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RESEARCH ARTICLE

Biodiversity and direct ecosystem service regulation in the community gardens of Los Angeles, CA

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Abstract
Context Urban community gardens are globally prevalent urban agricultural areas and have the potential to fulfill human needs in impoverished neighborhoods, such as food security and access to open space. Despite these benefits, little research has been conducted evaluating environmental and socioeconomic factors influencing community garden plant biodiversity and ecosystem services (ES).
Objective Our study investigated the drivers of managed plant richness, abundance, and ES production in community gardens across Los Angeles County, CA from 2010 to 2012 at regional, garden, and plot scales.
Methods Fourteen community gardens were visited in the summers of 2010–2012 for comprehensive species surveys across regional, garden, and plot scales. We compared biodiversity to household income, plot size, and gardener ethnicity.
Results In total, 707 managed plant species were recorded in summer surveys over a 3-year period. Ornamental plant richness increased with neighborhood income, while edible and medicinal richness increased with size of garden plots. Gardener ethnicity also influenced the composition of managed species, especially edible species.
Conclusions We explain these patterns through a hierarchy of needs framework; gardeners preferentially plant species progressively less connected to human need. Ornamental plant increases in high-income regions may be explained by their requirement for financial investment and maintenance time. Cultural and provisioning ES are important for immigrant populations, resulting in ethnically distinct crop assemblages. Finally, distinct species–area relationships imply high demand for food abundance and biodiversity. Our quantitative results indicate that community gardens contribute to a biologically diverse urban ecosystem and provide valued ecosystem services in food insecure regions.

Keywords Hierarchy of need · Beta diversity · Species–area relationship · Socioeconomics · Urban agriculture · Food security

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Introduction

Urban gardening has been integral to city life throughout the world for thousands of years (Fedick 1996; Hynes 1996; Smith et al. 2006; Stark and Ossa 2007). Globally, private gardens and peri-urban agriculture within metropolitan regions currently range between 16 % (Stockholm, Sweden: Colding et al. 2006) and 36 % (Dunedin, New Zealand: Mathieu et al. 2007) of total land area. One common type of urban agriculture is the community garden, defined as urban agricultural land managed by multiple residents (Jackson et al. 2013; Lawson and Drake 2013). Recent surveys estimate 10,000 community gardens are functioning throughout the U.S. with more than 1 million participants (Lawson and Drake 2013). With recent rapid increases in urban expansion, community gardens may act as oases of functional biodiversity in urban landscapes dominated by impervious surfaces and lacking in native biodiversity (Colding et al. 2006; Gaston and Gaston 2011).

Community gardens are important sources of direct, benefits directly experienced by people, and indirect, processes which lead to benefits, ecosystem services (ES) (Splash 2008; MEA 2005). Given that urban areas and their residents are increasing, with projections of more the 2.5 billion residents by 2015, (a 64 % increase from current distributions; UNDESA 2014), preserving urban biodiversity and ES production, even from exotic plant communities such as community gardens, is increasingly important to overall human health and well-being (Grimm et al. 2008; Smith et al. 2013). Direct ES from gardens may be provisioning, such as edible crop production (Alaimo et al. 2008) or cultural, such as aesthetics (Smith et al. 2013). Indirect ES include processes not directly based on cultivation, such as aiding pollinators (Matteson et al. 2008), mitigation of the urban heat island (Jenerette et al. 2011), and pollution reduction (Manes et al. 2012). Though research on community gardens has been increasing, the majority of studies have been qualitative and descriptive (Draper and Freedman 2010; Guitart et al. 2012). Our study addresses this knowledge gap, focusing on how garden biodiversity and ES throughout an urban landscape change across spatial scales according to the needs and values of residents from different economic and cultural backgrounds.

Community gardens feature extensive social and biological diversity, whose dynamics depend on the interaction between human desires and perceptions with biological processes and products, also known as a coupled human and natural system or CHaNS (Liu et al. 2007). Each of 10–150 sub-sections (plots) in a garden is individually maintained for species selection, soil preparation, and applications of fertilizers and irrigation. Surveying multiple gardens allows for quantification of biodiversity at three different ecological scales (Anderson et al. 2011): α (alpha diversity: individual plot scale), γ (gamma diversity: whole garden scale) and β diversity (turnover between plots in a single garden). Variation in biodiversity across these scales may be influenced by multiple interacting factors including management, neighborhood income, gardener social background, ES demand, and planting area.

