Drivers of plant biodiversity and ecosystem service production in home gardens across the Beijing Municipality of China

Lorraine Weller Clarke · Liangtao Li ·
G. Darrel Jenerette · Zhenrong Yu

Abstract Home gardens have been recognized as repositories of agricultural biodiversity across the world. The influence of socioeconomics and location-specific factors on urban gardening patterns merits continued study. Using Beijing Municipal Province in China, a rapidly urbanizing region, as a case study, we address two questions: 1) How do biodiversity patterns change between different urbanized regions in Beijing? 2) How do ecosystem services provided by Beijing home gardens change with socioeconomic status and location-based preferences of gardeners? We surveyed 104 home gardens in Beijing Municipal Province for plant biodiversity, abundance, and species ecosystem services (ES) (provisioning or cultural uses). The gardens were distributed across three urbanized regions (suburban, peri-urban, and exurban). We found that species biodiversity and abundance shift according to a hierarchy of need from ornamentals (cultural ES) to edibles (provisioning ES) with increasing distance from Beijing. These trends are related to reduced income, lowered food security, and lack of urban markets in exurban regions. Rarefaction curves indicate ornamental species drive β diversity. Ordination also showed a shift in species composition with increasing isolation from the city; Suburban and exurban gardens were the most different, while peri-urban gardens were similar to both others. Only exurban gardens had a positive relationship between species and area. High edible cover and high species density indicates that demand for edibles in exurban regions may be higher than space constraints allows. Our study better quantifies species biodiversity patterns in Beijing, and can inform urban planners about the value and usefulness of home garden space.

Keywords Biodiversity · Ecosystem services · Species-area relationship · Socioeconomics · Agriculture · Food security

Lorraine Weller Clarke and Liangtao Li contributed equally to this work

L. W. Clarke (✉) · G. D. Jenerette
Department of Botany and Plant Sciences, University of California Riverside, 2150 Batchelor Hall, Riverside, CA 92521, USA
e-mail: Lorraine.clarke@email.ucr.edu

L. Li · Z. Yu
College of Agricultural Resources and Environmental Sciences, China Agricultural University, Beijing 100193, China

Introduction

Home gardens are complex, multi-layer systems of trees, shrubs, and annuals around home-steads (Kumar and Nair 2004; Michon and Mary 1994; Del Angel-Pérez and Mendoza 2004). They are a ubiquitous landscape across the world, with an estimated 15–36 % of residential land in the UK, India, Africa, and China occupied by home gardens (Loram et al. 2008; Davies et al. 2009; Cilliers et al. 2012; Huai et al. 2011; Jaganmohan et al. 2012; Baudry and Yu 1999). These ecosystems are increasingly the focus of coupled human-natural systems re-search (Kirkpatrick et al. 2007; Aguílar-Stoen et al. 2009; Bernholt et al. 2009; Kabir and Webb 2009) with increased scientific demand for quantification of home garden plant species abundance, community diversity, ecosystem functioning, and ecosystem services (Huai and Hamilton 2009; Jaganmohan et al. 2012). Variation in garden biodiversity and abundance can be high, even within a single urbanized region, due to socioeconomic or cultural status of residents (Lubbe et al. 2011; Cilliers et al. 2012; Jaganmohan et al. 2012). Home gardens are potentially hotspots of agricultural biodiversity (Airriess and Clawson 1994; Nguyen 2003; Kumar and Nair 2004; Galluzzi et al. 2010), which stand in contrast to mono-cultured commercial croplands. Quantitative studies of home garden agro-ecosystems can provide opportunities for rapid increases in fundamental knowledge of how biological organization directly affects local nutrition, biodiversity, and global food security.

Our study incorporates an ecosystem service (ES) approach to home garden research in Beijing, China, focusing on how demand for certain services, and thus plant choice changes spatially with the needs of residents across an urbanizing gradient (McDonnell and Hahs 2008; Cilliers et al. 2013). The Millennium Ecosystem Assessment (2005) defines ES as "the benefits people receive from ecosystem function". These can be subdivided into provisioning (food, water, timber), regulating (those affecting disease, climate, flooding), cultural (recrea-tional, aesthetic, spiritual) and supporting (nutrient cycling, photosynthesis) services. Chinese urban and rural settlements have a long cultural history of home gardens, providing both provisioning and cultural ES to participants (Baudry and Yu 1999; Huai and Hamilton 2009; Huai et al. 2011). Increased biodiversity in landscapes can also provide indirect supporting ES such as soil nutrient cycling, pollinator biodiversity, and biological control of pests. For example, Beijing villages with biodiverse field margins and home gardens have higher carabid biodiversity, important predators for agricultural systems (Yu et al. 2006). High biodiversity in edible, ornamental, and useful shade plants as part of home garden ecosystems contribute to provisioning and cultural ES production, as well as supporting ES (Galluzzi et al. 2010; Mitchell and Hanstad 2004).

Research on home gardens outside of Europe has been primarily focused on rural gardens (Del Angél-Perez and Mendoza 2004; Huai et al. 2011), though some recent work has examined urban regions in the developing world (Molebatsi et al. 2010; Lubbe et al. 2011; Jaganmohan et al. 2012). Urban gardens have been shown to maintain local food security (Wezel and Bender 2003), especially in the rapidly urbanizing regions of the developing world (Cilliers et al. 2013). Beijing, China is one of the most rapidly urbanizing regions in the world, and its food systems are threatened with rapid and extensive conversion of agriculture to urban and non-agricultural uses (Ho and Lin 2004; Zhang et al. 2006). China must feed 22 % of the world's population on 6.4 % of the global land area, 7.2 % of the world's farmland, and with 5.8 % of the world's annual water resource (CCICCD 1996). The structure and size of agricultural land in China has been changing since reforms in the 1970s (Baudry and Yu 1999). The current challenge for home gardens outside of Beijing is their uncertain land tenure; land use policy enacted in 1995 states that agricultural land around the city cannot be effectively protected by the government unless it is competitive with other urban land uses

(Zhang et al. 2009). Quantifying the value of home gardens as compared to urban developments and improved transportation connectivity can be difficult, especially in areas with reduced socioeconomic resources.

Socioeconomic factors have been widely shown to influence plant biodiversity in human dominated ecosystems (Hope et al. 2003; Kinzig et al. 2005). One framework to better understand regional socioeconomic effects on garden species choice is a hierarchy of needs (Lubbe et al. 2011; Clarke et al. 2013). Within this framework gardeners are expected to plant species according to their needs, from food and medicine to aesthetics. Rural villages generally have reduced local income and access to urban food markets as compared to urban dwellers (Zimmer and Kwong 2004) and may be expected to select garden species providing provisioning ES, like edibles or medicinals with less emphasis on aesthetic species (Lubbe et al. 2010; Cilliers et al. 2012). In addition, agricultural knowledge and participation has been closely linked with edible biodiversity in rural areas; this secondary hypothesis indicates that gardeners who rely monetarily on garden success may be more likely to plant edible species (Fu et al. 2006; Lubbe et al. 2011; Galluzzi 2012). In contrast, higher incomes and access to urban markets in suburban villages may cause a garden composition shift towards ornamentals, which provide aesthetic and cultural ES. A pattern of increased ornamental diversity and decreased edible abundance correlated with income has also been observed across Europe (Galluzzi et al. 2010; Loram et al. 2008).

Local agricultural traditions and preferences may also influence composition of crops providing a specific ecosystem service, resulting in less species turnover between gardens in a single urban region (Barau et al. 2013). Participant agricultural background and local traditions have been closely linked to preferences for specific edible crops in urban agricultural spaces, suggesting that villages who share agricultural background will also share species compositions (Fu et al. 2006; Lubbe et al. 2011; Galluzzi 2012). Reduced road access and distance from local markets can further influence biodiversity patterns by reducing the need to grow cash crops, which can create unique patterns of species in rural villages (Abebe et al. 2013). Peri-urban villages, intermediate between rural and suburban villages, combine agricultural participation with intermediate income and market access and may contain gardens with both high edible and ornamental biodiversity which overlap compositionally with both suburban and rural gardens.

As resident needs change across a distance and socioeconomic gradient from city boundaries, so may garden management and species density. Variation in species-area relationships, the change in number of species with habitat area (Koellner and Schmitz 2006), is often indicative of community assembly processes in natural and human dominated ecosystems (Gotelli and Colwell 2001; Breuste et al. 2008). Studies in both European and Asian home gardens indicate a positive linear relationship between garden size and species biodiversity (Smith et al. 2005; Loram et al. 2008; Huai et al. 2011; Abebe et al. 2013). Unmet demand for provisioning ES may drive strong relationships between species and garden area within a specific village. Variation in species-area relationships between villages may result from differences in agricultural knowledge. Increased agricultural knowledge and greater plant needs in exurban villages may encourage complex garden structures in more rural regions where garden sizes are constrained (Airriess and Clawson 1994; Kumar and Nair 2004). In contrast suburban villages may have a more limited palate of species they can cultivate and may plant a similar number of species independent of available garden space. Such patterns of size invariant species planting has been shown in some French home gardens in densely population regions (Marco et al. 2008).

