each of the three identified Dietary Profiles. $N =$ 2367 rural households. OS, Oils and Sugars; FV, Fruits and Vegetables.

for statements concerning particular population groups, such as women (Verger et al., 2019). As the LSMS records food consumption at the household level rather than the individual level, we constructed a modified household dietary diversity score (MHHDS) using the ten food groups recommended to construct MDD-W (Minimum Diet Diversity of Women) (FAO and FHI 360, 2016) but based on a recall of the past 7 days. The MHHDDS includes the following food groups: (1) starchy staple foods (cereals, white roots, tubers, plantains), (2) vitamin A-rich vegetables and fruits, (3) dark green leafy vegetables, (4) pulses (beans and peas), (5) nuts and seeds, (6) flesh foods (meat, poultry and fish), (7) dairy, (8) eggs, (9) other vegetables, and (10) other fruits (FAO and FHI 360, 2016). In addition to the food groups included in the MHHDDS, we also compared the consumption of sweets, oils and fats, but these food groups were not included in the calculation of the MHHDDS.

Developing Farming Systems Typologies

What households produce is known to influence their diets (Jones et al., 2014; Jones, 2015, 2016, 2017). To move beyond the influence of crop counts on diets, we used the LSMS data on household farm crop production and livestock holdings to develop farm typologies across Ethiopia using cluster analysis. We based the analysis on relative production of each crop and livestock species (calculated separately). From the surveyed households in the LSMS data we identified 187 unique crop and 10 livestock species. We converted reported crop production into common units of mass (kg), and livestock into Tropical Livestock Units (TLU) based on conversions developed by Gryseels (1988) for the Ethiopian context. We extracted data on elevation [from the MODIS digital elevation model (250 m resolution)] and annual mean precipitation [from Worldclim.org (Fick and Hijmans, 2017)] using geo-location data from the LSMS and included these with agricultural production data in the cluster analysis. We used a Hierarchical Cluster Analysis based on euclidean distances to group farms based on these characteristics and assessed the appropriate number of clusters based on the dendrogram and in discussion with experts familiar with farming systems in Ethiopia (F. Baudron, B. Powell).

Co-Variates Expected to Influence Dietary Profiles

Wealth Groups

While wealthier households may be better able to purchase costly nutrient-rich foods, which can improve diets and lead to higher dietary diversity (Sibhatu et al., 2015), scholars have also argued that nutritional outcomes do not necessarily improve with higher incomes (Herforth and Ahmed, 2015). To evaluate the association between wealth and diets, we constructed an asset-based wealth score as a proxy for households' long-term economic status. Following the approach of Filmer and Pritchett (2001), we dichotomized all household assets to indicate whether or not each household owned each of the assets listed (see **Table S1** for list of assets included in the analysis). In addition, the type of roofing material and toilet facilities were likewise dichotomized [1=Modern, 0=Non-modern (no toilet or shared facilities)]. We then applied a PCA to the dichotomized data (ownership/no ownership) with the first PC used to compute wealth quintiles (the higher the value, the wealthier the household is), which were then re-coded into three wealth groups: bottom (1st and 2nd quintile), middle (3 and 4th quintile), and top wealth class (5th quintile). We used this grouping over the use of equally sized terciles to better distinguish the more wealthy households from the generally low asset levels that characterizes much of the population in Ethiopia. We chose an asset-based score rather than an income based metric as the former has been shown to be a good proxy for the wealth of a household over time and is less susceptible to measurement error (Hjelm et al., 2016). Moreover, metrics such as proportion of income spent on food might be problematic as households tend to spend proportionally less on food as their disposable income increases (Smith et al., 2014).

Landscape Variables (Forest Cover and Market Access)

Landscape elements surrounding a household can influence both their dietary opportunities (Ickowitz et al., 2014; Galway et al., 2018; Rasolofoson et al., 2018) and their farming system (Baudron et al., 2017). To characterize local landscapes, we obtained data on forest cover in 2016 (the year of the LSMS) from a publicly available 30 m resolution annual global tree cover dataset from 2000 to 2016 (Hansen et al., 2013). We downloaded tiles covering the spatial extent of Ethiopia and derived tree cover by masking water, adding forest cover gain and subtracting forest cover loss from the base year 2000. The data show the percentage tree cover in each pixel with trees defined as vegetation taller than 5 m. To create a forest cover map, we classified each pixel to a binary forest/no forest classification, using a "forest" threshold definition of 30%. We tried other threshold definitions [10 and 60%, based on common thresholds used by the United Nations Food and Agricultural Organization and United Nations Framework Convention on Climate Change (FAO, 2000, 2005)] and chose 30% as it resulted in forest cover maps that best matched national land cover maps.

