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Original citation & hyperlink:
Chappell, MJ, Moore, J & Heckelman, A 2016, 'Participation in a city food security program may be linked to higher ant alpha- and beta-diversity: An exploratory case from Belo Horizonte, Brazil' *Agroecology and Sustainable Food Systems*, pp. 804-829
https://dx.doi.org/10.1080/21683565.2016.1160020

DOI 10.1080/21683565.2016.1160020
ISSN 2168-3565
ESSN 2168-3573

Publisher: Taylor and Francis

This is an Accepted Manuscript of an article published by Taylor & Francis in Agroecology and Sustainable Food Systems on 3rd March 2016, available online: http://www.tandfonline.com/10.1080/21683565.2016.1160020

This is an Accepted Manuscript of an article published by Taylor & Francis in Agroecology & Sustainable Food Systems *on 03 March 2016, available online:* [**http://www.tandfonline.com/**10.1080/21683565.2016.1160020](http://www.tandfonline.com/10.1080/21683565.2016.1160020).

Participation in a city food security program may be linked to higher ant α- and β-diversity: An exploratory case from Belo Horizonte, Brazil

ABSTRACT

This paper reports the results of a case study examining the connections between municipal food security policy and biodiversity in the region of Belo Horizonte, a populous city in the heavily fragmented Brazilian *cerrado* (savannah)/Atlantic forest transition region. Belo Horizonte, through its Secretariat of Food and Nutrition Security (SMASAN), has generated increased food security in the city, in part by economically supporting local small farmers. Farmers' economic security has been previously linked to their agricultural practices and sustainability; thus SMASAN's programs potentially affect biodiversity in the region's agricultural matrix and rainforest fragments through their work with farmers. In order to examine this dynamic, we compared ground-foraging ant diversity on four "SMASAN" and three "non-SMASAN" farms and adjoining forest fragments. Supported by data from farmer interviews, sampling in 2005 and 2006 indicated SMASAN farms had: (a) higher alpha and beta diversity; and (b) potentially greater overlap between species found on-farm and in adjacent forest fragments. This case study may be the first directly linking biodiversity conservation with food security and changes in local food policy institutions, emphasizing the importance of an approach integrating politics and ecology, and the potential for human well-being and conservation to go hand-in-hand.

Keywords: Agriculture, ants (Formicidae), Atlantic forest, biodiversity conservation, Brazil, food security, landscape ecology, political ecology, rural-urban linkages

INTRODUCTION

With 40% earth's land surface under agriculture and a majority of the world's organisms existing outside of protected natural areas, and considering the key role agriculture plays in threatening biodiversity, it is clear that two of the most pressing problems facing us today—rapid biodiversity loss and the food insecurity and malnutrition facing as many as 1 billion people in the world—are inextricably linked (Tscharntke et al., 2012). Specifically, it has been well established that what happens in the matrix—the areas surrounding "natural" habitat fragments, such as farms and pastures situated around fragmented forest areas—strongly influences the ecology within such fragments (Perfecto et al., 2009; Mendenhall et al., 2014). A high quality matrix—i.e., agricultural land managed such that it is more similar to the native ecosystem—may very well function in the way that habitat corridors were expected to function, decreasing patch isolation and potentially leading to higher levels of biodiversity in both the native habitat fragments and in the agricultural system itself (Ricketts, 2001; Perfecto and Vandermeer, 2008; Melo et al., 2013). Further, existing research provides strong evidence that farmers' socioeconomic resources and well-being are important predictors of their use and uptake of various agroecological/sustainable/conservation practices (Upadhyay et al., 2003; Marshall, 2009; Baumgart-Getz et al., 2012).

Given the possibility of creating high quality matrices on agricultural land, a significant body of research has developed around assessing the relative biodiversity conservation and production value of agroecological, (and related) practices as compared to high-input "conventional" agricultural approaches. A particular recent focus has been the so-called "land-sparing/land-sharing" debate, which seeks to identify direct trade-offs between agricultural productivity per unit area and biodiversity (Phalan et al. 2014). The debate in the literature has tended to revolve around terms set out by specific early works in this area (e.g. Balmford et al. 2005) which typically implicitly or explicitly conflate food security (access by all people in a society at all times to enough culturally and nutritionally appropriate food for a healthy and active lifestyle) with productivity. That is, many sparing/sharing studies have equated greater per unit area agricultural productivity with greater food security. This is intuitive, but in fact this relationship is empirically weak in contemporary systems, as most areas of the world suffering from food insecurity already have access to sufficient calories and see limited, if any, improvement merely from increased productivity (Sen 1981; Smith et al. 2000; Smith and Haddad 2015). The debate around this and other points is still heavily contested on empirical, theoretical, and epistemological grounds (e.g. Fischer et al. 2013), but the focus has overwhelmingly been on potential tensions between food security, different agricultural methods, and biodiversity (Balmford et al. 2005; Phalan et al. 2014) or alternatively, the possible positive effects of biodiversity on food security and livelihoods (e.g. Remans et al. 2010; Chappell et al. 2013).

