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Wild native trees in tropical homegardens of Southeast Mexico: Fostered by fragmentation, mediated by management

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ABSTRACT

Tropical homegardens (THGs) are a model system for rural development that may reconcile food production with social resilience and biodiversity conservation, particularly in rapidly changing landscapes. This study quantified the sink function of THGs for wild native trees in relation to tree cover fragmentation, garden management and household socio-economics. Abundance, richness and diversity of naturally established spared native trees were recorded for 59 rural THGs in Southeast Mexico, along a gradient of tree cover fragmentation. The majority of native species and individuals encountered in THGs had arrived naturally. Contrary to previous work, both the abundance and diversity of spared native trees increased with tree cover fragmentation. However, this sink function was strongly mediated by the type of garden management: lush, multi-layered gardens and gardens with few exotics and low labour input had more spared native trees of more species, while simple-structured gardens and gardens with high labour input and many exotic fruits had only few. Overall, the results indicate that tree cover fragmentation determines which species come in, and management determines how many of each stay. Our results clearly demonstrate that THGs are crucial sinks for wild native trees in deforested fragmented landscapes. THGs are ubiquitous, and could also be key sources for reforestation; here we coin *homegarden-based natural regeneration* as a new concept. Since garden management has a clear impact, further research is needed as to how socio-economic, cultural and ecological functions of THGs can be optimised in different landscape contexts.

1. Introduction

The world is facing a biodiversity crisis that is problematic for humanity both *via* decreased ecosystem services (Cardinale et al., 2012; Díaz et al., 2006; Hooper et al., 2012; Isbell et al., 2011) and the impoverishment of local knowledge and tradition, food sovereignty and social resilience (Adger, 2000; Berkes et al., 2003; Elmqvist et al., 2003; Jackson et al., 2007). Since agricultural expansion is widely considered to be a major factor of biodiversity loss, there is growing attention to find ways of reconciling food production with rural development and biodiversity conservation (Flynn et al., 2009; Henle et al., 2008; Sodhi et al., 2010; Tscharntke et al., 2012a). Particularly in the tropics, forests are converted and fragmented at a high pace, and in wide areas the original forests have already been replaced completely (Clements et al., 2014; Laurance, 1999). Forest transitions lead to newly increasing forested areas by natural secondary growth in abandoned plots (Meyfroidt and Lambin, 2011; Poorter et al., 2016; Rudel et al., 2005), and nature may also rebound in managed systems that are still operational (Blitzer et al., 2012; Perfecto and Vandermeer, 2008; Tscharntke et al., 2012b). The result is a human-modified, highly dynamic and complex landscape matrix that is poorly understood, yet offers ample opportunity to conserve biodiversity and improve rural livelihoods (Laurance, 1999; Sodhi et al., 2010; Steffan-Dewenter et al., 2007).

Many studies focus on how the landscape matrix supports the functioning of managed agro-ecosystems, demonstrating that matrices with higher tree cover and more (semi-) natural elements support a higher diversity of functional species groups, such as predators and pollinators (Clough et al., 2009; Tscharntke et al., 2005). However, some studies report that small-scale, structurally diverse farming systems can offer habitat for a range of species to establish, even if the surrounding landscape is largely deforested (Engelen et al., 2016; Lemessa et al., 2015c). Thus, managed systems can also support surrounding (semi-) natural systems, rather than just the opposite. This remains largely unstudied, particularly in fragmented tropical landscapes (Blitzer et al., 2012).

Tropical homegardens (THGs) may be particularly suitable as model system for biodiversity conservation in such landscapes (Abebe et al., 2013; Bardhan et al., 2012; Galluzzi et al., 2010; Kumar and Nair,

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2006b; van der Wal and Bongers, 2013; Webb and Kabir, 2009). THGs are diversified agroforestry systems around homesteads, with a multilayered structure and a high direct use value, delivering products for food, medicine and construction (Kehlenbeck et al., 2007). THGs are well documented in the literature by a wide range of disciplines; their ecological, economic and socio-cultural importance are increasingly recognised (Bardhan et al., 2012; Belcher et al., 2005; Galluzzi et al., 2010; Kehlenbeck et al., 2007; Kumar and Nair, 2006b; Perfecto and Vandermeer, 2008; Webb and Kabir, 2009). It is widely supported that they are local biodiversity hotspots for functional groups such as predators, pollinators and decomposers (Engelen et al., 2016; Huerta and van der Wal, 2012; Lemessa et al., 2015a,b,c; Tscharntke et al., 2008), and particularly for higher plants, such as heirloom crops and/or rare wild native trees (Abebe et al., 2013; Bardhan et al., 2012; Cruz-Garcia and Price, 2014; Galluzzi et al., 2010; Salako et al., 2014). As THGs provide many families' forest needs right around the house, they may help to reduce deforestation (Das and Das, 2005; Kehlenbeck et al., 2007). Furthermore, they are numerous and ubiquitous in the tropics, particularly in Mesoamerica and South-East Asia (Kumar and Nair, 2006a).

THGs can be both sinks and sources of tropical biodiversity. As sinks they provide refuges for natural forest species that could become (locally) extinct with increasing deforestation (Engelen et al., 2016; Griffith, 2000); and as sources of natural forest species they can help refurbish the landscape (Del Castillo, 2015). Quantitative evaluation as to what extent THGs may contribute to the sink and source dynamics of biodiversity in deforested landscapes, and how this depends on tree cover fragmentation, household socio-economics, garden management and autecological factors, is largely lacking.