Our multi-scale agricultural study describes the vegetative composition and ecosystem services produced in home gardens in five villages in three urbanized regions, suburban,

peri-urban, and exurban, within the Beijing Municipality of China. These regions are organized along a distance gradient from the city, as isolation from urban resources impact income, population density, access to urban markets, and occupation; a pattern observed in cities generally and specifically in Beijing (McDonnell and Hahs 2008; Huai et al. 2011; Yunlai and Fengying 2009; Table 1). We focus on how the coupling between socioeconomic status, access to markets, and agricultural knowledge of residents in each region affects overall biodiversity and species uses (whether plants provide provisioning or cultural/aesthetic ES). Studying socioeconomic change across an urbanizing gradient provides a framework for incorporating residents into ecological system dynamics (Alberti et al. 2003; McDonnell et al. 2012; Boone et al. 2012).

Our study focuses on answering two distinct questions: 1) *How do biodiversity patterns change between different urbanized regions in Beijing?* 2) *How do ecosystem services provided by Beijing home gardens change with socioeconomic status and location-based preferences of gardeners?* In answering these questions we address hypotheses describing the selection and biodiversity across gardens in different urbanized regions and species uses within each garden. Our analyses provide data for comparisons with home gardens across the world to help quantify their overall contribution to urban biodiversity and ES.

Methods

Study area

The Beijing municipality of the northeast coast of China spans 16,800 sq. km, and has a population of over 20 million people, which is a 54% increase since 2001 (National Bureau of Statistics 2010; Beijing Bureau of Statistics 2012). Of these, 86 % of total residents reside in urbanized Beijing, and over 35% of the total population are migrants from other provinces (Beijing Bureau of Statistics 2012). Residents living in exurban areas in China earn less than half much as their urban counterparts (Zimmer and Kwong 2004), and are more likely to get

Table 1 Description of village level characteristics for five sampled villages. Population density and income per capita were obtained through the National Bureau of Statistics, while village density, % of people in agricultural jobs, and village area were established through visits to local government offices

| Variable	Urbanization region				
	Suburban	Peri-urban		Exurban	
Village abbreviation	SHZ	DXZ	XZY	XSY	NPY
Income/capita (yuan)	13,755	10,172	11,000	6,000	5,900
Distance from 5TH ring road (km)	14	30	32	59	61
Average temp (°C)	12.1	11.5	11.5	10.8	10.8
Population density (person/km)	1,262	381	455	61	32
Households in village	1,917	294	350	214	210
Area of village (m^2)	2795.5	2,574	1,775	19,000	8,960
% residents with agricultural jobs	9.67 %	28.5 %	32.4 %	40 %	37.9 %
Average garden size (m^2)	131	207	210	237	157
Gardens sampled	30	30	15	17	12

their income from farm activities (CCICCD 1996; Huai and Hamilton 2009). Beijing municipality contains some of the country's most productive agricultural land. The shape and management of agricultural land near Beijing has changed since agricultural reforms were instituted in 1978 (Yu et al. 1999). These reforms de-collectivized state land and instituted an individual household-based farming system (Lin 1992). Enactment of these reforms has encouraged more vegetable farming, and expanded both home gardens and multi-crop productivity throughout China (Yu et al. 1999; Baudry and Yu 1999; Ouyang et al. 2004). However, agricultural land is in decline, with the loss of over 545,000 ha of agricultural land near Beijing to urbanization in the last 20 years (Ho and Lin 2004). Urban growth policies in Beijing province include the replacement of courtyard centered villages with more compact modern housing, reducing local planting space in the process (Drew 2008; Kessel and Gillet 2011).

Data collection

Five different villages across a distance gradient from the border of Beijing, China were surveyed for home garden biodiversity and ES production. These villages were located in one of three different urbanized regions, defined by their distance from the fifth ring road (the city border) in Beijing (Fig. 1). For each region, villages representative of regional environmental and socioeconomic variability were chosen, based on population density, number of households, income, agricultural production, and elevation. We determined population density through the National Bureau of Statistics (NBS), while percent of people in agricultural jobs, and village area were established through visits to local government offices (Table 1). Income per capita was estimated through combined NBS estimates and interviews with village officials. Even so, many forms of income go unreported; NBS income data does not include income generated from household property (e.g. rentals) or unofficial income from crops sold from farmland (Sicular et al. 2007). Supplemental income was aggregated from previous interviews with local gardeners and officials, though village scale variation may be larger than the sample. Despite these limitations, our income estimates correspond well to other urban to rural estimates in China (Zimmer and Kwong 2004; Yunlai and Fengying 2009; Huai et al. 2011), and we believe they are appropriate for our analysis scale.

The closest village to Beijing, ShangZhuang (SHZ), was located less than 15 km from the city border, and was the only suburban village. Though relatively small in total area, SHZ is home to ~3,500 individuals and ~2,000 households and is typical of suburban development around Beijing (Table 1). Agricultural production is lowest in this village. Two villages, DongXinZhuang (DXZ) and XinZhouYing (XZY), were sampled in the peri-urban region, ~30 km from the city border. A higher percentage of residents from these two villages work in agriculture than in suburban villages and more of the village area is dedicated to agricultural use. At around 55 km from Beijing, near the base of the Yanshan mountains, we sampled two exurban villages, NiuPenYu (NPY) and XiaoShuiYu (XSY) (Fig. 1). These villages were large (9 and 19 km^2) and contained the lowest density of households. Near 40 % of villagers work in agricultural jobs, and both income per capita and cost of living are low.

To representatively sample gardens in each village, stratified random sampling procedures were followed to include a statistically robust number of gardens (Bartlett et al. 2001). The total number of households in each village was obtained through the national census and local government offices. We conducted focused interviews with city officials to estimate the number of households containing home gardens within each village. Our goal was to sample 5–10 % of the existing gardens within each village. On average, 42 % of households have home gardens. With a total of 2,985 households across all five villages (Table 1), we estimated that 1,254 contained home gardens. According to a statistical method for estimating adequate
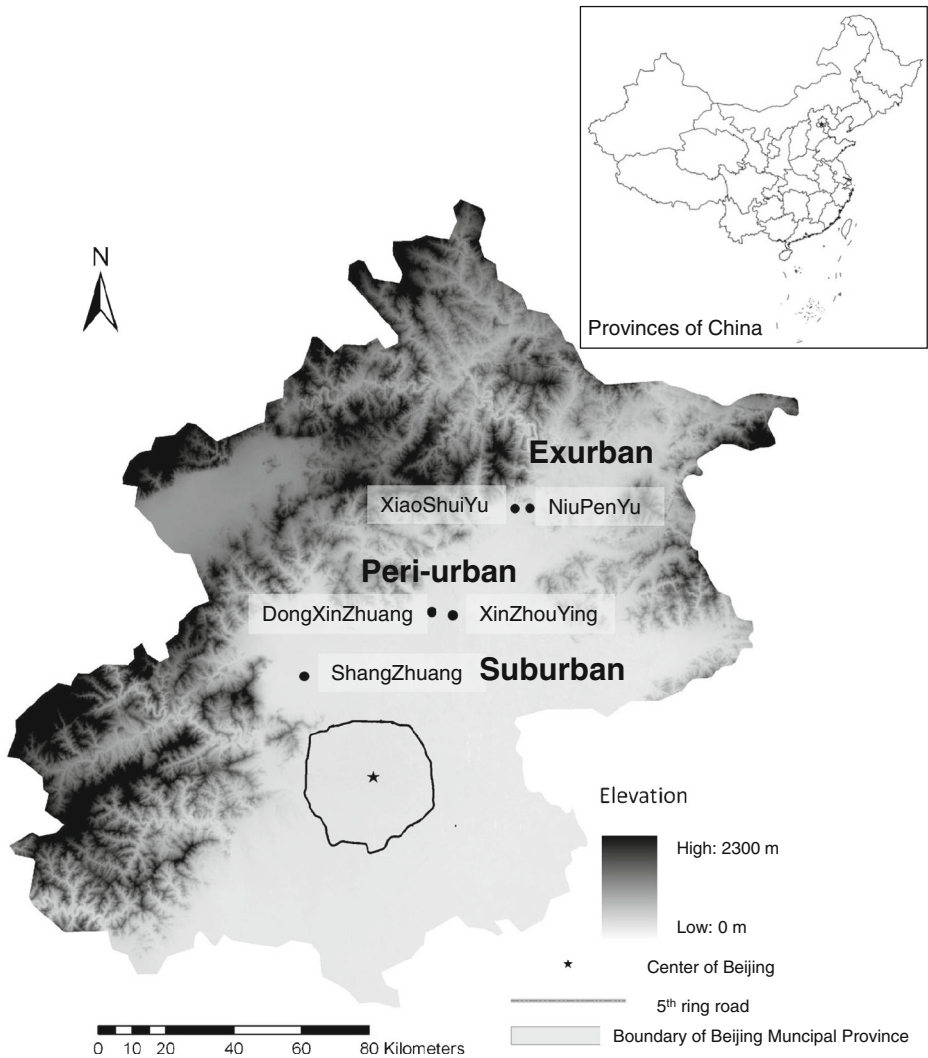
Fig. 1 Map of Beijing municipal district indicating locations of sampled villages in Beijing Municipal Province. Inset map shows borders of all provinces in China, with Beijing noted as a star. Elevation is indicated through *shading*. The *star* indicates the center of Beijing and the *circle* around it is the urban border (the 5th ring road). *Circles* indicate villages, and urbanization regions are indicated in *bold text* near villages