The current study examines the same nexus of relationships from a somewhat "inverse" perspective that has rarely been examined: can increased food security support biodiversity? Specifically, the work presented in this paper forms one component of a larger project examining the food and agricultural system of the Brazilian city of Belo Horizonte and its surrounding landscape. Belo Horizonte founded a Municipal Secretariat of Food Security (the *Secretaria Municipal Adjunta de Segurança Alimentar e Nutricional*, known by its Brazilian acronym, SMASAN) in 1993, which has since been recognized for fostering dramatic improvements in food security within the city (Rocha and Lessa 2009; World Future Council 2009). One of

SMASAN's flagship initiatives has been its Straight from the Countryside (*Direto da Roça*) program, where small (<50 ha, though most are <10 ha), local family farmers are selected through a public process and provided with low-cost access to produce stand locations in high-traffic areas of the city (Rocha and Lessa 2009; Chappell, forthcoming). Through the efforts of this program, farmers and urban consumers appear to be sharing the economic benefits of avoiding intermediary sellers, who farmers and city officials report as charging up to a 100% mark-up (*authors' interviews*). These local farmers are, in turn, situated in a highly fragmented tropical landscape and biodiversity hotspot. Thus through the SMASAN programs generally, and the Straight from the Countryside program specifically, food security in Belo Horizonte is connected to the condition of biodiversity in the region's agricultural matrix and rainforest fragments, mediated by the practices of the farmers participating in the program. We sought to test if SMASAN's documented positive effects on food security may in fact have been connected to positive effects on local biodiversity.

Study System

Belo Horizonte, the capital of the Brazilian state of Minas Gerais, has approximately 2.5 million residents and is situated in the "mega-biodiverse" Atlantic forest/Brazilian Savannah (*cerrado*) transition region in southeastern Brazil (Figure 1). The Atlantic forest is widely described as being 90% deforested (Dean 1995), though this may be an overestimate, with small but ecologically significant fragments being overlooked (Vandermeer and Perfecto 2007; Decocq et al. 2016). Interviews with farmers, city officials, and local extension agents indicate that mining, expanding urban borders, and expanding agricultural land present the greatest threats of on-going deforestation, though recent evidence from at least one municipality in the area indicates that agriculture has not been a significant contributor of changes in forest cover in recent years (Oldekop et al. 2015).

----*Figure 1 about here*----

The state of Minas Gerais is economically dependent on ore mining, with mining activities increasing over the past two decades, both state-wide and in the greater Belo Horizonte landscape (IBGE 2013; authors' interviews). In the studied agricultural landscape, approximately 40 km SW of Belo Horizonte, agricultural production is almost exclusively horticultural, focusing particularly on leafy vegetables. Most farmers in the region appear to produce almost exclusively for commercial sale rather than for subsistence, and livestock and production of other cash crops at any significant scale are uncommon (*pers. obs.*; *authors' interviews*). Farmer interviews indicated that low prices for their products (especially from intermediary sellers), expanding urban borders/suburbanization, mining, and labor shortages represented the largest threats to their well-being, which corresponds with the recent account by Oldekop et al. (2015).

Background on SMASAN and Straight from the Countryside

Belo Horizonte's government made access to food a right of citizenship, creating the Secretariat of Food and Nutrition Security (SMASAN) in 1993 in order to guarantee this right. SMASAN has presided over unprecedented successes in enhancing food security, such as reductions in infant mortality and malnutrition by more than 50% since 1993 (Aranha, 2000; Alves et al., 2008). SMASAN's programs also connect it with local, small family farmers in the surrounding Atlantic Rainforest. The goal of programs connecting with local farmers, such as Straight from the Countryside, is to improve farmer incomes and well-being while offering consumers lower prices for high-quality produce. The programs also aspire to thus slow regional rural-urban migration that puts additional strain on city services (Rocha et al. 2012), although at least with regards to Straight from the Countryside, which enrolls between 15 and 60 farmers/year, such a result is purely aspirational.[1] Nevertheless, given the links between farmers' socioeconomic resources and well-being and their use of agroecological practices, and farms' influence on landscape biodiversity, as mentioned above, the study system represents a possible example where increased food security may be affecting farmer practices, and thus, positively affecting biodiversity conservation in the local landscape.

As was stated, the work presented here is part of a larger project examining the political ecology of the formation and persistence of SMASAN's policies, including its effects on farmers and biodiversity in Belo Horizonte and its surrounding landscape. The social aspects of the project took a mixed methods approach and was conducted roughly along the lines of Geertz's (1993) concept of "thick descriptions." We used a combination of formal interviews, examination of documentary evidence, participant observation with members of SMASAN's staff and management, and cultural immersion and interactions with SMASAN-partnered and non-SMASAN area farmers in order to understand the qualitative "webs of significance" spun around SMASAN and Belo Horizonte, in search of deeper causal links found beneath the perceptions and appearances of SMASAN and its partnerships.[2]

One part of the social elements of our larger project sought to find the effects of SMASAN partnerships on farmers' incomes, well-being, and farming practices. SMASAN farmers were solicited from a list (provided by SMASAN) of 20 farmers who had been participants of Straight from the Countryside the previous year. After getting zero positive responses to requests for participation, we took the tactic (suggested by SMASAN) of un-announced site visits, which were treated far more positively by area farmers than attempts to schedule appointments at their produce stands or by phone. However, as a result of the difficulties in this process, only three SMASAN farmers (one of whom owned two sites) were interviewed. (Three additional SMASAN farmers declined.) Ants were sampled at all four of these SMASAN sites. Using snowball sampling (asking SMASAN interviewees for suggestions of neighboring or local farmers with similar backgrounds and farm production), a total of ten non-SMASAN farmers were interviewed (with two additional farmers declining). The thirteen farms represented approximately 8% of farming households in the area, according to Brazilian census data. Based on data provided by SMASAN, the three farmers interviewed represent approximately 16% of the farmers in Straight from the Countryside in 2005.