To ensure confidentiality, the LSMS does not provide geo-locations for individual households, but rather for their corresponding "enumerator area" (in most cases corresponding to a village). Ninety-nine percent of the geo-referenced points for enumerator area locations are randomly displaced by 0–5 km. The remaining 1% of enumerator areas are displaced up to a maximum of 10 km. We constructed a 10 km radius circle surrounding each enumerator area to account for this random spatial displacement as well as to capture a reasonable distance that people are likely to travel for hunting and collecting wild foods (Layton et al., 1991). We used Fragstats 4.2 (McGarigal et al., 2002) to extract percentage forest within each 10 km radius circle around villages. Finally, we also extracted data from the LSMS on the distance to nearest major road and major population centers (>20,000 inhabitants), which act as proxies for market access, for each enumerator area.

Analyses

Based on a comparison of the proportion of households consuming each food item, we used one-way ANOVAs to identify key foods within food groups that were responsible for driving differences in intake patterns between the dietary profiles. We also used separate one-way ANOVAs to compare the (a) MHHDDS and (b) proportion of households consuming each food group over the past 7 days between the three identified dietary profiles. We used Chi-square analysis to test whether the dietary profiles were related to asset-based wealth classes or farming systems.

We also tested the associations between household's dietary profile and (a) the percentage of forest in the surrounding landscape and (b) their farming system through separate one-way ANOVAs. We further tested for a relationship between the percentage of forest and (c) MHHDDS and (d) their consumption (0 or 1) of each food group over the prior 7 days. For (c) and (d), we used generalized linear mixed models (GLMMs) from the "nlme" package (for binomial data) in R (Pinheiro et al., 2018) with the enumerator area as our random effect. We included explanatory covariates previously identified by Rasmussen et al. (2019) as important predictors of diet outcomes in Ethiopia, namely the age, gender, and highest education level of the head of the household, household wealth class, the distance to nearest major road as well as annual mean temperature and rainfall. We used both a pairwise correlation matrix as well as the Variance Inflation Factor (VIF) to assess potential collinearity among the independent variables included in our models after fitting regressions. Variables were removed if the correlation coefficient was >0.5 and/or if VIF exceeded a value of 10. A full model including all covariates was run, and results from these models are reported in **Table S3** (MHHDDS as outcome variable), S4a-f (Consumption of various food groups as outcome variables), and S6a-b (consumption of oils and fats, and sweets as outcome variables). All analyses were carried out in the software R 3.4.2 (R Core Team, 2017).

RESULTS

Identification of Dietary Profiles

Using cluster analysis on food consumption data we identified three distinct dietary profiles across Ethiopia. We characterized dietary profiles based on the food items composing each of these diets (**Figure 1**) and statistical differences amongst the dietary profiles (**Tables 1**, **2**). The three identified profiles were:

1. "**Diverse Diet High in Oils and Sugars**" (hereafter referred to as "*Diverse OS*") was the smallest cluster and included 720 households (30%). It was composed of households who—in an Ethiopian context—had a relatively high average MHHDDS of 5.4. This diet had particularly high consumption of pulses, vitamin A-rich fruits and vegetables and other vegetables (e.g., onion), as well as more widespread consumption of eggs than the other dietary profiles. Beyond the consumption of the 10 food groups included in the calculation of the MHHDDS, this dietary profile had a significantly higher rate of households consuming oils and fats (97%) and sugars (71%) (**Table 2**). Also, there was a higher rate of households consuming processed foods such as pasta/macaroni (22%) as compared to the two other dietary profiles. The main staple food consumed in this diet was wheat (65% of households). Unlike the two other dietary profiles, almost all households with this profile consumed at least one food item classified as a vitamin A-rich vegetable or fruit, with the most commonly consumed item being red peppers (97%).
2. "**Low Diversity Diet**" was the most widespread dietary profile with 1052 households (44%). The distinguishing feature of this profile was the low mean MHHDDS of 3.8, which was significantly lower than the other two dietary profiles ($p < 0.001$). This diet was characterized by a reliance on maize as the staple crop (49% of households in this profile) and a generally lower proportion of households consuming most food groups than the other two dietary profiles. A lower consumption rate across all food groups—except one—suggests that it may also be a lower intake diet. The one exception was that a higher proportion of households consumed sugary food items than in the Diverse FV diet (described below).
3. "**Diverse Diets High in Fruits and Vegetables**" (hereafter '*Diverse FV*') included 861 households (36%). This dietary profile had the highest mean MHHDDS of 5.5 and maize was the most commonly consumed staple crop (90% of households), although sorghum, wheat and teff were also consumed by some households. Of the 10 food groups included in the MHHDDS, this diet had a high proportion of households consuming dark green leafy vegetables, vitamin A-rich foods, as well as "other fruits" and "other vegetables". In particular, this diet was characterized by a greater proportion of households consuming greens (75%), haricot beans (54%), vitamin A-rich sweet potato (26%), and banana (44%) compared to the other dietary profiles. In contrast to the *Diverse OS* diet, which also exhibited a high MHHDDS, fewer households following a *Diverse FV* diet consumed processed foods such as sugar (18%) and pasta/macaroni (13%).