sample size in a given population (outlined in Bartlett et al. 2001), our sample size should be 108 total gardens (3 % margin of error, a=0.05). As each village varied in number of households, we visited between 15 and 30 gardens per village, for a total combined sample of 104 gardens, close to our target sample goal.

As villages do not keep accurate home garden census records, we visited home gardens opportunistically. Using maps of the villages, we visited each of the four quadrants of the village over the course of a few days. We walked the length of streets in that region and looked for residents at home. If a garden or intentional cultivation of any kind were present in that household courtyard, we asked for permission to survey their garden, regardless of crop

coverage. Residents were open to the survey, and our research group was only denied entry twice. We then asked each participant about households with active gardens in this village quadrant, effectively identifying new survey participants (a technique outlined in Russell 2006). For two of the villages, XZY and XSY, we were able to work more closely with village officials, who set up appointments with 5–10 households with gardeners. The remainder of the visited gardens were identified through interviews with these participants.

For each visited home garden, we estimated garden (delineated region used for planting) and parcel size (space owned by residents, including courtyard, shed, and house). As many dwellings had complex structures, gardens and parcels were difficult to measure directly. Instead, smaller identifiable structures were measured for reference (e.g. a 30 m^2 shed) and then full size was estimated by sight and interview with household members. All deliberately cultivated plants and trees were identified and percent cover of each species estimated. Larger tree canopies were measured on-site and smaller plant cover percentages were estimated visually. Species, not subspecies or specific varieties, were recorded, with a few exceptions. If different parts of the plant were used between varieties or the local use was different, we recorded them separately. For instance, some *Cucurbita pepo* subspecies were used as decorative gourds, and defined separately from food species. Residents were asked about the identity of any unknown species. Proper taxonomic identification for any unusual species was assured through photos and collection of voucher specimens for expert identification and archiving at the China Agricultural University herbarium. In addition, gardeners were asked about how each species was used, determining if each species provided provisioning or aesthetic/cultural ES. Use categories included edibles (E) and medicinals (M), both provisioning uses, and ornamentals (O), plants with cultural or aesthetic service value. In addition, we include an "Other" category (D) for less common provisioning services, which included shade, timber, fiber, fencing, or windbreaks. Many species had multiple uses and were noted once for each use, thus making the accumulated number of all individually used species greater than overall diversity.

Analysis

To compare garden biodiversity and abundance between villages, we conducted analyses of variance (ANOVA) at two size scales, garden and village (SPSS 16.0). Garden level biodiversity and abundance were compared between individual villages and between urbanization regions. These were further separated into number and abundance of each species use (edible, medicinal, ornamental, and other). Size of gardens and parcels was compared similarly at the village and region scale. In order to evaluate compositional differences between villages and urbanization regions, we conducted a principal components ordination using a program previously developed by Exequiel Ezcurra and used for biodiversity assessments (Altesor et al. 1998; Garcillán and Ezcurra 2003). The program solves both Correspondence Analysis (Hill 1973) and Principal Component Analysis (Noy-Meir 1973; Noy-Meir et al. 1975) as an eigenvector decomposition problem. The eigensolutions are calculated using the numerical algorithm proposed by Press et al. (2007). Garden similarity was compared within calculated 2D ordination space and individual species were projected onto the same ordination space.

We conducted a linear regression to assess the relationship of cover and biodiversity values to size of the garden plot and parcel (SPSS 16.0). These regressions were repeated for each garden, urbanization region, and all plant uses within them. Since parcel and plot size are intrinsically linked (a garden plot is limited by the overall size of the parcel), we first conducted a controlled regression to determine which had the most influence over biodiversity and cover. This controlled regression indicated that, while garden size is significantly

correlated to parcel size, all observed relationships between species-area and cover-area were only significant for garden size. Therefore, we report species-area relationships for garden size, not parcel size.

We constructed sample-based rarefaction curves, randomized and smoothed species accumulation curves, for each sampled region to compare α diversity (regional biodiversity), sampling adequacy, and species saturation. Rarefaction curves are produced by repeatedly re-sampling the pool of N samples, so measures of α biodiversity can be directly compared at any sampling intensity (Gotelli and Colwell 2001; Colwell et al. 2004). While we are confident that our sampling strategy has produced a representative sample of all possible garden-containing households (Bartlett et al. 2001), some garden species are likely missing. One difficulty in sampling managed vegetation is that even exhaustive sampling may not produce an asymptotic curve, indicating sufficient sampling effort. In addition, a rigorous comparison of rarefaction curves also requires well-defined confidence intervals, which, until recently, were less reliable, as they were based on sample size (Colwell et al. 2012). We use a new computational technique to extrapolate rarefaction curves (described in Colwell et al. 2012; EstimateS 9.0), which resamples observed data stochastically. This technique is more robust than analytical models, allowing estimates of the total number of gardens needed to reach asymptote, biodiversity at asymptote, and confidence intervals independent of original sample size. We extrapolated rarefaction curves to 90 gardens for each region to equalize regional comparisons and for comparison to other studies, which often include 100 or more gardens per urbanized region (Jaganmohan et al. 2012; Lubbe et al. 2012; Huai et al. 2011). We also calculated a species richness estimate, using the first order jackknife estimator, which minimizes bias and allows estimation of total species without an asymptotic species accumulation curve. This estimator is a function of rare species; with every rare (n) species found, the jackknife estimate is $1/n(n-1)$ more than the total number found (Heltshe and Forrester 1983), and is calculated based on observed samples. As we hypothesize species composition will be different across regions, rarefaction and extrapolation are shown separately for all species, edible species, and ornamental species at the regional scale.

β diversity, turnover between gardens, can also be estimated regionally using rarefaction curves modeled with a power law function – $y = Cx^z$—where C is a constant and z is the slope of the function (Koellner et al. 2004). The exponent, z, is a measure of β diversity in each region, as it describes the rate of species accumulation (Arita and Rodríguez 2002; Zhao et al. 2010; Clarke et al. 2013). The slope of z ranges from 0 to 1, with one indicating that there are no shared species between gardens in a given region (high β diversity) and zero indicating identical species in each garden in a given region (low β diversity). Power law functions were based on the extrapolated 90 garden sample created by EstimateS, and repeated for each region and plant use.

Results

Suburban gardeners have the highest income per capita and fewest jobs in agriculture, indicating financial and physical access to city resources, while exurban gardeners have the highest agricultural participation and lowest income per capita (Table 1). Village population and density is highest near the city and decreases towards exurban areas. Finally, climatic variables of average temperature and precipitation both decrease with distance from the city. Though a few very large gardens were found in exurban areas (>500 M^2), garden size generally ranged between 150 and 200 M^2. The similarity of sizes was supported by an ANOVA showing no significant differences in garden area between villages or urbanization regions ($p > 0.05$).

Overall, 278 distinct species and sub-species were found across the five villages, most of which were in the edible (100) or ornamental (152) use category (Table 2). Individual villages had between 76 and 163 species in total (Table 2). Exurban villages had similar numbers of edible species as peri-urban (70 and 79 respectively), despite having fewer sampled gardens (29 vs. 45 gardens). Suburban and peri-urban gardens contained more unique ornamentals found solely in that region (30 and 41 respectively) compared with exurban villages (7). While differences between medicinal and other species uses were not clear at the village scale, peri-urban gardens had the highest biodiversity of those uses as well.

Near 50 % of all ornamental species and 30 % of edible species were unique to one of the five villages. Diversity differences were more pronounced at the regional level, and peri-urban gardens had a higher number of species than suburban or peri-urban ($p<0.05$; Fig. 2b). Peri-urban gardens have high ornamental biodiversity similar to suburban gardens, while retaining high edible biodiversity, resulting in the highest garden scale biodiversity ($p<0.05$; Fig. 2a, b). For cover, no individual villages had significant differences between any use category of cover (Fig. 3a). When gardens were grouped into regions, edible cover was highest in exurban villages (Fig. 3b).