In terms of recruiting for Straight from the Countryside, SMASAN works with local governments and extension agents to solicit interested farmers. Farmers responding to the solicitation are informed about the quality and safety standards required by the program (basic practices of safe and proper storage, handling, sanitation, and use of agricultural chemicals), and a series of visits are arranged for the state extension agent assigned to SMASAN to inspect farms for compliance. Although established partner farmers nominally get precedence during selection, in practice, there are more than sufficient spaces to accommodate qualifying farmers, with interviews indicating that the barriers to larger number of farmers participating being primarily (1) insufficient dissemination of information about the programs to area farmers (a theme that nearly every farmer emphasized); (2) challenges for farmers in meeting the basic standards of the programs; and (3) arranging transportation and staffing for produce stands, which imposes possible additional demands in terms of costs and labor, although farmers are encouraged to join cooperatives so that they can share these and lighten the load on each farmer.

Once they are part of the Straight from the Countryside program, farmers are visited by SMASAN's extension agent at least once a year as condition of the program, to confirm continued compliance with SMASAN's standards for quality and safety. (For example, while SMASAN cannot ban the use of synthetic pesticides, use of what the extensionist deems an excessive amount is not permitted.) This system means that the extensionist becomes the primary point of contact between the Belo Horizonte government and the farmers. This may be particularly relevant as the current extension agent and his predecessor have both been enthusiastic proponents of organic agriculture and agroecology, offering technical advice and vocal support for using less synthetic inputs and more agroecological methods to the farmers (*pers. obs.*).

As we will return to in our discussion, this relationship with extension agents may be an important element of the studied dynamics. Part of the overall study's hypothesis was that association with SMASAN may have altered farmer practices. However, our interviews were not able to recover the anticipated level of detail on the farmers' practices. The responses that were obtained did not indicate any systematic differences between SMASAN and non-SMASAN farms, with some SMASAN farms using (legally allowable) synthetic pesticides and fertilizers, for example, and some non-SMASAN farmers reporting that they were essentially uncertified organic producers (Chappell, forthcoming).

Ants as bioindicators

Ants were used in this study to gauge effects on landscape biodiversity. The diversity and richness of arthropod groups has in the past been shown to be reasonable indicators for general biodiversity and changes in agroecological habitat (Alonso and Agosti 2000, Vandermeer et al. 2002). Ants, specifically, are a classic bioindicator with a long history as indicator species for diversity in agroecological matrices and for documenting differences between farm management systems (Peck et al., 1998, Agosti et al., 2000, Leslie et al., 2007) and can show strong correlations to diversity at other levels (Armbrecht et al., 2004). Further, ants play a number of different ecological roles including interactions at multiple trophic levels, are ubiquitous, extremely diverse, and highly studied, and their sensitivity to environmental changes can help indicate ecosystem health (Alonso and Agosti 2000).

Additionally, pairing indicator species data with data on land use and agricultural practices improves the ability to make inferences about a landscape's ability to support biodiversity more broadly, rather than only being able to speak to the patterns of the indicator species (Billeter et al. 2008). Thus, based on our interviews, if we saw consistent differences in farmer practices between SMASAN and non-SMASAN farms, we should be able to combine those to make a stronger inference about matrix quality than would be possible with ant sampling alone. Nevertheless, a single taxon cannot stand in for all biodiversity (Lawton et al. 1998), meaning that any results from this study must be considered as a very provisional assessment of biodiversity and matrix quality in the studied system.

METHODS

In 2005 and 2006, the first author interviewed SMASAN staffers and SMASAN and non-SMASAN farmers, and examined the potential effects of SMASAN participation on ground-foraging ant diversity on farm fields and adjacent forest fragments (Table 1). All farms were located less than 40 km to the SW of Belo Horizonte (19° 55' 0" S, 43° 56' 0" W) with the farthest distance between farms being under 10 km (see Figure 2; specific locations are not given in order to maintain producer confidentiality). Farm production area ranged from 1-5 ha. All were primarily vegetable farms, with lettuce varieties predominating.

----*Figure 2 about here*----

----*Table 1 about here*----

SMASAN farmers had spent approximately eight to eleven years working with the program. Farms were chosen by the willingness of farmers to participate, but all farms were similar in size (with the exception of SEDD, which was excluded from parts of our analysis as an outlier; see below). Sampling was conducted using tuna baits in eleven locations on seven farms (four SMASAN partners; three non-SMASAN). Samples were collected between February and April, corresponding to the transition between the "Rainy" and "Dry" seasons. The seven farms were owned by: 1) Dona Marta (two farms, DM and DM2); 2) Seu Ricardo (SR); 3) Seu Edmar and Dona Diana (SEDD); 4) Seu Henri (SH); 5) Os Santos (OS); and 6) Seu Herbert (SHB). DM, DM2, SR and SEDD were "SMASAN" farms; SH, OS, and SHB were not. (Farmers' names have been changed to preserve confidentiality.) All farms lie between 730-840 m in elevation and receive approximately 1500 mm of rainfall a year (Instituto Nacional de Metereologia (INMET) 2008). At the time of this study, all farmers in the Atlantic Rainforest region were required to keep 20% of their land set aside to preserve extant rainforest fragments, although there were no fragments present on two farms (SEDD and DM). Fragments of the Atlantic Rainforest on farmers' properties can be generally characterized as established secondary, closed-canopy forest, such that understory growth and light gaps are relatively rare in the interior of the fragments.