Spatial Co-Occurrence of Dietary Profiles

The three dietary profiles were not evenly distributed spatially across Ethiopia (**Figure 2**). Households following the *Diverse FV* dietary profile were mainly located in the southwest of the country, while households following the *Diverse OS* or *Low Diversity* dietary profile were spread across the north and east of the country, with a few located along the borders with Sudan to the west and Kenya to the south.

We also assessed the co-occurrence of dietary profiles within villages to determine if particular pairs of profiles more commonly co-occur. In 23% of villages ($n = 53$), all surveyed households followed a single common dietary profile; the most frequent being the *Low Diversity* diet ($n = 30$). In 58% of villages ($n = 134$), we found two co-occurring dietary profiles, with the most frequent combination being the *Diverse OS—Low Diversity* in 67 villages, followed by *Low Diversity—Diverse FV* in 51 villages. In only 16 villages did *Diverse OS-Diverse FV* co-occur. The three dietary profiles co-occurred in 19% of the villages examined ($n = 45$).

TABLE 1 | The mean Modified Household Dietary Diversity Score (MHHDDS±SE) and the percentage of households that consumed each food group of the households following each dietary profile.

Dietary profile	MHHDDS	% Households consuming items in each food group								
		Pulses	Nuts/ seeds	Dairy	Meat/ fish poultry/	Eggs	Dark green leafy veg.	Vit.-A rich fruit/veg	Other veg.	Other fruit
Diverse OS	5.4 ± 1.2^{a}	87.4^{a}	8.3^{a}	45.1^{a}	20.6^{a}	18.2^{a}	44.9^{a}	97.4^{a}	99.7^{a}	19.4^{a}
Low diversity	3.8 ± 1.1^{b}	67.9^{b}	4.2^{b}	23.8^{b}	14.7^{b}	6.4^{b}	17.8^{b}	71.0^{b}	69.9^{b}	5.3^{b}
Diverse FV	5.5 ± 1.5^{a}	80.4^{c}	$7.1^{a,b}$	40.8^{a}	20.7^{a}	12.0^{c}	74.8^{c}	74.8^{b}	86.6^{c}	49.2^{c}

Mean MHHDDS values were compared by one-way ANOVA and Tukey Post-hoc comparisons. Values with different superscript letters are statistically different ($p < 0.01$). N = 2,367 rural households.
Cereals were consumed by all households across all dietary profiles and is not shown.

TABLE 2 | Percentage of households consuming various food items within the past 7 days by dietary profile.

Dietary profile	% Households consuming each food item							
	Sugar	Oil/fat	Pasta/ macaroni	Potato	Sweet potato	Greens	Haricot beans	Banana
Diverse OS	70.6^{a}	96.8^{a}	21.8^{a}	56.0^{a}	4.0^{a}	36.5^{a}	12.5^{a}	16.8^{a}
Low diversity	26.0^{b}	61.2^{b}	4.8^{b}	19.2^{b}	2.9^{a}	14.9^{b}	16.3^{a}	3.8^{b}
Diverse FV	17.7^{c}	78.6^{c}	13.2^{c}	29.3^{c}	26.0^{b}	74.8^{c}	53.5^{b}	44.0^{c}

One-way ANOVAs with Tukey Post-hoc comparisons where $p < 0.01$ are considered statistically significant. Values with the same superscript letters were not statistically different. N = 2,367 rural households.