Over all 104 gardens, the species accumulation curve did not reach asymptote at the observed 278 species, even when extrapolated to 200 gardens, indicating species will further increase if more gardens were added (Fig. 4). This appeared to be due to the steady increase of ornamental species, which were estimated to increase from 152 to 181 species with the addition of 100 gardens (Fig. 4). In contrast, edible species diversity was near asymptote at 104 gardens, with only ten more species extrapolated for the addition of over 100 more gardens. Likewise, species accumulation curves did not reach asymptote individually in any of the three urbanized regions based on the 30–45 gardens sampled, (Fig. 5a). Extrapolated continuations of the rarefaction curves showed that at near 90 gardens per urbanized region, suburban and exurban gardens were nearing asymptote, at an estimated 240 and 159 species

Table 2 Number of species in each sampled village and each urbanization region, divided into plant uses. Unique species refers to those found only in that village or region. The number of edible species was similar between peri-urban and exurban gardens, while the number of ornamentals was similar between peri-urban and suburban gardens

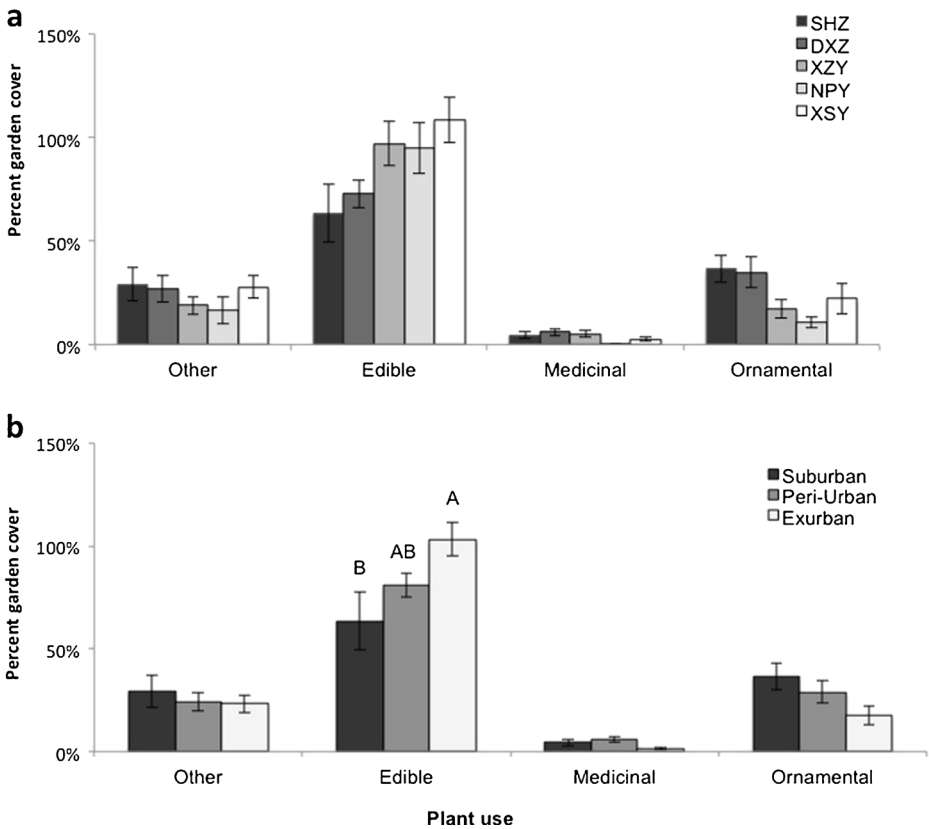| Villages										
Use category	SHZ		XZY		DXZ		NPY		XSY		All villages
	Total	Unique	Total	Unique	Total	Unique	Total	Unique	Total	Unique	
Other	11	3	12	1	13	5	7	2	10	1	23
Edible	54	5	58	6	64	11	51	6	54	3	100
Medicinal	15	3	11	3	16	7	3	0	9	3	27
Ornamental	92	30	57	9	87	26	20	2	26	4	152
Total	157	44	119	17	163	45	76	10	84	13	278

Urbanization region							
Use category	Suburban		Peri-Urban		Exurban		All villages
	Total	Unique	Total	Unique	Total	Unique	
Other	11	3	18	5	12	4	23
Edible	54	5	79	20	70	11	100
Medicinal	15	3	20	3	10	3	27
Ornamental	92	30	109	41	39	7	152
Total	157	44	203	69	117	22	278
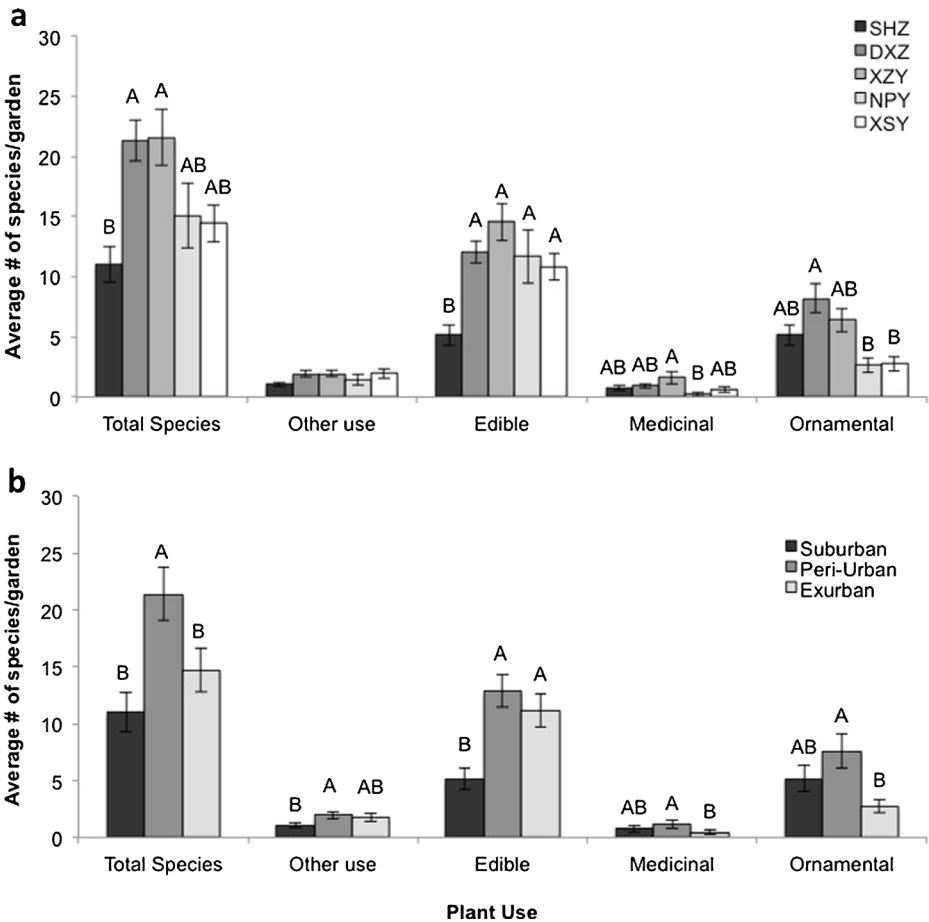
Fig. 2 Average number of species per garden in villages (**a**) and urbanization regions (**b**), separated into plant uses. Different *letters* in each use denote significant differences between average number of species per garden in each village or region (*p<0.05*). *Error bars* represent standard error

respectively (Fig. 5a; Table 3), while peri-urban regions were still increasing in biodiversity at 244 species. These numbers are supported by the first order jackknife indicator, which calculated that asymptote would be reached for suburban gardens at 236 (+/−16) species and exurban at 165 (+/−11), while peri-urban gardens would not reach asymptote until 270 (+/−9) species (Table 3). Confidence intervals for all species overlapped for suburban and peri-urban regions, indicating they did not have significantly different numbers of species; however, both regions had significantly more species than exurban gardens with non-overlapping confidence intervals (Fig. 5a). For edible species, all three urbanized regions had overlapping accumulation curve confidence intervals (Fig. 5b), indicating that each region was not significantly different in species richness, even though the per-garden comparisons show suburban gardens with fewer edible species (Fig. 2b). All three regions reach edible biodiversity saturation at 90 gardens, results supported by the first order jackknife indicator, as confidence intervals overlap with the calculated asymptotic range (Table 3). For ornamental species, suburban and peri-urban accumulation curves overlap in confidence intervals, indicating that they have similar

Fig. 3 Average percent species cover in villages (**a**) and urbanization regions (**b**), separated into plant uses. Different *letters* in each use denote significant differences in garden cover for that use at that scale ($p<0.05$). At the village scale (**a**), there were no significant cover differences within use categories. *Error bars* represent standard error

ornamental composition and exurban garden ornamental biodiversity is much lower (Fig. 5c), data supported by per garden comparisons (Fig. 2c). Extrapolation to 90 gardens does not show ornamental saturation in any region, (Fig. 5c) supporting the unique ornamental composition in each region (Table 2).