Economic and social factors have been widely shown to influence plant biodiversity in managed landscapes. According to the well-established "luxury effect," urban plant biodiversity generally increases with residential income (Hope et al. 2003; Kinzig et al. 2005; Peña 2005; Cocks 2006). One framework for better understanding why economics influences biodiversity and direct ES is a hierarchy of needs, where ES are expected to be organized by needs progressively less connected immediately to survival (Lubbe et al. 2011; Clarke et al. 2013; Wu 2013). Financial resources necessary for investment in garden maintenance and purchase of purely ornamental species is dependent on the economic status of individual gardeners (Pickett et al. 2011; Lawson and Drake 2013). In large metropolises, median family income varies widely across regions, affecting local garden resources and demand (Jackson et al. 2013). Low-income gardeners may have unmet nutritional and culturally specific food needs that focus their output on edible species, while higher income gardeners may have their food needs met commercially and therefore select more ornamentals that fulfill aesthetic desires (Gaston and Gaston 2011; van Heezik et al. 2013).

In addition, a socio-cultural hypothesis predicts that the set of food, medicinal, and ornamental species planted in a garden will be distinct to the participant's cultural background and country of birth, due to cultural socialization and agricultural experience. Though all gardeners may share the same basic ES needs (food, aesthetic beauty, medicines), the palette of species valued for services varies across cultures (Fraser and Kenney 2000; Kinzig et al. 2005; Wakefield et al. 2007). Variation in ethnic diversity and high immigrant participation in gardens across urban regions potentially contributes to proliferation of culturally specific crops in gardens (Gottlieb 2006; Wakefield et al. 2007). Immigrant gardeners may also be more likely to come from agricultural regions that have strong gardening traditions, which may contribute to high crop density in gardens and ethnic crop composition (Barthel et al. 2010; Minkoff-Zern 2012).

Biodiversity variation across scales may also be linked to production of ES demanded by gardeners. Crops that supply culturally important provisioning services, such as food or medicine, may be planted for abundance, not diversity (Cilliers et al. 2012). Edible β diversity may be low in gardens where multiple participants value the same food species. In contrast, residents may cultivate a variety of unique ornamental plants to express individuality (Kaplan and Herbert 1987; Marco et al. 2008), creating extensive aesthetic β diversity. This high β diversity in ornamentals may encourage higher biodiversity with each progressive year of cultivation, due to participant turnover and legacies left by previous gardeners. A legacy hypothesis predicts that older, well-established gardens will be more bio-diverse than more recently established gardens due to legacies of species from previous managers, similar to biodiversity legacies observed across entire cities (Larsen and Harlan 2006; Pickett et al. 2011; Clarke et al. 2013).

Separate from socio-cultural influences, a fundamental ecological relationship explaining biodiversity is the species–area relationship (Lawton 1999; Koellner and Schmitz 2006). Some studies have shown a positive relationship between domestic garden size and species biodiversity (Smith et al. 2005; Loram et al. 2008; Huai et al. 2011), although this relationship is not always observed (Albuquerque et al. 2005; Clarke et al. 2014b). With increased space, more species are planted to address ES demands, leading to a strong species–area relationship, a pattern also described in home gardens (Loram et al. 2008) and family subsistence home gardens (Méndez et al. 2001; Kabir and Webb 2009), though not in larger farms (Blanckaert et al. 2007). Our modified species–area hypothesis predicts that garden species diversity will be linked to plot size, the scale of individual gardener choice, in individually-based gardens if ES demands exceed local space available for planting.

Our study investigated temporal and spatial-scale variation of biodiversity and ES production across fourteen community gardens in Los Angeles (LA), CA for 3 years. Through this study, we ask, *what factors regulate community garden plant biodiversity, abundance and their direct ecosystem service production?* Our overall aim is to quantify the biodiversity of LA community gardens and establish important economic, social, and biophysical factors influencing garden biodiversity, composition, and plant uses contributing to direct ecosystem services. We expect interactions between different mechanisms affecting biodiversity and direct ES—garden management style, socioeconomics, gardener ethnicity, species–area relationships—will create complex patterns of vegetation diversity and direct ES production. Our research activities may lead to better understanding of ES production in impoverished urban regions and improved urban sustainability through policy change in support of urban agriculture.

Methods

Study area

The socio-ecological heterogeneity of LA provides a useful site to study variability among community gardens. There are 99 officially recognized community gardens across LA, 60 % of which are set in low-income neighborhoods with high immigrant populations (Fig. 1). Over 30 % of LA County's population is foreign-born, with 45 % of the population of Hispanic descent (U.S. Census Bureau 2010). Neighborhood median household income ranges widely from $9,000 to $200,000. Low-income neighborhoods in LA have some of the highest immigrant and minority concentrations in the entire U.S. (U.S. Census Bureau 2010). Impoverished neighborhoods in LA are classified as food deserts, areas of reduced access to affordable and healthful food options (USDA 2014). These food poor regions have only grocery store per 46,000 residents, as compared to one per 20,000 in more affluent regions (Shaffer 2002). These food deserts are intensified by reduced transportation options and high unemployment rates, leading to increased health issues among low-income residents (Sharkey et al. 2009; Azuma et al. 2010).