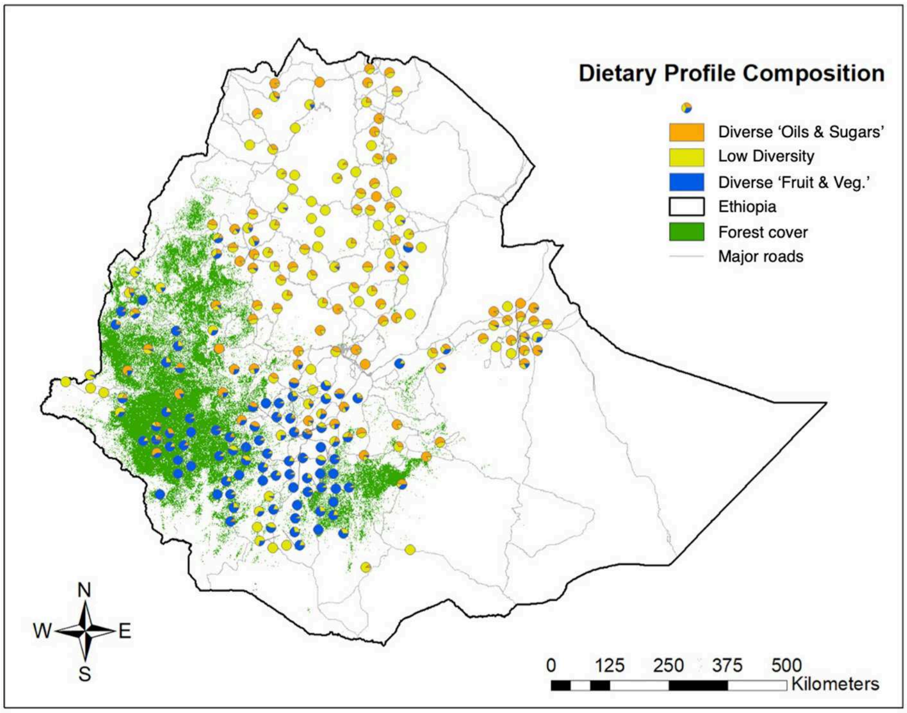

FIGURE 2 | The geographic distribution of the three household dietary profiles across Ethiopia. The World Bank's LSMS used an enumerator area (mostly corresponding to a village) as the geographical sampling unit. The pie-chart represents the percentage of households following each dietary profile within each enumerator area.

Identification of Farm System Types

Ethiopia is characterized by large gradients in elevation and precipitation, which have, in part, contributed to the rise of many different farming systems (Amede et al., 2017). Using a clustering approach we identified 7 farming system types based on the relative production of crops grown and livestock species owned as well as annual mean precipitation and elevation (**Figure 3**, **Table 3**, as well as **Figure S2** and **Table S2**). These corresponded well with, though were coarser than, the 16 farming systems identified in Ethiopia by Amede et al. (2017). Two of these farming system types were focused on low to mid-elevation sorghum production with an average crop diversity of 4.41 and 4.26 per farm; two were based on low to mid-elevation coffee agroforestry and had the highest crop diversity with 7.10 and

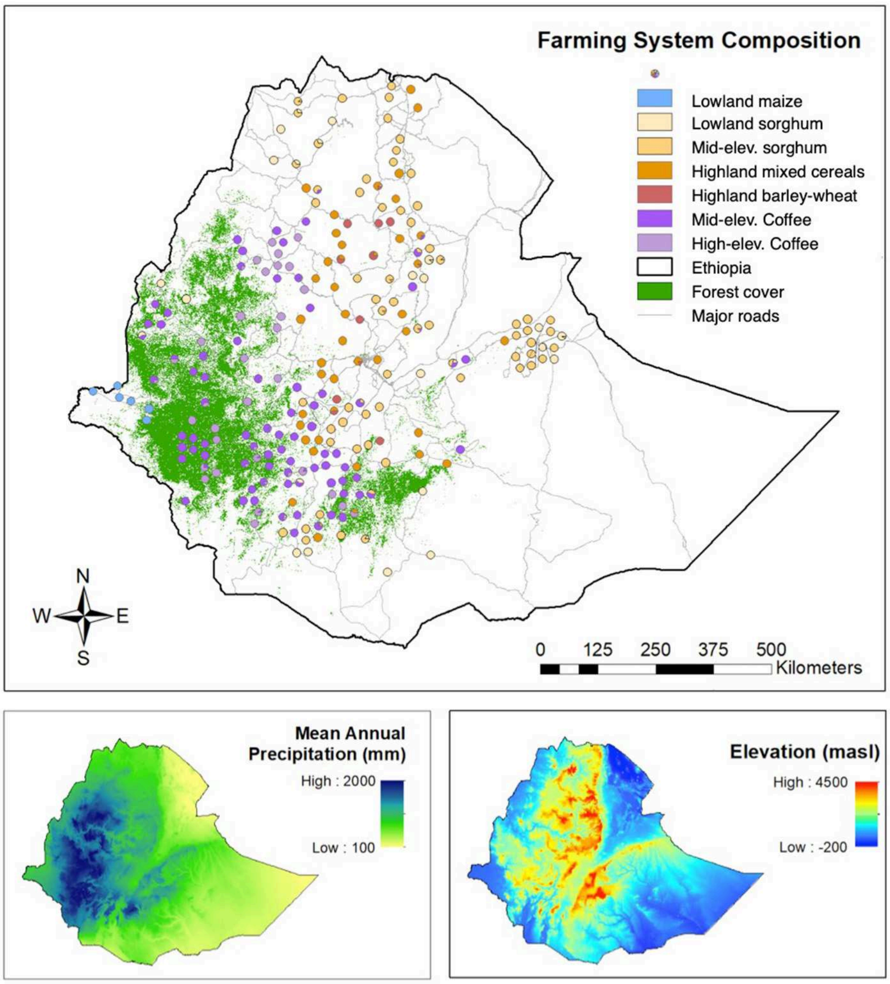

FIGURE 3 | Geographic distribution of the seven farming system types across Ethiopia. The World Bank's LSMS used an enumerator area (mostly corresponding to a village) as the geographical sampling unit. The pie-chart represents the percentage of households engaged in each farming system type within each enumerator area. The bottom left panel shows the mean annual precipitation (mm) across the country, while the bottom right panel shows the elevation (masl).