Suburban gardens had the highest overall beta (β) diversity (0.59) in comparison to peri-urban (0.47) and exurban (0.48) regions, indicating greater species turnover between suburban gardens (Table 3). β diversity was lower for edible species in peri-urban (0.34) and exurban regions (0.37), though notably less so between suburban gardens (0.46). In contrast, ornamental β diversity was the highest among all uses, indicating decreased species overlap in garden ornamentals in all regions (suburban: 0.67; peri-urban: 0.54; exurban: 0.64).

Peri-urban gardens showed compositional similarity with both suburban and exurban areas (Fig. 6), supporting diversity similarities observed for both suburban ornamentals and exurban edibles (Fig. 2). Ordination also indicated that gardens in exurban villages have the most similar species compositions to each other, as indicated by their similar location on the ordination space (Fig. 6). An ANOVA showed no difference between exurban gardens and XZY on axis 1, while DXZ and SHZ gardens occurred in different locations (Fig. 6). Along axis 2, the two peri-urban villages had no significant differences, while the cluster of XSY, an
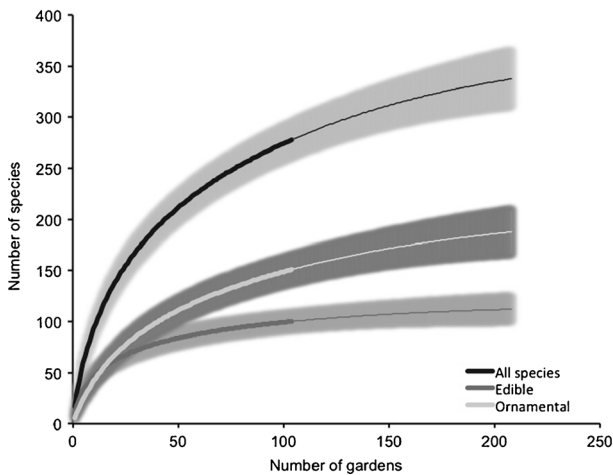
Fig. 4 Sample-based rarefaction curves for all 104 visited gardens. One curve represents all species found (*black line*), and others represent the most common species uses: Edible (*dark grey*) and Ornamental (*light gray*) species. Each *curve* has been extrapolated to 208 gardens using EstimateS 9.0 (Colwell et al. 2012); *thick lines* indicate observed species richness and *thin lines* of the same color represent extrapolation. *Shaded* region surrounding each line represent 95 % confidence levels

exurban village, was in a significantly different area than all others. Because of the high number of species found in only one or two gardens across an entire village, variation between gardens was too high to distinguish species groupings.

Garden and parcel sizes were not distinctly different between villages or urbanization regions. Across all village types, only exurban garden diversity was related to size ($r^2 = 0.440$, $p < 0.001$; Fig. 7). When broken down into use types, edible ($r^2 = 0.337$, $p < 0.001$), medicinal ($r^2 = 0.196$, $p = 0.016$), and ornamental ($r^2 = 0.154$, $p = 0.034$;) species all increased with garden size in exurban villages.

Discussion

The results of our intensive study provide comprehensive information on home garden biodiversity and species uses and their regional variation near a megacity of China. Such information is currently limited for cities in developing countries, although essential for the quantification of ecosystem services and human well being in locations of rapid urbanization (Cilliers et al. 2013; Jaganmohan et al. 2012; Lubbe et al. 2012). Our data show high α and β diversity across all villages, with distinct species composition for each urbanized region (Figs. 4 and 5; Table 3). Quantitative data on biodiversity and species cover across multiple urbanized regions may aid in local protection for agricultural land by making it competitive with other urban land uses (Zhang et al. 2009).

One important result of our study is that species uses and ES production in home gardens change across an urbanizing gradient, supporting a hierachy of need hypothesis. Poorer exurban communities with reduced access to urban markets are more likely to select garden species providing edible, medicinal, shade, and other provisioning services than more affluent suburban and peri-urban communities (Figs. 2b and 5). These plants may provide additional
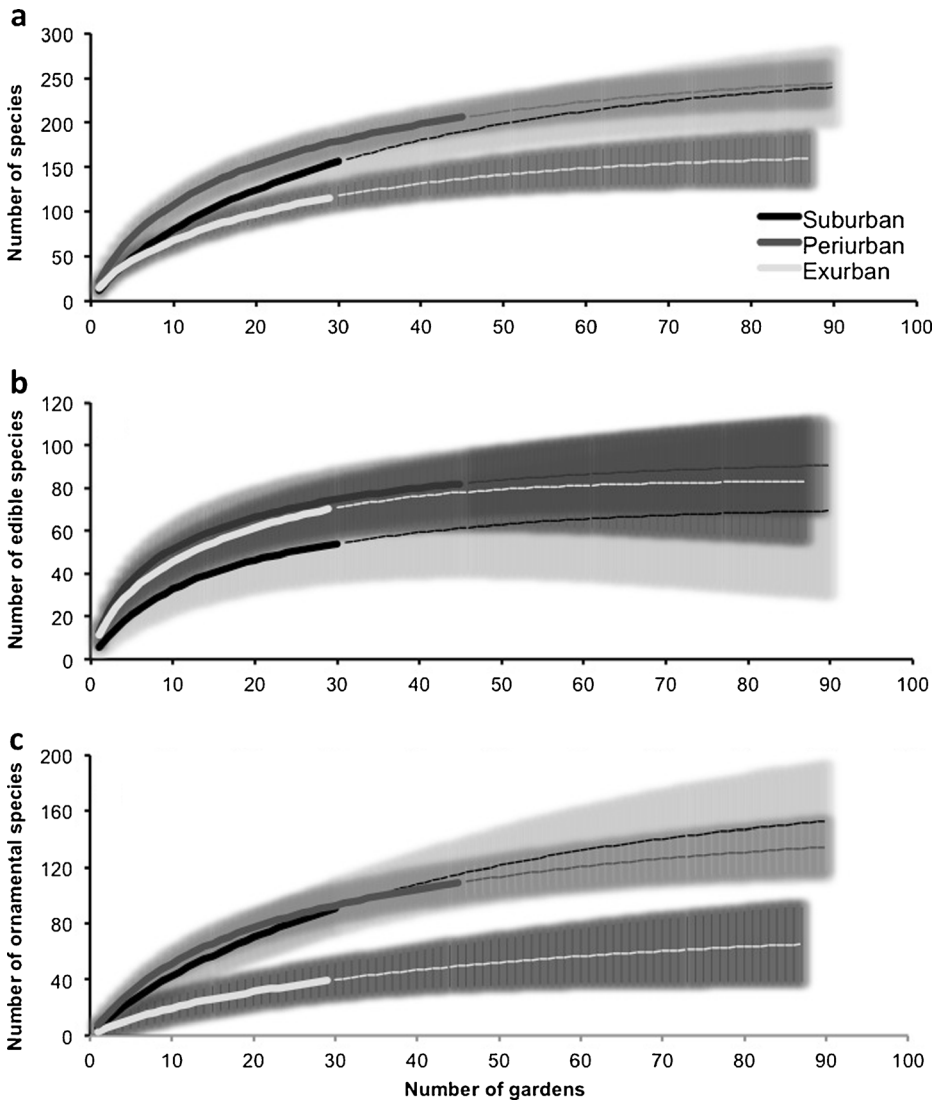
Fig. 5 Each panel represents one of the rarefaction curves from Fig. 4 divided into the three urbanized regions: All species (**a**), edible species (**b**), and ornamental species (**c**). These are each divided into, suburban gardens = *black line*; peri-urban = *dark grey line*; exurban = *light grey line*. *Shaded* regions surrounding each line represent 95 % confidence levels. Each *curve* has been extrapolated to 90 gardens. *Solid lines* indicate observed patterns and *dotted lines* indicate extrapolation. When confidence levels do not touch, that region is significantly different in species diversity than other regions. First order jack-knife estimates of diversity and β diversity slopes based on these data are reported in Table 3

income or improve gardener livelihoods through providing a basic need (Lubbe et al. 2010; Cilliers et al. 2012; Cilliers et al. 2013). The higher number of ornamental species and decreased edible cover in suburban and peri-urban gardens (Figs. 2 and 3) may be attributed to luxury investment in cultural and ornamental services as well as decreased provisioning needs. This shift from cultural to provisioning ES with distance from the city and declining