<u>Data Collection</u>

At each farm, samples were collected within an inactive plot in the farm field and, where present, in the interior of an adjacent forest fragment, using a grid of 50 tuna baits to attract ants (5 rows X 10 columns, 2 m separation between each bait). Where forest fragments were present, baits began 25-50 m from the forest edge. Tuna baiting was selected as it is a common method for quick surveying of ground-foraging ant communities (Agosti et al. 2000, Philpott et al. 2004). Each bait of 1-5 g of canned tuna was placed directly on the soil after clearing leaf litter or other debris. After waiting approximately 15-20 min, each bait was surveyed for the presence of ants, and voucher specimens of each species present were aspirated and placed into a vial containing 75% ethanol for later identification. (Due to missing baits and other circumstances, some sites ended up with a total of less than 50 baits collected.) In 2005, only four farms were sampled, two participating in SMASAN (DM and SR) and two non-participants (SH and OS). In 2006, all previous sites were re-sampled, and three sites were added: two SMASAN (DM2 and SEDD), and one non-SMASAN (SHB).

All collections were identified to species or morphospecies in laboratory. EstimateS (Colwell 2005) was used to produce resampling-based rarefaction curves and extrapolate diversity measures for appropriate comparisons. Voucher specimens were deposited at the Laboratory of Myrmecology, Center for Cacao Research of the Executive Planning Commission for Cacao Farming (CEPEC/CEPLAC), Itabuna, Brazil.

<u>Data Analysis</u>

Species richness can be characterized in terms of alpha diversity—the total number of species in a given site—as well as evenness, guild (or functional group) diversity, guild (or functional group) evenness, and beta diversity (the turnover in species identity from site to site or time period to time period). With regards to alpha diversity, we used the EstimateS's Incidence-Based diversity metric (ICE) to measure species richness (simple number of species); the Shannon diversity index (H), which incorporates both species richness and evenness; and Pielou's evenness (E). (Guild assignments were based on Andersen 2000, and Brown 2000.) Values for species evenness (E_{spp}) were derived from the Shannon indexes (H_{spp}) calculated by

EstimateS; guild evenness (E_{fx}) was derived from manually calculated Shannon indexes for guilds (H_{fx}). Abundance at the study sites was approximated using bait incidence as a proxy for abundance, normalized to the total number of sample baits at each site (NormSPIN).

Beta diversity, which is often overlooked in applied ecological studies, despite the fact that it can be the major component of biodiversity in agricultural systems (Clough et al. 2007), can be assessed using its direct complement, (species) similarity. That is, two different sample sites might both contain three species at the same levels of evenness: they have equivalent levels of alpha diversity. However, in terms of beta diversity, if they contain the exact same three species (spp. A, B, C), then there is complete similarity between the sites, and zero beta diversity. At the other end of the spectrum, if one site has species A, B, and C, and the other species D, E, and F, they have zero similarity and the highest level of beta diversity possible for the two sites.

For our study, we measured beta diversity by comparing Sørensen similarity (S), where lower similarity means higher beta diversity: Sørensen ranges zero to one, where zero indicates no species overlap, and one indicates complete overlap. We computed S in EstimateS, using Chao's incidence-based estimators, which attempts to account for shared species that were not directly detected in the samples recovered, using the probability that two randomly chosen individuals (one from each of two sites) both belong to species that are shared by both samples, though not necessarily the same shared species (Colwell 2005). Because these comparisons must be done pair-wise between individual sites, they were analyzed using randomization (resampling without replacement) tests; see Data Analysis, below.)

Analysis of alpha diversity

Although our study's intent is to assess possible impacts of participation in SMASAN on ground-foraging ant diversity in the region, this diversity will also naturally be affected by the typical drivers in fragmented landscapes, such as the number and area of forest fragments, edge area, distance of sampling from the nearest forest fragment, etc. With this in mind, these variables were examined and included in our analysis in order to control for their effects.

To obtain data on these local landscape characteristics, images of each site were recovered using Google Earth (Google Inc. 2008). These images were processed using the program ImageJ (Rasband 1997-2008) to detect and approximate the extant forest fragments in the landscape. After processing, distances between fragments and field sites were recorded, and ImageJ's "Analyze Particles" function was used to recover area and perimeter data on all fragments greater than 1 ha in size. Following image analysis, linear mixed-effects models (LMM) were created based on the following collection and landscape characteristics: collection year (YEAR); collection farm (FARM); collection day (a proxy for seasonality; DAY); total of all the fragment perimeters (i.e., total fragment edge) within 2 km (LCLEDGE); total area of forest cover within 2 km (LCLAREA); number of fragments within 2 km (FRAGNUM); nearest fragment distance (FRAGDIST); participation in SMASAN (SMASPART); and shape index (the ratio of the actual perimeter to the minimum possible perimeter for the same amount of area) (SHPIDX). (See Chaves 2010 on the use of LMMs to avoid pseudoreplication in ecological research.) These variables were chosen based on established literature on matrix effects and fragmentation (Fahrig 2003; Kupfer et al. 2006, Perfecto and Vandermeer 2002).

To assess the possible effect and magnitude of effect of each variable on biodiversity, linear and linear mixed models were created in R (version 3.1.2, R Core Team, 2014) using the "LME4" package (version 0.999999-0) based on our nine independent variables: DAY,

LCLAREA, LCLEDGE, FRAGNUM, FRAGDIST, SMASPART, and SHPIDX were fixed effects variables; YEAR and FARM were treated as random effects variables. These independent variables were tested for collinearity, and pairs whose r^2 values exceeded 0.7 were removed from the analysis. LCLEDGE and FRAGNUM were correspondingly removed; the pairwise r^2 value of the remaining variables were all < 0.6. Additionally, prior to creating the LME models, data exploration was conducted using Cleveland dot plots. One outlier was identified (SEDD) and removed from data.[3]

Following this data exploration and preparation, we generated candidate models to analyze using an information-theoretic approach. The strength of the evidence for candidate models was analyzed using AICc (Akaike's Information Criterion corrected for small sample size): Akaike (AICc) weight, which ranges from zero to one, is roughly analogous to the probability that a given model is the best model given the data analyzed (Symonds and Moussalli 2010).