TABLE 3 | Distribution (%) of household dietary profiles across 7 different farming system types.

Dietary profile	Farming system type						
	Lowland maize	Lowland sorghum	Mid-elev. sorghum	Low elev. coffee	Mid elev. coffee /cereal	Highland diverse cereal	Highland barley /wheat
Diverse OS	0.3	8.5	33.6	20.2	13.9	20.0	3.4
Low diversity	4.4	11.4	29.5	17.3	9.1	21.6	6.7
Diverse FV	1.1	4.1	12.3	56.6	17.6	8.4	NA

6.88 crops per farm; two were characterized by high elevation mixed cereal production with an average of 4.43 and 6.0 crops per farm; and one was based on lowland maize with an average of 2.35 crops per farm. All farming system types had on average between 2.2 and 3.6 different types of livestock on farm with the Lowland maize type having the fewest and highland cereal producers having the most.

Factors Associated With Dietary Profiles

Farm System Type

There is a significant association between households' dietary profiles and the types of farming they pursue ($X^2 = 523.43$, df $= 12$, $p < 0.001$). Households with the *Diverse FV* dietary profile are predominantly coffee-agroforestry farmers (74%), with a small percentage pursuing mid-elevation sorghum production (12%) (**Table 3**). The *Low Diversity* diet households also pursue coffee agroforestry (26%), albeit to a much lesser extent. Rather, they are more engaged in low and mid-elevation sorghum production (41%) and high-elevation diverse cereal production (22%). The lowland maize as well as highland barley-wheat system types are almost primarily represented by households with the *Low Diversity* dietary profile. Finally, households with the *Diverse OS* dietary profile are stratified across sorghum, coffee and diverse cereal production systems.

Wealth

We found a significant relationship between households' dietary profile and their asset-based wealth class ($X^2 = 204.78$, df $= 4$, $p < 0.001$). Given that we had uneven numbers of households per wealth class due to our chosen classification of wealth, we looked at the relative percentage of households per wealth class following a particular dietary profile (**Table 4**). We found that while 49% of the households in the bottom wealth class followed the *Low Diversity* diet, only 14% followed a *Diverse OS* diet. In contrast, of the households in the top wealth class, 45% followed a *Diverse OS* diet and 35% a *Diverse FV* diet, with only 20% consuming a *Low Diversity* diet. Households in the middle wealth class were equally likely to follow any of the three dietary profiles. Together these results suggest that high-income households are more likely to consume higher diversity diets, while lower diversity diets are more often consumed by households in the bottom wealth group. An important caveat is that, 37% of households in the bottom wealth class followed a *Diverse FV* diet, suggesting that this dietary profile was not exclusive to wealthier households.

TABLE 4 | Percentage of households across wealth classes following each identified dietary profile.

Dietary profiles	Wealth class		
	Bottom%	Middle%	Top%
Diverse OS	14	30	45
Low diversity	49	37	20
Diverse FV	37	33	35
Total	100	100	100

Landscape Context

Finally, we explored how the landscape elements surrounding a household were associated with their dietary profile using one-way ANOVAs (**Table 5**). We found that households with a *Diverse FV* dietary profile had a significantly higher proportion of forest (35%) in a 10 km radius circle surrounding their villages than households with the *Diverse OS* or *Low Diversity* dietary profile, which both had <10% forest cover. Also, the *Diverse FV* dietary profile was generally found in areas of higher mean annual precipitation and lower elevation as compared to the other two dietary profiles. These trends are consistent with the finding that many households with this dietary profile were involved in coffee-agroforestry farming systems, where coffee was grown in tropical moist mountain forests (1,000–2,000 masl). Interestingly, we also found that households following a *Diverse OS* diet tended to have (a) significantly shorter distances to major roads than the other households, and b) the average shortest distance to major population centers (i.e., markets). These observations suggest that the availability of processed foods might be higher for these households. Moreover, households following a *Low Diversity* diet were found furthest from roads and major population centers (i.e., they were likely more disconnected from markets), which might also partly explain the lower wealth associated with this group.

In order to tease apart whether forest cover has an impact on diet, independent of forest-based farming systems, we looked at those households in cereal-based farming systems only—i.e., we excluded households engaged in agro-forestry farming systems where the cultivated trees on farm might be counted as forest. Using a linear mixed model, with the enumerator areas as our random effect, we found no significant association

TABLE 5 | Landscape variables expected to influence dietary profiles.