Table 3 Alpha and beta diversity estimations. The jack-knife estimate is analytically derived based on observed data, while the rarefaction estimate is based on extrapolated curves for 90 gardens in each region. β diversity is based on the power law relationships of extrapolated rarefaction curves ($y = Cx^z$), where z is a proxy for β diversity. Values can range from 0 to 1, with higher values indicating fewer overlapping species between gardens

| Urbanization region	Jack-knife estimated asymptotic diversity	Rarefaction extrapolation estimate	Exponent z (β diversity)
All species (Figs. 4 and 5a)			
All gardens ($n=104$)	371 (+/−11)	337 (+/−26)	0.4575
Suburban ($n=30$)	236 (+/−16)	240 (+/−38)	0.5934
Peri-urban ($n=45$)	269 (+/−9)	244 (+/−21)	0.4568
Exurban ($n=29$)	165 (+/−11)	159 (+/−27)	0.475
Edible species (Figs. 4 and 5b)			
All gardens	120 (+/−5)	112 (+/−12)	0.3277
Suburban	72 (+/− 4)	69 (+/−16)	0.4569
Peri-urban	99 (+/−4)	91 (+/−9)	0.3434
Exurban	91 (+/−6)	83 (+/−14)	0.3708
Ornamental species (Figs. 4 and 5c)			
All gardens	207 (+/−9)	188 (+/−21)	0.5341
Suburban	146 (+/−12)	153 (+/−33)	0.6721
Peri-urban	148 (+/−7)	134 (+/−17)	0.5414
Exurban	61 (+/−7)	65 (+/−22)	0.6436

socioeconomic status has been observed in home gardens across the world (Thaman et al. 2006; Bernholt et al. 2009; Lubbe et al. 2010; Cilliers et al. 2012).

Purposeful plant biodiversity was extensive in home gardens, with 278 species found across the three urbanized regions and 337 estimated in the extrapolated species accumulation curve of 200 gardens (Tables 2 and 3). Peri-urban and suburban regions had not reached asymptote at our sampling intensity, and were only approaching it at an extrapolated 90 gardens (Table 3; Fig. 5a), suggesting that species will further increase as more gardens are sampled. More
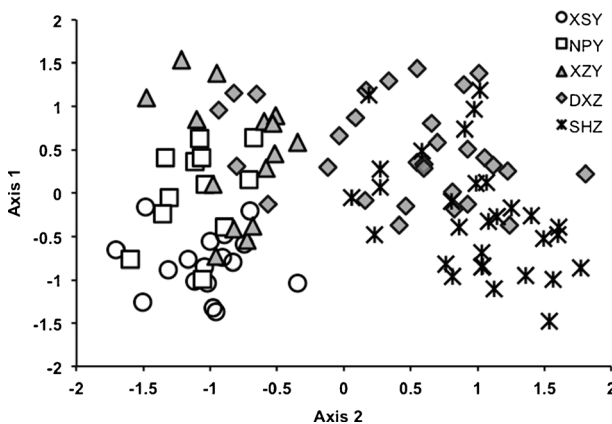
Fig. 6 Principal components orientation of garden scale biodiversity between villages. Each point represents the biodiversity of one garden. Exurban gardens are *white circles and squares*; Peri-urban gardens are *gray triangles and diamonds*; suburban gardens are *black stars*. The two axes plotted account for only 12 % of the variation between gardens, due to the large variation in species
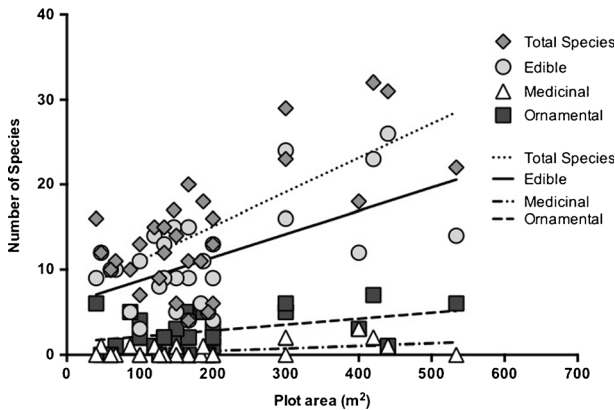
Fig. 7 Number of species per garden as a function of garden area. Only exurban gardens are depicted here. Garden area is positively related to total number of species (*diamonds*; $r^2=0.440$, $p<0.001$), number of edibles (*circles*; $r^2=0.337$, $p<0.001$), number of ornamentals (*squares*; $r^2=0.154$, $p=0.035$), and number of medicinals (*triangles*; $r^2=0.196$, $p=0.016$) in each garden. No other villages or regions showed species-area relationships

intensive studies than ours, with 100–300 gardens sampled per urbanization region, also did not reach species saturation (Lubbe et al. 2011; Cilliers et al. 2012; Jaganmohan et al. 2012). Species saturation is unlikely in managed garden systems, as species choices are only limited by the available plant pool at nurseries, which can range into thousands of species (Smith et al. 2005).

The main driving force behind our non-saturating species accumulation curves appears to be unique ornamental species; of the 152 ornamental species found in our survey, 78 of them were unique to a single region and most of those were only found in a single garden (Table 2). We also see this reflected in the high β diversity of ornamental species in peri-urban and suburban regions (Table 3; Fig. 5c). Indeed, the only urbanized region we estimated to reach species saturation at 90 gardens was in exurban villages (Fig. 5a), which have the highest edible coverage and biodiversity and are significantly lower in ornamental biodiversity (Figs. 2b and 3b). Since every region reached saturation in edible species (Fig. 5b), low β diversity within exurban gardens is unsurprising (Table 3). Likely, the need for provisioning ES in exurban villages translates to a reduced demand for cultural ES such as aesthetics, which drive α and β biodiversity (Cilliers et al. 2013). Though overall garden β diversity (0.45–0.56) is relatively high in comparison to temperate deciduous forests (0.2–0.4; Connor et al. 1983; Koellner et al. 2004), some highly urbanized areas have even greater heterogeneity (0.7–0.9; Clarke et al. 2013). We interpret the intermediate β diversity as a product of ES demand; though high ornamental novelty is desired, especially near the city (β=0.54–0.67; Qian et al. 2007), provisioning species are more constant, with lower turnover within a region (β=0.34–0.45), tempering garden species turnover.

Though our sample size of 104 gardens was a relatively small subset of total village area, (<20,000 m² total), an extensive biodiversity survey within Beijing found only 500 weedy and cultivated species in over 42,800 m² in the urban landscape (Wang et al. 2012) as compared to our 278. Overall, this dense biodiversity in home gardens is consistent with research from other cities showing that home garden biodiversity surpasses most other urban land uses (Lubbe et al. 2010, 2011). The biodiversity level we found is comparable to a study of 300 home gardens in rural India (*n*=258; Jaganmohan et al. 2012), and 100 low income urban gardens in Africa (*n*=270; Cilliers et al. 2013), and is three times as diverse as a recent survey of 15 villages in Southwestern China (Huai et al. 2011). In contrast, peri-urban and

urban regions of home gardens in other developing countries may have higher species biodiversity, as shown by a recent 100 garden survey in South Africa with over 800 cultivated species (Lubbe et al. 2011). Considering the high β diversity for ornamental species in our survey, regional valuation of cultural services and differences in socioeconomic status between our study region and those in past studies may account for the disparity in observed biodiversity patterns (Bernholt et al. 2009; Cilliers et al. 2012).

While food species diversities across urbanized regions overlap heavily in species accumulation curves (Fig. 5b), our ordination shows strong differences between suburban and exurban village species compositions, with peri-urban villages similar to both suburban and exurban villages (Fig. 6). We suggest these changes occur from linked local traditions and economic factors, as agricultural participation, ES demand, and socioeconomic factors all varied by both village and urbanized region. Similar compositional species changes have been observed in African home gardens, where the main food species grown in gardens changes from leafy vegetables near the city to grain crops in deep rural villages (Molebatsi et al. 2010). Isolation from major markets may further encourage distinct species biodiversity in exurban regions (Abebe et al. 2013). As a large disparity exists between food security in urban vs. exurban areas across China (Yunlai and Fengying 2009), exurban gardeners may cultivate species uncommon to commercial farms, as specialized provisioning species are harder to find outside of urban markets in China (Qian et al. 2007; Akkinfesi et al. 2010).