Due to the lack of strong evidence for a single model for any of the response variables (i.e., the weight of the top model was not >0.9), multimodel inference—specifically, model averaging—was chosen as the best method to explore the effect of independent variables on the various diversity measures (Burnham and Anderson 2002; Burnham and Anderson 2004; Whittingham et al. 2006; Burnham et al. 2011). As compared to stepwise/model selection approaches, model averaging prevents the loss of information contained in the alternate models for which there is still support, and avoids the necessity of having to choose a "best" model when numerous models have near-equal support (Burnham and Anderson 2002; Mazerolle 2006). This approach does, however, require that the results be interpreted cautiously (Galipaud et al. 2014).

We used the dredge function of R's "MuMIn" package (version 1.9.5, Bartoń, 2013) in order to automate our analysis, with all possible models and submodels generated based on the independent variables remaining after the removal of LCLEDGE and FRAGNUM. Using AICc, we retained the set of most likely models with cumulative Akaike weight of 0.95. The Akaike weights and the coefficients estimated in each individual model were then used to create weighted averages and 85% confidence intervals[4] for each of the coefficients included in the retained models; r^2 values were used to assess model fit (Burnham and Anderson 2002; Burnham and Anderson 2004; Burnham et al. 2011). We used full average coefficients; this method assumes a zero value for any parameter not in a specific model in the retained set. It is the recommended approach when there was not a single best model with an Akaike weight >0.9 (Symonds and Moussalli 2010). This naturally has a tendency to shrink averages towards zero, making them a more conservative estimate than the conditional average, which only averages a parameter from the subset of models that actually contain said parameter. A comparison of model marginal and conditional r^2 values can be then used to assess the amount of variance explained solely by the fixed effects (marginal) and the combined variance explained by the fixed and random effects (conditional). For all diversity measures except normalized species index, the marginal and conditional r^2 values were nearly identical, indicating the random effects accounted for little to no variance. Thus, for our main analysis, the random effects terms were removed for models of all diversity measures except normalized species index, meaning they were analyzed with linear models rather than linear mixed models (see Nakagawa and Schielzeth 2013). Lastly, distributions for the models were determined by graphing the values assuming different standard distributions and analyzing residuals to choose the best fit. The values best fit a normal distribution for all diversity measures.

Beta diversity

Potential differences in beta diversity between SMASAN and non-SMASAN farm fields and adjacent forest fragments were tested via pairwise comparisons between each site, and averaging beta diversity within categories (SMASAN fields, non-SMASAN fields; SMASAN forests, non-SMASAN forests). The differences in averages were compared via randomization tests—resampling without replacement—using 10,000 iterations for each test with the Resampling Stats for Excel package (Resampling Stats, Arlington, VA, USA). Randomization testing was chosen for its simplicity and minimal assumptions it requires (Good 2006), though it comes with specific caveats (see below).

Study Limitations

Given the small number of farmers in SMASAN's programs, our intention was to compare a random set of SMASAN farms to socioecologically similar neighboring farms to form a rough natural experiment on the effects of SMASAN on farmer practices and therefore differences in biodiversity within the local agroecological matrix (both farm fields and adjoining forest fragments). Although the response rates we obtained were reasonable, the usual caveats apply; farmers who agreed to be interviewed may differ systematically from those who declined. Further, due to limits on time and resources, the agroecological similarities of SMASAN and non-SMASAN farms were based on the farmers' own evaluations in the snowball sampling process, and their self-reports with regards to agricultural practices. A number of non-responses and vague answers on income make exact socioeconomic comparison difficult, but the similarities in size, age, education levels, history, and crops grown, and the farms' close proximity to each other support our decision to treat them as an adequate sample for exploratory analysis. Based on this limited data, the one obviously notable difference between SMASAN and non-SMASAN farms was in average income; we will return to this in our discussion.

Although small sample size is more likely to increase Type II ("false negative") rather than Type I errors, the small number of farms sampled for our study does raise the possibility that the full variation of farmer and forest conditions was not captured by our sampling. This is especially true given that partner farms of SMASAN range up to 100 km away from the city, in multiple compass directions, although the area we sampled is the site of the majority of SMASAN-partnered farms. And in terms of potential overfitting in our models given the small small sample size: AICc severely penalizes adding parameters when using a small data set, making our analysis conservative in some respects.

With regards to the randomization tests used to compare beta diversity, potential biases from non-representative sampling is also a highly pertinent concern, and means that our results should be viewed extremely tentatively. That is, in our case randomization tests give a precise answer as to how likely a difference in means at least as large as that observed between the groups present in the sample would be to arise by chance, but it does not itself allow inference about the larger population(s) the groups are drawn from. Rather, the validity of inferring to the larger population of farms depends entirely on whether or not the sampled farms are in fact representative of their larger populations.

Thus with the novel nature of this study's questions and approach and the small sample size, it is very important that our results be understood to be exploratory. The caveat that they should be re-examined by further research drawn from a representative sample, and specifically designed to test our preliminary conclusions, holds even more strongly than usual.