Field methods

Beginning in 2010, we selected 14 community gardens within Los Angeles County for inclusion in this study. Gardens were chosen from an initial a pool

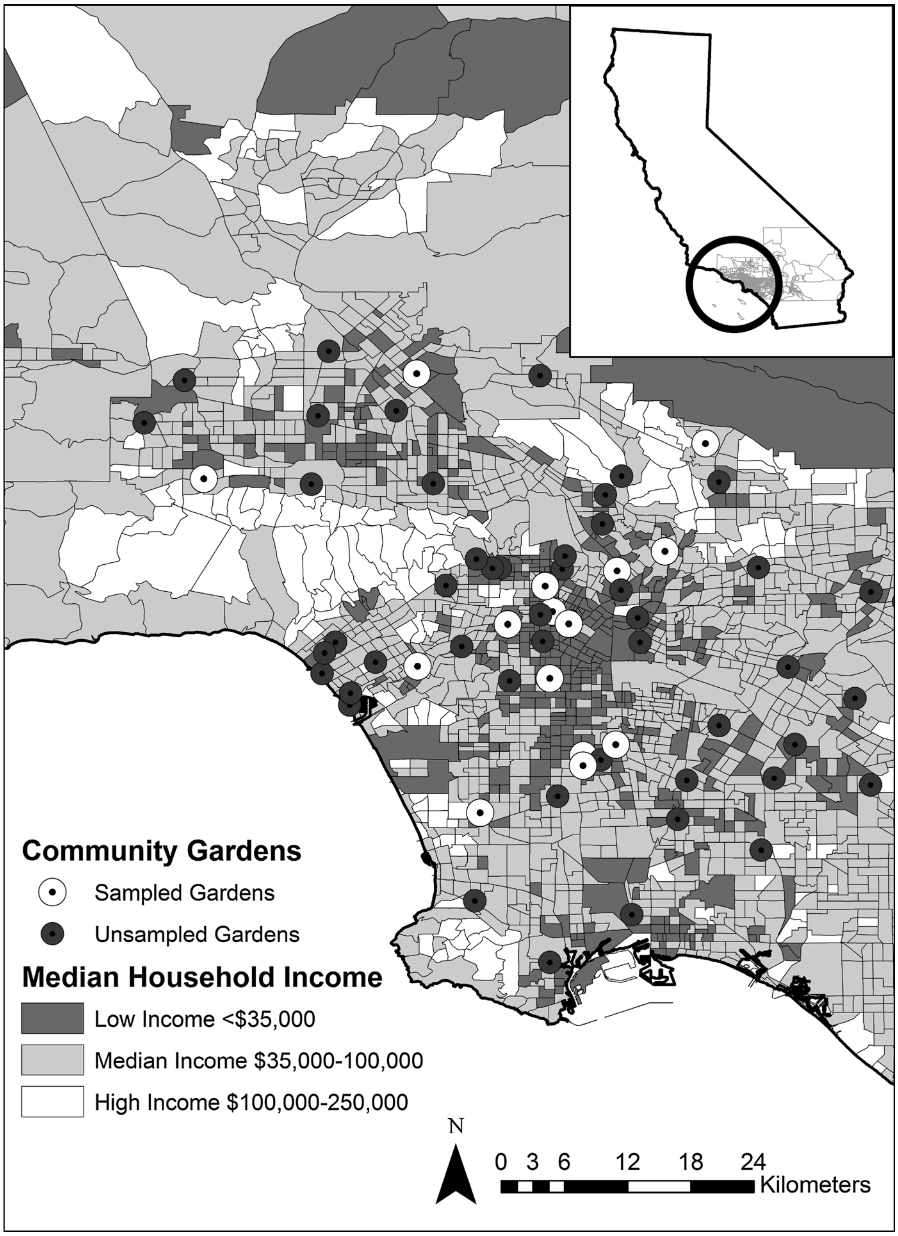

Fig. 1 Map of Los Angeles County showing census tract boundaries (*background lines*) and median household income variation (*dark gray* is low income, *white* is high income, and *light gray* indicates moderate income). Income data is based on 5-year estimates from the American Community Survey. The *circles* indicate the location of 99 community gardens in Los Angeles County, with *white circles* indicating surveyed locations and *dark gray circles* for all other gardens

of 25 randomly chosen gardens, and included based on their willingness to participate in plant surveys and continued interest in the research. These gardens were located in neighborhoods with median incomes between \$25,000 and \$90,000, range in size between 400 and 10,000 m^2, and were established between 1963 and 2009 (Table 1). Through informal interviews with managers and interaction with garden participants, we identified the major ethnic groups that were part of each garden. Seven selected gardens had primarily or exclusively Hispanic immigrant participants from Mexico, Guatemala, El Salvador, and Costa Rica. One garden had a majority of Korean immigrants. Together, these 8 community gardens were categorized as "immigrant" gardens. The remaining 6 had a majority of U.S. born residents, and were categorized as "non-immigrant" gardens. Of these, one garden was made up exclusively of