A relationship between individual home garden size and number of species has been observed in multiple countries, including other villages in China (Loram et al. 2008; Kabir and Webb 2009; Huai et al. 2011). Surprisingly, we only found a relationship between garden size and species diversity in exurban villages (Fig. 7). Reduced income in exurban regions coupled with reduced availability of food markets contributes to lowered food security, making productive gardens necessary to local food systems (Kabir and Webb 2009; Yunlai and Fengying 2009; Galluzzi et al. 2010). In addition, specific food needs fulfilled by each edible species cannot be substituted by replacement with other species (Peña 2006). A complex garden structure with multiple plant layers is usually observed in more rural regions (Michon and Mary 1994; Del Angel-Pérez and Mendoza 2004; Akkinfesi et al. 2010), a pattern reflected in our study where exurban gardens had edible species cover of over 100 % (Fig. 3). The high edible cover and complex vertical garden structure indicates that demand for species diversity in exurban regions may be higher than space constraints can support. With increased space, more species are planted to address demand, leading to the observed species-area relationship. For peri-urban and suburban gardens, species are not planted as intensely, indicating that space does not limit species choice. If a gardener desires ten food species and all ten species can be grown in the available space, increased increments of garden space may be used for expanding existing species, not adding new varieties. Other studies that show a consistent species-area relationship across all urbanized regions indicate that garden space is insufficient to support all local species needs (Kabir and Webb 2009; Loram et al. 2008; Albuquerque et al. 2005). Gardeners in exurban regions also have been shown to have a higher agricultural knowledge base than their urban counterparts, and are therefore more able to maintain a maximum species density in their gardens (Thaman et al. 2006; Airriess and Clawson 1994; Albuquerque et al. 2005). Therefore, exclusively exurban species-area relationships can be explained through a combination of the space-species demand mismatch and agricultural ability in exurban areas to maintain high species densities.

Conclusions and implications

Our research provides quantitative data on biodiversity, species abundance, and the ways participants use gardens to supplement their health and well-being. The mechanisms regulating garden biodiversity that we propose in this paper (hierarchy of need, local agricultural traditions, size of managed area) can be applied broadly to urban garden systems across the world, an essential part of advancing urban ecological science (McDonnell and Hahs 2013). We show that urban garden biodiversity shifts across different urbanized regions in Beijing according to a hierarchy of need. Gardeners change from cultivating aesthetically pleasing species (cultural ES) to more useful edible species (provisioning ES) with increasing isolation from the city and decreased socioeconomic status. Edible and ornamental composition also shifted, possibly due to cultural shifts between suburban and exurban villages. Surprisingly, we also show that the hierarchy of needs also influences species area relationships; low-income exurban communities may have greater demand for species than they have planting space, leading to a clear increase in species with any new increment of space.

Large-scale agriculture has outcompeted many small farmers in China, though they lack vegetable diversity and local varieties (Yunlai and Fengying 2009), and many varieties of crops in China are vanishing due to reduced traditional ecological knowledge (Pei et al. 2010; Huai and Hamilton 2009). Though our survey did not identify individual varieties of common vegetables, other studies have shown that home gardens can be germplasm banks for the conservation of local varieties (Huai et al. 2011; Levasseur and Olivier 2000). Measures and policies to encourage agricultural biodiversity in Chinese farmlands are sparse, and diversified cropping systems seen in home gardens are lacking in modern farms (Liu et al. 2011). Food production in Chinese gardens reduces the demand on commercial agriculture (Zhang et al. 2006) and may increase local food security (Wezel and Bender 2003; Huai et al. 2009). Our study highlights how biodiversity in home gardens changes along socioeconomic gradients, shifting from cultural to provisioning ES with decreased gardener income and access to important food resources.

References

Abebe T, Sterck FJ, Wiersum KF, Bongers F (2013) Diversity, composition and density of trees and shrubs in agroforestry homegardens in Southern Ethiopia. Agrofor Syst 87:1283–1293
Aguílar-Stoen M, Moe SR, Lucia Camargo-Ricalde S (2009) Home gardens sustain crop diversity and improve farm resilience in Candelaria Loxicha, Oaxaca, Mexico. Hum Ecol 37(1):55–77
Airriess CA, Clawson DL (1994) Vietnamese market gardens in New Orleans. Geogr Rev 84(1):16–31
Akkinfesi FK, Sileshi GW, Ajayi OC, Akinnifesi AI, de Moura EG, Linhares JFP, Rodrigues I (2010) Biodiversity of the urban homegardens of São Luís city, Northeastern Brazil. Urban Ecosyst 13:129–146
Alberti M, Marzluff JM, Shulenberger E, Bradley G, Ryan C, Zumbrunnen C (2003) Integrating humans into ecology: opportunities and challenges for studying urban ecosystems. Bioscience 53(12):1169–1179
Albuquerque UP, Andrade LHC, Caballero J (2005) Structure and floristics of homegardens in Northeastern Brazil. J Arid Environ 62:491–506

Altesor A, Di Landro E, May H, Ezcurra E (1998) Long-term species change in a Uruguayan grassland. J Veg Sci 9(1):173–180

Arita HT, Rodríguez P (2002) Geographic range, turnover rate and the scaling of species diversity. Ecography 25: 541–550

Barau AS, Ludin ANM, Said I (2013) Socio-ecological systems and biodiversity conservation in African city: insights from Kano Emir's Palace gardens. Urban Ecosyst 16:783–800

Bartlett JE, Kotrlik JW, Higgins CC (2001) Organizational research: determining appropriate sample size in survey research. Inf Technol Learn Perform J 19(1):43–50

Baudry J, Yu Z (1999) Landscape patterns changes in two subtropical Chinese villages as related to farming policies. Crit Rev Plant Sci 18(3):373–380

Bernholt H, Kehlenbeck K, Gebauer J, Buerkert A (2009) Plant species richness and diversity in urban and peri-urban gardens of Niamey, Niger. Agrofor Syst 77:159–179

BMBS (Beijing Municipal Bureau of Statistics) (2012) China Statistics Press, Beijing. (in Chinese)

Boone CG, Cook E, Hall SJ, Nation ML, Grimm NB, Raish CB, Finch DM, York AM (2012) A comparative gradient approach as a tool for understanding and managing urban ecosystems. Urban Ecosyst 15:795–807

Breuste J, Niemela J, Snep RPH (2008) Applying landscape ecological principles in urban environments. Landsc Ecol 23:1139–1142

China National Committee for Implementing the United Nations Convention to Combat Desertification (CCCICCD) (1996) China country paper to combat desertification. China Forestry Publishing House, Beijing, pp 18–31

Cilliers S, Siebert S, Davoren E, Lubbe R (2012) Social aspects of urban ecology in developing countries, with an emphasis on urban domestic gardens. In: Richter M, Weiland U (eds) Applied urban ecology: a global framework, 1st edn. Blackwell Publishing Ltd

Cilliers S, Cilliers J, Lubbe R, Siebert S (2013) Ecosystem services of urban green spaces in African countries—perspectives and challenges. Urban Ecosyst (September) 1–22

Clarke LW, Jenerette GD, Davila A (2013) The luxury of vegetation and the legacy of tree biodiversity in Los Angeles, CA. Landsc Urban Plan 116:48–59

Colwell RK, Mao CX, Chang J (2004) Interpolating, extrapolating, and comparing incidence-based species accumulation curves. Ecology 85(10):2717–2727

Colwell RK, Chao A, Gotelli NJ, Lin S-Y, Mao CX, Chazdon RL, Longino JT (2012) Models and estimators linking individual-based and sample-based rarefaction, extrapolation, and comparison of assemblages. J Plant Ecol 5:3–21

Connor EF, McCoy ED, Cosby BJ (1983) Model discrimination and expected slope values in species-area studies. Am Nat 122:789–798

Davies ZG, Fuller RA, Loram A, Irvine KN, Sims V, Gaston KJ (2009) A national scale inventory of resource provision for biodiversity within domestic gardens. Biol Conserv 142(4):761–771

Del Angel-Pérez AL, Mendoza BMA (2004) Totonac homegardens and natural resources in Veracruz, Mexico. Agric Hum Values 21(4):329–346

Drew J (2008) In Beijing, no answer to the bulldozer. Washington Post, Saturday, April 2008. Online available at <http://www.washingtonpost.com/wp-dyn/content/article/2008/04/25/AR2008042503503.html>

Fu Y, Guo H, Chen A, Cui J (2006) Household differentiation and on-farm conservation of biodiversity by indigenous households in Xishuangbanna, China. Biodivers Conserv 15:2687–2703

Galluzzi G (2012) Agrobiodiversity and protected areas: another approach to synergies between conservation and use?. In: On-farm conservation of neglected and underutilized species: status, trends and novel approaches to cope with climate change – Proceedings of the International Conference, Friedrichsdorf, Frankfurt, 14-16 June 2011

Galluzzi G, Eyzaguirre P, Negri V (2010) Home gardens: neglected hotspots of agro-biodiversity and cultural diversity. Biodivers Conserv 19(13):3635–3654