Dietary profile	Precipitation (mm)	Temp. (°C)	Elevation (m)	Forest cover (%)	Distance to roads (km)	Distance to population center (km)
Diverse OS	$1,119^a$	18.1^a	2051^a	9.4^a	11.5^a	32.1^a
Low diversity	$1,086^a$	18.8^b	1952^b	7.6^a	16.5^b	39.2^b
Diverse FV	$1,354^b$	19.2^b	1821^c	34.9^b	15.6^b	35.4^a

Mean values and one-way ANOVA: Tukey Post-hoc comparisons with p < 0.01 considered statistically significant. Mean values with the same superscript letters were not statistically different.

between forest cover and MHHDDS ($p = 0.86$), controlling for significant covariates (household size, age and gender of the household head, and wealth class, all $p < 0.05$, **Table S3**). When we used households' consumption of individual food groups as our binary response variable, we found a positive relationship between the proportion of forest in the landscape and a household's consumption of dark green leafy vegetables ($p < 0.005$), and "other fruits" ($p < 0.05$), controlling for the covariates shown to influence dietary patterns (see **Tables S4A–F** for model results). The estimate sizes from these models suggest that for each additional percentage of forest in the 10 km radius circle surrounding a household, the odds of consuming dark green leafy vegetables increases by 3.8%, while the increase was 2.2% for other fruits. We also found a negative relationship between forest cover and the consumption of vitamin A-rich vegetables and fruits ($p < 0.1$) which would result in a 2.2% decrease in the odds of consuming this food group with one additional percentage forest cover. Testing the influence of distance to nearest major road with the full dataset of households, distance was not a significant predictor of the MHHDDS (**Table S5**), nor was it related to the probability of consuming "oils and fats" or sweets (**Table S6**). Consumption of these foods were only significantly predicted by household wealth ($p < 0.001$, **Tables S6A,B**) and age in the case of sweets ($p < 0.05$, **Table S6B**).

How Farming System Type, Wealth, and Landscape Context Shape Diets

To visualize these identified relations we plotted an alluvial flow diagram (**Figure 4**) tracing the number of households from the three wealth classes (bottom, middle, and top wealth) that followed each of the three identified dietary profiles. The figure also traces the breakdown of dietary profiles across the seven farming system types.

Looking across the wealth class and farming systems, we see that although the *Low Diversity* dietary profile was made up of streamlines emanating mostly from the bottom and middle wealth classes (left-hand side of figure), it was not restricted to a particular farming system type (right hand side of figure). In other words, the *Low Diversity* profile seems to reflect the impact of poverty on diet rather than the farming system in place. In contrast, looking at the *Diverse FV* dietary profile, we see on the left-hand side that the streamlines are proportional (∼30% of households) across the three wealth classes, while on the right-hand side of the panel, the streamlines from this diet flow primarily to low- and mid-elevation coffee systems. This might suggest that this dietary profile, high in fruits and vegetables is reflective more of the farming system type than the wealth of households. Finally, when looking at the *Diverse OS* diet profile the streamline coming from the top wealth class was relatively large as compared to the two other dietary profiles, and the streamlines to farming systems were more equally sized. This suggests that households pursuing this dietary profile were engaged in most types of farming systems, indicating that the diet was driven by wealth.

DISCUSSION

To our knowledge, no previous studies have combined comprehensive assessments of dietary profiles and farming system types to identify how household characteristics, market access, on- farm production, and the landscape context combine to shape diets. We found that both household wealth and farming system type were associated with rural households' dietary patterns. Specifically, a low diversity diet was strongly associated with households in the bottom and middle wealth classes, while households with a diverse diet profile also rich in oils, fats and sugars primarily came from the middle and top

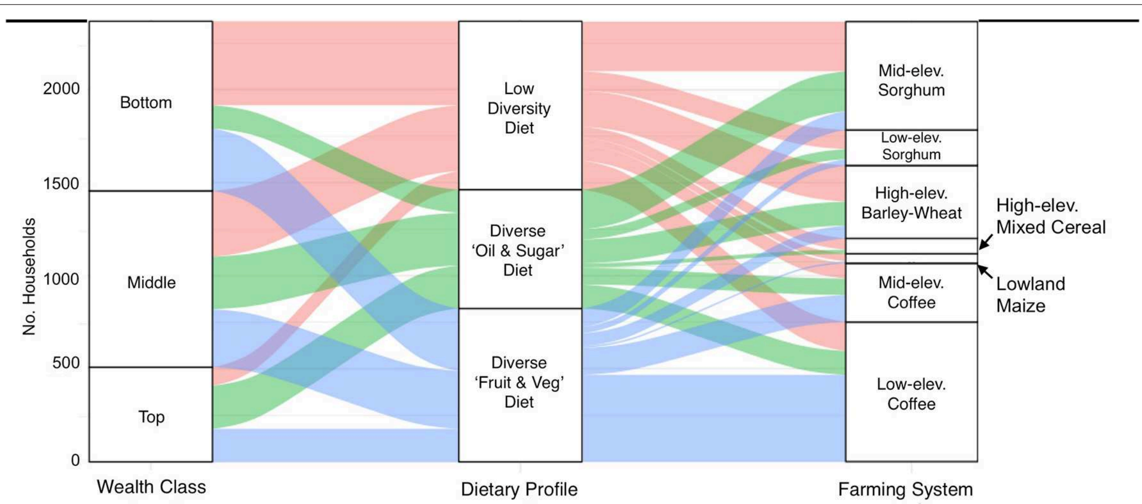

FIGURE 4 | Alluvial flow diagram showing the (left-hand side) breakdown in the numbers of households from the bottom, middle, and top wealth classes with a Low Diversity (pink), Diverse OS (green) and Diverse FV (blue) diet profile, and (right-hand side) the breakdown in the number of households from different farming systems in each diet profile.

wealth classes. By contrast, the pursuit of a diverse diet high in fruit and vegetables did not appear to be limited to any particular wealth class, and was primarily embraced by households with coffee-agroforestry farming systems.