RESULTS

A total of 76 species and morphospecies in 22 genera and 6 sub-families were collected from 11 sites across 7 farms. Overall, there was an average of 14.4 species per site (standard deviation 6.05) as estimated by ICE. Farm fields averaged 10.7 species per site; forest fragments averaged 19.5 species per site. The sub-family accounting for the most species was by far Myrmecinae (40), followed by Formecinae (19), Dolichoderinae (6), Ponerinae (7), Ectatominnae (3), and Ecitoninae (1). In terms of functional groups, ants classified as Tropical Climate Specialists were by far the most numerous. This is in large part due to the ubiquity of the fire ant *Solenopsis saevissima*, which was found at almost every site, usually in both the field and forest areas.

Species Richness (ICE)

As can be seen in Table 2, our analysis indicates substantial support for the effects of two variables (i.e., the 85% confidence interval for their coefficients does not include zero) on species diversity as measured by ICE: FRAGDIST (coefficient: -0.123; 85% CI: -0.196, -0.061) and SMASPART (coefficient: 1.716; 85% CI: 0.285, 7.831). Marginal r^2 values for models containing FRAGDIST ranged from 0.33 to 0.58. Models containing SMASPART had marginal r^2 values ranging from 0.44 to 0.58. (Some models contained both; see Table S1 in Supplementary Materials.) The relatively high degrees of fit for these models strengthens the inference that both of these variables notably affect species diversity as measured by ICE.

----*Table 2 about here*----

Species Abundance (Normalized Species Incidence)

For our abundance proxy, Normalized Species Incidence, our data indicated substantial support for the effects of two variables: DAY (coefficient: -0.564; 85% CI: -0.912, -0.374); and FRAGDIST (coefficient: -0.593; 85% CI: -0.921, -0.523) (Table 2). Marginal r^2 values for models including the variable(s) of interest ranged from 0.24 to 0.58 (for collection day) and 0.29 to 0.58 (for nearest fragment distance) (Table S1).

Species Diversity and Evenness (Shannon, Species Evenness)

Model-averaging indicated substantial support for effects of FRAGDIST (coefficient: -0.0209; 85% CI: -0.0288, -0.0130), SMASPART (coefficient: 0.219; 85% CI: 0.0878, 0.930), and DAY (coefficient: -0.0041; 85% CI: -0.0221, -0.0009) on species alpha diversity as measured by the Shannon index (Table 2). Marginal r^2 ranged from 0.48 to 0.78 for models containing nearest fragment distance, 0.51 to 0.78 for SMASAN participation, and 0.57 to 0.78 for collection day (Table S1). For species evenness (E), there was substantial support for the effects of the variables FRAGDIST (coefficient: -0.005; 85% CI: -0.007, -0.003) and SHPIDX (coefficient: 0.052; 85% CI: 0.015, 0.114) (Table 2). Marginal r^2 ranged from 0.33 to 0.64 (nearest fragment distance) and from 0.55 to 0.64 (shape index) (Table S1).

Guild Diversity and Evenness

Substantial support for effects on guild diversity was detected for FRAGDIST (coefficient: -0.003; 85% CI: -0.011, -0.001) and SHPIDX (coefficient: 0.050; 85% CI: 0.004, 0.197). (See Table 2.) For nearest fragment distance, models including it had marginal r^2 that ranged from 0.13 to 0.47; for shape index it ranged from 0.14 to 0.47 (Table S1). With regards to guild evenness, evidence supported the effects of the same two variables: FRAGDIST (coefficient: -0.002; 85% CI: -0.005, -0.001) and SHPIDX (coefficient: 0.028; 85% CI: 0.011, 0.089). Marginal r^2 ranged from 0.13 to 0.45 (nearest fragment distance) and from 0.17 to 0.45 (shape index).

Beta diversity

Beta diversity was compared in terms of the species similarity (overlap) among SMASAN farm fields versus similarity among non-SMASAN farm fields; the species similarity between farm fields and associated forest fragments on SMASAN vs. non-SMASAN farms; and temporal species similarity (species similarity at the same site in different years) for SMASAN vs. non-SMASAN farms.

Average estimated Sørensen similarity between SMASAN farm fields was significantly lower (i.e., beta diversity was higher) than between non-SMASAN farm fields in 2006 when compared via randomization testing (S of 0.352 vs. 0.746; p=0.0233; see Table 3). (There was insufficient data to compare fields in 2005.) This analysis, however, included site SEDD, which was excluded as an outlier in our analysis of alpha diversity. Although SEDD's values for beta diversity were not similarly identified as outliers, when SEDD is excluded for consistency, average beta diversity remains higher (average similarity is lower) between SMASAN farms, but the result is no longer significant at p=0.05 (S= 0.502 vs. 0.741; p=0.098).

When comparing fields and forest fragments on the same farm, the mean similarity between SMASAN farm and forest fragments was higher than that the mean similarity between non-SMASAN farms and their adjacent fragments when compared via randomization testing, although this result was just shy of significance (0.381 vs. 0.0874; p=0.052; Table 4). No other comparisons of beta diversity were close to significance.

---*Table 3 about here*---

---*Table 4 about here*---

DISCUSSION

The study we present here was designed as an initial exploration of the potential effects of participation in SMASAN's programs on regional biodiversity. We measured and analyzed characteristics of the larger landscape in order to control for them in our analysis. For this reason, disentangling the precise mechanisms and dynamics of fragmentation, as suggested by Fahrig 2013 and Kupfer et al. 2006, is beyond the scope of the current work. Our analysis and modeling approach were, practically speaking, agnostic towards which of the dynamics outlined by Fahrig 2013 may in fact be the dominant or true mechanism driving fragmentation's effects on biodiversity. For this reason, our discussion focuses on the results involving SMASAN participation, and does not specifically explore the results from the point of view landscape characteristics.[5]

Our analysis did reveal initial evidence for positive effects of participation in SMASAN on alpha diversity, specifically in terms of ICE and the Shannon index. In terms of ICE, participation in SMASAN may correspond on average to the presence of somewhere between a quarter and almost eight more species per site (85% CI = 0.285 – 7.831). With a total of 76 species found overall, and an average ICE about 14 species per site, the 85% CI for SMASAN participation represents a potentially meaningful effect size. Similarly, the 85% CI of SMASAN participation's effects on the Shannon index (0.0878 - 0.930) reinforces this initial evidence for a biologically meaningful effect; Shannon diversity typically ranges from 1.5 to 3.5 (Magurran 2013).