Garcillán PP, Ezcurra E (2003) Biogeographic regions and beta-diversity of woody dryland legumes in the Baja California peninsula. J Veg Sci 14(6):859–868

Gotelli NJ, Colwell RK (2001) Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. Ecol Lett 4:379–391

Heltshe JF, Forrester NE (1983) Estimating species richness using the jackknife procedure. Biometrics 39(1):1–11

Hill MO (1973) Reciprocal averaging: an eigenvector method of ordination. J Ecol 61:237–249

Ho SPS, Lin GCS (2004) Non-agricultural land use in post-reform China. China Q 179:758–781

Hope D, Gries C, Zhu WX, Fagan WF, Redman CL, Grimm NB, Nelson AL, Martin C, Kinzig A (2003) Socioeconomics drive urban plant diversity. Proceedings of the National Academy of Sciences of the United States of America 100:8788–8792

Huai H, Hamilton A (2009) Characteristics and functions of traditional homegardens: a review. Front Biol China 47:151–157

Huai H, Xu W, Wen G, Bai W (2011) Comparison of the Homegardens of eight cultural groups in Jinping County, Southwest China. Econ Bot 65(4):345–355

Jaganmohan M, Vailshery LS, Gopal D, Nagendra H (2012) Plant diversity and distribution in urban domestic gardens and apartments in Bangalore, India. Urban Ecosyst 15:1–15. doi:10.1007/s11252-012-0244-5

Kabir ME, Webb EL (2009) Household and homegarden characteristics in southwestern Bangladesh. Agrofor Syst 75:129–145

Kessel J, Gillet K (2011) The fate of old Beijing: the vanishing hutongs. Online documentary, produced by China Green. Available at: http://sites.asiasociety.org/chinagreen/feature-hutong/

Kinzig AP, Warren P, Martin C, Hope D, Katti M (2005) The effects of human socioeconomic status and cultural characteristics on urban patterns of biodiversity. Ecology and Society 10(1):23

Kirkpatrick JB, Daniels GD, Zagorski T (2007) Explaining variation in front gardens between suburbs of Hobart, Tasmania, Australia. Landsc Urban Plan 79:314–322

Koellner T, Schmitz OJ (2006) Biodiversity, ecosystem function, and investment risk. Bioscience 56:977–985

Koellner T, Hersperger AM, Wohlgemuth T (2004) Rarefaction method for assessing plant species diversity on a regional scale. Ecography, 27:532–544

Kumar BM, Nair PKR (2004) The enigma of tropical homegardens. Agrofor Syst 61:135–152

Levasseur V, Olivier A (2000) The farming system and traditional agroforestry systems in the Maya community of San Jose, Belize. Agrofor Syst 49:275–288

Lin JY (1992) Rural reforms and agricultural growth in China. Am Econ Rev 82(1):34–51

Liu Y, Duan M, Yu Z (2011) Agricultural landscapes and biodiversity in China. Agric Ecosyst Environ 166:46–54

Loram A, Thompson K, Warren PH, Gaston KJ (2008) Urban domestic gardens (XII): the richness and composition of the flora in five UK cities. J Veg Sci 19:321–330

Lubbe CS, Siebert SJ, Cilliers SS (2010) Political legacy of South Africa affects the plant diversity patterns of urban domestic gardens along a socio-economic gradient. Sci Res Essays 5:2900–2910

Lubbe CS, Siebert SJ, Cilliers SS (2011) Floristic analysis of domestic gardens in the Tlokwe City Municipality, South Africa. Bothalia 41(2):351–361

Marco A, Dutoit T, Deschamps-Cottin M, Mauffrey JF, Vennetier M, Bertaudiere Montes V (2008) Gardens in urbanizing rural areas reveal an unexpected floral diversity related to housing density. C R Biologies 331(6):452–465

McDonnell MJ, Hahs AK (2008) The use of gradient analysis studies in advancing our understanding of the ecology of urbanizing landscapes: current status and future directions. Landsc Ecol 23(10):1143–1155

McDonnell MJ, Hahs AK (2013) The future of urban biodiversity research: moving beyond the 'low-hanging fruit'. Urban Ecosyst 16:397–409

McDonnell MJ, Hahs AK, Pickett STA (2012) Exposing an urban ecology straw man: critique of Ramalho and Hobbs. Trends Ecol Evol 27:255–256

Michon G, Mary F (1994) Conversion of traditional village gardens and new economic strategies of rural households in the area of Bogor, Indonesia. Agrofor Syst 25:31–58

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: synthesis. Island Press, WA, 155 pp

Mitchell R, Hanstad T (2004) Small homegarden plots and sustainable livelihoods for the poor. LSP working paper no.11. Food and agriculture organization of the United Nations, Rome

Molebatsi LY, Siebert SJ, Cilliers SS, Lubbe CS, Davoren E (2010) The Tswana tshimo: a homegarden system of useful plants with a particular layout and function. Afr J Agric Res 5(21):2952–2963

National Bureau of Statistics, China (2010) Online available at: http://www.stats.gov.cn/english/statisticaldata/yearlydata/

Nguyen MLT (2003) Comparison of food plant knowledge between urban Vietnamese living in Vietnam and in Hawai'i. Econ Bot 57(4):472–480

Noy-Meir I (1973) Data transformation in ecological ordination: I. Some advantages of non-centering. J Ecol 61:329–341

Noy-Meir I, Walker D, Williams WT (1975) Data transformation in ecological ordination: II. On the meaning of data standardization. J Ecol 63:779–800

Ouyang J, Song C, Yu Z, Zhang F (2004) Farm households choice of land use type and its effectiveness on land quality and environment in Huang-Huai-Hai Plain. J Nat Resour 19(1):1–11 (In Chinese)

Pei S, Hamilton AC, Yang L, Huai Y, Yang Z, Gao F, Zhang Q (2010) Conservation and development through medicinal plants: a case study from Ludian (Northwest Yunnan, China) and presentation of a general model. Biodivers Conserv 19:2619–2636

Peña D (2006) Toward a critical political ecology of Latina/o urbanism. The Acequia Institute. Electronic document <http://www.acequiainstitute.org/researchreports.html>

Press WH, Teukolsky SA, Vetterling WT, Flannery BP (2007) Numerical recipes 3rd edition: the art of scientific computing, 3rd edn. Cambridge University Press, Cambridge

Qian T, Joyce D, He S (2007) China's ornamentals industry is in 'full bloom'. Centre for Native Floriculture Report. March, 2007. 6 pp

Russell BH (2006) Research methods in anthropology: qualitative and quantitative methods, 2nd edn. Altamira Press, Lanham

Sicular T, Ximing Y, Gustafsson B, Shi L (2007) The urban-rural gap and income inequality in China. Rev Income Wealth 53(1):93–126

Smith RM, Gaston KJ, Warren PH, Thompson K (2005) Urban domestic gardens (V): relationships between landcover composition, housing and landscape. Landsc Ecol 20:235–253

Thaman RR, Elevitch CR, Kennedy J (2006) Urban and homegarden agroforestry in the Pacific Islands: current status and future prospects. In: Kumar BM, Nair PKR (eds) Tropical homegardens: a time-tested example of sustainable agroforestry. Advances in agroforestry, vol 3. Springer, Dordrecht, pp 25–41

Wang H, MacGregor-Fors, López-Pujol J (2012) Warm-temperate, immense, and sprawling: plant diversity drivers in urban Beijing, China. Plant Ecol 213:967–992

Wezel A, Bender S (2003) Plant species diversity of homegardens of Cuba and its significance for household food supply. Agrofor Syst 57:37–47

Yu ZR, Baudry J, Zhao BP, Zhang H, Li SQ (1999) Vegetation components of a subtropical rural landscape in China. Crit Rev Plant Sci 18(3):381–392

Yu ZR, Liu Y, Axmacher JC (2006) Field margins as rapidly evolving local diversity hotspots for ground beetles (Coleoptera : Carabidae) in northern China. Coleopt Bull 60(2):135–143

Yunlai X, Fengying N (2009) A report on the status of China's food security. China Agricultural Science and Technology Press, Beijing

Zhang K, Li X, Zhou W, Zhang D, Yu Z (2006) Land resource degradation in China: analysis of status, trends and strategy. Int J Sustain Dev World Ecol 13(5):397–408

Zhang F, Cai J, Liu G (2009) How urban agriculture is reshaping peri-urban Beijing? Open House Int 34(2):15–24

Zhao J, Ouyang Z, Zheng H, Zhou W, Wang X, Xu W, Ni Y (2010) Plant species composition in green spaces within the built-up areas of Beijing, China. Plant Ecol 209:189–204

Zimmer Z, Kwong J (2004) Socioeconomic status and health among older adults in rural and urban China. J Aging Health 16(1):44–70