Disentangling Wealth, Farm Systems, and Forest Cover

Our finding that approximately half of the households in the top wealth group consume a diverse diet also rich in fats, sugars and processed foods, is potentially suggestive of households in a "nutritional transition." A nutritional transition is a phenomenon where increased economic development leads to the gradual replacement of traditional diets high in fiber and local crops by foods more reflective of a Western diet, such as processed foods high in oil and refined sugars (Popkin, 1993; Cockx et al., 2018). This transition can lead to overweight and obesity, with significant implications for health and the rates of non-communicable disease such as diabetes and hypertension (Popkin, 2001). Previous studies from Ethiopia have indicated a nutritional transition by showing (a) that poor diets and increasing rates of obesity are common in urban households (Amare et al., 2012; Tebekaw et al., 2014), and (b) a rise in the amount of sugar and foods cooked in oil over the past decade, though also an increase in the consumption of fruits and vegetables (Aurino et al., 2017).

In our study, although households with a *Diverse OS* diet live on average closer to roads (and by extension markets), we found that household wealth level, rather than distance predicted consumption of oil and sugar. When we look at the spatial distribution of households consuming the *Diverse OS* diet, we find that this consumption pattern often occurs alongside households engaged in the same farming systems, but who follow the *Low Diversity* diet indicating that wealth may be the key determining factor of diet in these landscapes. However, isolating the role of wealth from market access with these types of datasets is tricky. Many of the assets used to create the wealth classes depend on having access to products and materials from the market. Interestingly, in a small subset of villages we see that the *Diverse OS* and the *Diverse FV* dietary profiles co-exist within villages and farming systems, which are not linked to wealth. Thus, additional factors need to be considered in order to explain the drivers of dietary patterns in these situations. It is possible that social factors such as cultural or ethnic group (Labadarios et al., 2011), may help to explain why certain of the households adopt the *Diverse OS* diet, while others do not although they live and farm in the same settings. In summary, these findings advance the argument that nutritional outcomes do not necessarily improve with higher wealth, although greater income is clearly beneficial for households (Herforth and Ahmed, 2015).

We also find that the farming system type is strongly associated with diets. This is not surprising given that on average over the year, 58% of the calories consumed in farming households in Ethiopia come from on-farm production (Sibhatu and Qaim, 2018). To date, most studies examining the relationship between farming practices and diet have focused on simple measures of on-farm diversity, such as counts of crop and livestock species to explain diet composition. Such studies have found that increasing household production diversity on Ethiopian farms improves children's dietary diversity (Hirvonen and Hoddinott, 2017). Interestingly little attention has been placed on how the farming system type can influence and help to explain rural diets. Farming systems categorization typically describes farms according to the resource base, land management, and off-farm strategies (Tittonell et al., 2010), and may lend insight into the orientation of the farm (commercial vs. subsistence) and household dependence on certain crops. We find that the diverse dietary profile high in fruit and vegetables, *Diverse FV*, is primarily composed of households with coffee-agroforestry farming systems. Coffee-agroforestry is both the most crop diverse of the farming systems identified, and likely the most strongly market-oriented due to the focus around a cash-crop thereby providing cash income to farmers to purchase market foods. Thus, these farmers may benefit from a confluence of factors encouraging a diversified diet. Emerging research evidence from Ethiopia also suggests that households located closer to markets enjoy better diets (Stifel and Minten, 2017), and their food consumption is less dependent on their own agricultural production (Hoddinott et al., 2015; Hirvonen and Hoddinott, 2017).

Other studies examining the potential factors that might influence diet diversity have also found a positive relationship between dietary diversity and forest cover across Africa (Johnson et al., 2013; Ickowitz et al., 2014; Galway et al., 2018; Rasmussen et al., 2019), however the mechanisms underlying this relationship remain poorly resolved. Here we lend insight into this relationship by showing that diverse diets high in fruits and vegetables in Ethiopia are strongly associated with coffee-agroforestry farming, a system with high tree cover. In the other cereal-based farming systems, forest cover is also positively associated with the consumption of dark green leafy vegetables and fruits, suggesting that trees outside of farming systems still positively contribute to diets. Yet, understanding why diets are more diverse in these systems, whether through the direct consumption of forest foods, the sale of forest products, or higher on-farm crop diversity requires deeper analysis into whether the food found on household plates are sourced from the forest. Unfortunately, such an analysis is not feasible with the LSMS data.