SMASAN farms also appeared to have significantly greater beta diversity among them than non-SMASAN farms (Table 3). The greater beta diversity seen among SMASAN farms means that they contribute more to the overall landscape (ɣ) diversity than non-SMASAN farms. Our results are comparable to recent research finding significantly greater between-site beta diversity for birds in low-intensity agricultural systems as compared to high-intensity systems (Karp et al. 2012); and greater between-site beta diversity for plants (Gabriel et al. 2006) and bees (Clough et al.) in organic fields compared to between-site beta diversity in fields under conventional management (Clough et al. 2007). Gabriel et al. and Clough et al. also found that beta diversity in their studied systems was the most significant contributor to total (ɣ) diversity.

Beyond the direction contributions to landscape diversity from the higher beta diversity seen among SMASAN farms, our results are broadly consistent with what one would expect to see in higher quality agricultural matrices surrounding forest fragments. Our results indicated some evidence for greater similarity between the species found in SMASAN fields and their adjacent forest fragments (average similarity was over four times greater, though the difference was marginally insignificant; $p=0.052$). Higher quality matrices can supply temporary habitats to a larger portion of the total pool of species in an area; because some or even many of the species cannot survive in the matrix indefinitely, there is constant turnover as different species emerge from the forest and temporarily colonize the matrix. In other words, higher quality matrices should have greater beta diversity. The higher estimated similarity between field and forest species on SMASAN farms further mirrors prior research comparing different farming methods' effects on matrix quality and biodiversity in coffee, cacao, silvopastoral, and home garden agroecosystems (see reviews in Perfecto and Vandermeer 2008 and Winqvist et al. 2012).

So, given that our results mirror prior works comparing alternative and conventional agricultural methods in terms of effects on both alpha and beta diversity, what are the differences, if any, between the practices used by SMASAN and non-SMASAN farmers, and can these differences be tracked back to the relationship with SMASAN? As we presented earlier in *Background on SMASAN and Straight from the Countryside*, interviews with farmers did not provide sufficient detail or evidence of systematic differences between the practices of

SMASAN and non-SMASAN farmers. Given this, there are several possible interpretations of our results.

The most straightforward possibility is that our small sample size generated false positives based on incomplete or inadvertently biased sampling of the populations. The snowball method used to recruit farmers, and the selection bias of farmers willing to participate may have generated an unrepresentative sample. Though there is no particular reason that these possibilities should have biased the results in favor of SMASAN, the possibility cannot be ruled out, particularly in the case of the results for beta diversity: inference from randomization tests depends entirely on how representative the sampled populations are of their source populations.

A second possibility is that the results are representative of SMASAN and non-SMASAN farms, but that SMASAN farms are not representative of farms overall. That is, the farmers who opt in to SMASAN programs may differ systematically somehow from farmers who do not, though in terms of the characteristics of the landscapes we included in our models and the socioeconomic background information retrieved from interviews (Chappell, forthcoming), there is no direct indication of this (outside of the potential income effects discussed below).

The third possibility is that involvement in SMASAN really has contributed to greater alpha and beta diversity on participating farms. If this were the case, it could be the result of the increased income and financial security SMASAN farmers appear to be receiving in terms of stable, reliable and fairly-priced markets for their produce, according to farmer interviews and demographic data (Chappell, *forthcoming*). Financial security and capital have been tied to the ability of farmers to implement conservation-oriented practices (Baumgart-Getz et al. 2012; Marshall 2009; Vanclay 2004), as we noted in the introduction. It is possible, therefore, that the better outlook and positive attitudes with regards to economic stability and security from

SMASAN farmers may be reflected in the quality of their management, encouraging biodiversity in subtle or indirect ways. For example, one SMASAN farmer reported that she diversified her crops in response to the stability and encouragement provided by the Secretariat; such planned biodiversity, in turn, has been shown to be strongly linked to "associated biodiversity" (Vandermeer et al. 2002). She additionally said that she dramatically cut down on pesticide use after she entered the program. This raises the additional possibility, in terms of mechanism, that the process of preparing for and adhering to SMASAN's quality and safety standards has altered farmer practices in ways that better support biodiversity. However, some non-SMASAN farmers also stated that they avoided pesticides or grew diverse crops.

In fact, based on direct observation, use of synthetic pesticide and fertilizers among all farmers varied and did not seem to differentiate neatly between SMASAN and non-SMASAN, though no farmers kept exact records of pesticide amounts or time of application, making precise comparison difficult. However, SMASAN staff working with the farmers (both the extensionists and the coordinator of the Straight from the Countryside program) often quite clearly encouraged them to reduce synthetic inputs and move towards organic production, which is unsurprising given that Chappell's forthcoming examination of SMASAN's goals established that sustainability and supporting organic production appeared as both formal and informal goals of the programs.