Landscape ecology studies show that fragmentation can affect seed dispersal in dramatic ways; particularly species that depend on zoochory by forest specialists suffer greatly (Corlett, 1998; Gorchov et al., 1993; Laurance et al., 2006b; Medellin and Gaona, 1999; Parrotta et al., 1997; Peña-Domene et al., 2016; Shiels and Walker, 2003). In complex agroforestry systems like THGs, people are an essential element alongside animals and plants, all highly dynamic in space and time (Michon et al., 1983). So next to ecological processes, socio-economics and management (henceforth referred to as socio-management) also play a pivotal role (Belcher et al., 2005; Kehlenbeck et al., 2007; Michon et al., 1983). Factors such as income class, family size, botanical knowledge, social networks for plant material exchange and individual preferences have all been found to affect THG biodiversity (Aguilar-Støen et al., 2009; Ban and Coomes, 2004; Buchmann, 2009; Galluzzi et al., 2010; Kehlenbeck et al., 2007). Thus, while the overall seed influx of wild[1] native species into THGs can be expected to decrease with fragmentation of tree cover (Cramer et al., 2007; Dallinga, 2015; Gorchov et al., 1993; Peña-Domene et al., 2016), the eventual survival of settlers likely depends more on the management practises of individual families, driven by, among other things, species knowledge, socio-economy and individual preference (Blanckaert et al., 2004; Buchmann, 2009; Kehlenbeck et al., 2007; Moreno-Black et al., 1996).

This study evaluated the sink function of THGs for wild native trees and how this is related to tree cover fragmentation, garden management and household socio-economics. The study area in southern Mexico consists of largely deforested fluvial plains where the great majority of rural inhabitants have a perennial homegarden (Méndez, 2012). The abundance and diversity of wild native trees was quantified along a gradient of tree cover fragmentation, while simultaneously recording a wide range of garden socio-management factors. We tested the following hypotheses:

1. The abundance and diversity of wild native trees in THGs declines with increasing tree cover fragmentation.

2. The abundance and diversity of wild native trees in THGs is better explained by garden socio-management than by tree cover fragmentation.

2. Methods

2.1. Study site

The study area consists of the fluvial plains of the municipality Comalcalco, in the lowlands of Tabasco state, southeast Mexico (Fig. 1). The municipality covers 723 km^2 and the central part, at the town Tecolutilla, is located at 18.3 N, −93.3 W (decimal degrees). The altitude ranges from 0 to 40 m asl, it has a mean annual precipitation of about 2000 mm and a mean temperature of 26.4 °C (Muñoz et al., 2006). Fieldwork was conducted from February to May 2016, during the dry season. The dominant soil type is gleysol with some cambisol and fluvisol, and solonchak in the most Northern part, close to the shore (Fig. 1b; INEGI, 2012). The main urban centres are Comalcalco city (ca. 42.000 inhabitants) and Tecolutilla (ca. 11.000 inhabitants). The rural population of Comalcalco is divided over small villages on both private lands and *ejidos* (a system of common land tenure; Aguilar et al., 2009; van der Wal and Bongers 2013). The entire region was deforested from the 1950's onwards as a result of agrarian government policy (Revel Mouroz, 1980; Roces et al., 1989) and today nearly all forest is planted cacao agroforest and THG (Fig. 1c–d).

2.2. Site selection

Starting with a database of 111 settlements in Comalcalco and land use data of 2012 (INEGI, 2012), a first selection of 29 villages was made in ArcMAP 10.1 (Esri, USA). Selected villages had (1) between 100 and 1000 inhabitants, (2) were at least 1 km away from the main urban centres (Fig. 1d) and (3) at least 1 km away from tree species-poor mangroves, saltmarshes and freshwater wetland (Fig. 1d).

Population size was kept within fixed, practical limits and settlements close to urban centres were excluded, since urbanisation and market orientation can affect THG plant species composition (Major et al., 2005; Poot-Pool et al., 2015). Villages close to mangrove stands, saltmarsh and other wetland (Fig. 1d) were excluded. Species from wetland habitats are unlikely to establish in homegardens, with habitats more alike those of the native vegetation on fluvial plains (Van der Wal, pers. obs).

To obtain a selection of villages more or less equally spread among the tree cover fragmentation gradient of the entire region, tree cover fragmentation in a 1000 m buffer area around each of the remaining 29 villages was classified into 3 groups from low to high, such that the range of group averages was maximised and each group contained at least 25% of the total sample. The metric edge density of tree patches (TED; Table 1; Table A.1) was used to characterise tree cover fragmentation for this initial classification, because Dallinga (2015) has shown that it explains the biodiversity of secondary forest better than does a simple metric like tree cover. Subsequently, 7 villages of the low TED group and 6 of the other two groups were randomly selected, with resampling in case of overlapping buffer areas. In each village, 3 to 4 homegardens were selected randomly, by numbering each homegarden using Google Earth (2016). Gardens estimated in the field to be < 500 m^2 or > 2000 m^2 were omitted to limit the effect of garden size (Peyre et al., 2006; van der Wal and Bongers, 2013). In total, we sampled 59 homegardens in 19 villages.

[1] 'Wild' is defined here as having established naturally, without direct user effort and intention. It may include subsequent cultivation and it may derive from cultivated progenitors in gardens or agricultural plots. There is no reference to genetics and it may thus differ from other definitions in the literature.