Benefits of Assessing Diets at the Food Item Level

One of the most striking results from our study is that those households who consume a diversified diet, as per the methods used to calculate MHHDDS, do so in two very different ways. In the case of the *Diverse FV* profile, more households consume dark green leafy vegetables and "other fruit"; meanwhile households with a *Diverse OS* diet include pulses, eggs, vitamin A-rich foods and other vegetables in their diets, as well as oils, fats, and sugars which are not included in the calculation of the MHHDDS. This is only apparent when we look at the food items composing diets, as both diets result in a score of ∼5.5, which we classify as "diverse" [although this value is low when compared to recent research from Malawi (Jones, 2016)]. These diets may have different nutrition and/or health related outcomes, however we were not able to test for this given data limitations. Although not considered in this study, looking at the number of food items within a food group that a household consumes may also provide insight into how robust or resilient the diet is to external stressors such as market or crop vagaries.

Knowledge gaps related to the factors explaining different dietary outcomes, not only arise from the sheer challenge of disentangling the influence of wealth, farming system type, and landscape context, but also from the relatively simplistic metrics used to assess dietary quality. While scholars have called for indicators of dietary quality that consider multiple dimensions to provide comprehensive assessments (Jones, 2017), the MHHDDS—which is widely applied to consumption surveys—is considered a common and useful measure to assess dietary diversity. Our joint use of MHHDDS and dietary profiles reveals, however, that eating a relatively diverse diet (high levels of MHHDDS) might also be associated with eating a greater variety of unhealthy food items. One reason for why scholars have focused relatively little on the composition of food items that people consume lies simply in the extensive nature of collecting food consumption data at the food item rather than the food group level. Yet, our results point to the need for a two-pronged approach—for example combining dietary diversity scores with additional information on household consumption of foods indicative of a nutrition transition (e.g., increased oils, sugars, processed foods)—to ensure that complementary dimensions of diet quality and diversity are included in dietary assessments. This seems advisable because dietary recommendations based solely on "eating a variety of foods" might fail to attend to whether people simultaneously consume both healthy food groups and less healthy food items.

While other work has found that dietary diversity can vary strongly with season in Ethiopia (Hoddinott et al., 2015; Abay and Hirvonen, 2017; Sibhatu and Qaim, 2018), our dietary profiles do not capture this variation. Almost all of the LSMS in Ethiopia were collected between December 2015 and February 2016, which coincides with the post-harvest season (Central Statistical Agency of Ethiopia, 2016). It is thus possible that our dietary profiles do not reflect consumption patterns common in other seasons, particularly the lean season in which wild foods may become more important (Cruz-Garcia and Price, 2014). In summary, future work in this field would benefit from stratified data collection efforts that (a) are carried out across seasons and (b) record the food provenance.

CONCLUSIONS

If we are to encourage policies to improve dietary quality, we also need a more sophisticated understanding of the factors that enable healthy diverse diets as compared to the factors that lead to either consumption of less healthy foods or low dietary diversity. This study shows, by drawing on a number of different datasets, how wealth, farming system type, and landscape context interact and influence the diets of rural households in Ethiopia. In particular, we found that a low diversity diet was strongly associated with households in the bottom and middle wealth classes that were primarily involved in cereal-based farming, although not exclusively. By contrast, our work shows that access to forests and coffee-agroforestry cultivation systems in Ethiopia are associated with diverse diets based on healthy food items, particularly vegetables and fruits. Given that the EAT-Lancet commission recently stated that the global consumption of fruits and vegetables (and nuts and legumes) will have to double to achieve health and environmental benefits (Willett et al., 2019), it is worrying that no attention is given to the role of forests in securing sufficient supply of these food groups. In terms of policy recommendations it is, however, neither feasible nor desirable for many cereal farmers to shift to coffee-agroforestry simply to achieve a higher quality diet. But our findings suggest that even small amounts of forest cover can contribute to diverse diets. Under the Bonn Challenge, Ethiopia has committed to forest landscape restoration across 15 million hectares of land [http://www.bonnchallenge.org/content/Ethiopia]; well-planned and implemented, such forest restoration can contribute to increasing food security in addition to other goals. Moreover, our findings show that households with a diverse diet profile also rich in oils, fats, and sugars primarily come from the middle and top wealth classes, which suggests that we should be cautious about the expectation that improved wealth leads to better nutritional outcomes. In summary, our work suggests that encouraging forest protection and restoration can help to reconcile goals of environmental protection and food security in tropical countries facing these combined pressures.

DATA AVAILABILITY STATEMENT

Publicly available datasets were analyzed in this study. This data can be found here: https://microdata.worldbank.org/index.php/catalog.

AUTHOR CONTRIBUTIONS

LR and SW led the design of the study and the analysis of the data. All authors contributed to data interpretation and writing of the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2020.00045/full#supplementary-material

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