A last (non-exclusive) possible explanation of the observed effects from SMASAN participation is the role of SMASAN extensionists. As stated earlier, participating farmers are visited by SMASAN's extension agent at least once a year, after a series of initial visits before they are allowed to join the program. Besides monitoring conformance to SMASAN standards, SMASAN's extensionists have occasionally visited to respond to specific issues arising between the farmer and SMASAN. The guaranteed yearly contact and occasional further interactions, and the fact that the current extension agent and his predecessor have both been enthusiastic proponents of organic agriculture and agroecology (*pers. obs.*) offer another potential, and direct, mechanism for any differences in SMASAN and non-SMASAN farms in terms of practices and biodiversity. The potential importance of such interactions appears all the greater in reference to the fact that all studied farmers cited guidance and interactions from extension as being fundamental in both their understanding of how to use pesticides effectively and safely, and in how to reduce pesticide use (i.e., as-needed spot treatments as opposed to regular broadcast applications) or use organic methods. Compared to the minimum guaranteed contact with SMASAN extensionists, farmers across categories reported difficulties in engaging with their local state extension. Farmers reported that it had become harder to find and enroll in the classes that state extension previously offered, and that it was increasingly difficult to get extensionists to visit promptly. One farming family felt that it now depended on local governments' to support extension and other aid to small farmers, despite the status of extension as a nominally state government-funded entity. Nabuco and Souki (2004) similarly commented that there had been a decrease in the number of technicians [extensionists] contracted with the state. Thus though regular extension is decreasing, SMASAN farms will nonetheless see an extension agent with some regularity who may serve as an additional prod and opportunity to learn, implement, or maintain sustainable practices.

Previous research has found that access to adequate information can be a key factor in the adoption of more sustainable practices (Baumgart-Getz et al. 2012; Marshall 2009) and farmers' and technicians' perceptions can influence practices and production results to a surprising and non-obvious degree (Bulte et al. 2014). The current and former SMASAN extensionists were observed to spend time consulting with the farmers and discussing the practical aspects of implementation with them. This time advising and consulting was, both extensionists admitted, beyond the strict scope of their job description, but something they nonetheless viewed as a priority and in keeping with the unwritten spirit of SMASAN's programs.

CONCLUSIONS

This study may be the first to directly link upstream food policy decisions with local effects on wild biodiversity and abundance, showing the ecological importance of examining not just human activities within the matrix, but also within the larger sociopolitical system (i.e. the influence of SMASAN and extension). The potential effects revealed by our data linking participation in SMASAN with higher ground-foraging ant alpha and beta diversity follows the general trend in studies reinforcing the importance of human social context and the matrix's role in maintaining and supporting biodiversity and conservation in larger landscapes (Perfecto et al. 2009), and reiterates the need to consider specific characteristics of human land use and social factors that determine the quality of the matrix. Based on the results presented here and in Chappell (forthcoming), a conventional ecological approach might miss the mechanisms at work if it focused only on factors within the landscape itself and not on participation in SMASAN, SMASAN's influence on economic security, and the increased access to extension. However, as we presented in our discussion, competing explanations cannot be ruled out at this stage and further research should build on our exploratory results.

Nonetheless, the possibility that the innovative food security programs of SMASAN may be indirectly supporting biodiversity conservation in the surrounding landscape, when sustainability and conservation were only secondary goals with limited resources behind them, is a novel and potentially important contribution to our understanding of the food security-biodiversity nexus. As one reviewer noted, the majority of the literature on food security and biodiversity rather addresses the ways biodiversity can support food security (e.g. Snapp et al. 2010) or the configurations of potential trade-offs between the two (Fischer et al. 2013; Phalan et al. 2014). The present study takes a different tact by examining the potentially positive effects of increased food security on biodiversity. It also re-emphasizes the importance of economic security and access to education and information for small farmers, specifically in terms of helping agriculture to be a more sustainable and integrated part of broader conservation strategies. Lastly, the possibility that food security and biodiversity conservation can be supported simultaneously contradicts the well-established common wisdom that human welfare and environmental conservation are, to some degree, inimical to each other. Along with recent work synthesizing information on production and biodiversity conservation (Chappell and LaValle 2011; Melo et al. 2013; Tscharntke et al. 2012), there is thus the potential that addressing the urgent needs of the many, in terms of food security at least, may be done in ways good for both humans and our environment through appropriate measures improving social, economic, and technical support for farmers.

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Supplementary Material

Appendix A

Ant species and morphospecies (organized by subfamilies) found in seven vegetable farms using tuna bait sampling over a two-year sampling period.

Appendix B

Table S1: Model selection tables for diversity measures

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[1] Since the original time of this research, a number of other local and national programs have sought to accomplish similar goals—including the famous national "Zero Hunger" programs—in terms of supporting farmers. See the Brazilian Ministry of Social Development and the Fight Against Hunger 2010; Rocha et al. 2012; Oldekop et al. 2015.

[2] Appropriate IRB approval was obtained; Application UMIRB B04-00006385-I.

[3] SEDD had several unique socioecological characteristics that reinforced our decision to remove it as an outlier in our analysis of alpha diversity.

[4] 85% confidence intervals are more consistent with our IT analytical approach than the customary 95% CIs; see Arnold 2010.

[5] However, one might note that our results for landscape characteristics are in fact consistent with previous studies on arthropod biodiversity, particularly the extensive work with ants in coffee agroecosystems (Perfecto and Vandermeer 2002; Armbrecht and Perfecto 2003; see also Tscharntke et al. 2007). Specifically, substantial support was found for the negative effects of increasing distance from the nearest habitat patch (nearest fragment distance) for measures of species and guild alpha diversity.