Table 1 Descriptive statistics for all gardens, including tested factors of management style, ethnicity, garden age, median family income, and area of gardens and plots

Garden	Management	Ethnicity	Year founded	Income	Garden area (m^2)	Plot area (m^2)	Plots	Gardeners
IMM1	Individual	Asian	1988	$30,558	1,440	46.46	32	32
IMM2	Individual	Hispanic	1999	$30,558	672	4.5	19	16
IMM3	Individual	Hispanic	2007	$49,006	4,500	11.88	60	75
IMM4	Individual	Hispanic	1999	$29,927	819	9	26	25
IMM5	Individual	Hispanic	1989	$26,757	852	5.7	34	27
IMM6	Farm	Hispanic	1994	$25,161	9,520	58.34	118	150
IMM7	Farm	Hispanic	1979	$53,150	2,006	37	44	40
IMM8	Farm	Hispanic	2006	$25,161	23,070	135	69[a]	69[a]
NIMM1	Individual	Mixed	2004	$82,676	10,117	60	57	133
NIMM2	Individual	Mixed	2009	$45,478	930	7	32	32
NIMM3	Individual	Mixed	1989	$29,904	900	4.5	24	11
NIMM4	Individual	Mixed	1963	$70,774	448	17.5	16	16
NIMM5	Farm	Mixed	1996	$89,946	2,244	52.63	25	20
NIMM6	Farm	African-American	1965	$25,161	6,120	85	44	60

Where number of gardeners exceeded number of plots, it meant that gardeners subdivided their plots with others or shared the work with family members

[a] There were over 200 plots, only a subsample of 69 was sampled through a random stratified sampling (5–10 plots per garden subsection)

African-Americans who immigrated to LA from the American Southeast. We noted this ethnic group separately from others due to the strong location-based origin of these gardeners. Ethnicity and immigrant status were used as proxies of cultural background to test our hypotheses of culture influencing garden species composition.

The management of community gardens may be more individually focused, with each managed subsection benefitting a single family, or more communally focused, where production across plots is shared between multiple participants (Jackson et al. 2013). Therefore, we categorized each garden by management style. Nine gardens were identified as individually-based gardens, where 1–2 participants manage small ($\sim$4.5–60 m^2) plots and the produce is not sold or used to support multiple families. In communally-based community gardens, crop production is shared between participants and marketable species are often sold or donated, as in church or school gardens. Five of our gardens were farms, defined as communally-based gardens with large ($\sim$60–135 m^2) plots, monocultured rows, shared crop production, and selling of produce for profit.

The area of each whole garden was measured using Google Earth and the size of each individual plot was measured on site. Garden managers provided information about date of establishment and history of the garden. Garden age was adjusted for each sequential year (e.g. a 20 year old garden in 2010 was recorded as 21 in 2011) and plot size was re-measured each year. Median income was estimated for each garden neighborhood using the neighborhood census data from 2010 compiled by the LA Times (http://projects.latimes.com/mapping-la/neighborhoods). This data was based on 5-year household income estimates from the American Community Survey (http://www.census.gov/acs/www/). Median neighborhood income was the same across survey years as the reported income was a conglomerate estimate across 5 years. There are distinct limitations to using median household income to aggregate garden participant income. Aggregate income data may over or underestimate participant income, as low income gardeners may seek out community gardens at a higher rate than high income participants, due to limited home gardening space and greater need for low cost food accessibility (Jansson and Polasky 2010; Clarke et al. 2013). Despite these limitations, we determined that neighborhood median income was the most accurate way to address local ES demands, as low-income neighborhoods have reduced food access due to transportation limitation and few local grocery stores (Azuma et al. 2010; Shaffer et al. 2002) and have high immigrant populations (U.S. Census 2010).

Comprehensive species presence and abundance inventories were completed in each individually owned plot and for the whole garden (including common areas) during summers of 2010–2012. Each garden was visited and surveyed once each year between the months of June–August. All deliberately cultivated plants were identified and percent cover of each species estimated based on visual inspection. Covers were grouped into five area categories (0–5 %; 5–25 %; 25–50 %; 50–75 %; 75–95 %; 95–100 %). We then estimated m^2 of each species in a plot by taking the midpoint proportion of each category and multiplying that by plot size. As some plots had multiple layers of crops, this technique allowed the area of crops in a plot to be >100 %.

Species, not varieties, were recorded with a few exceptions. If different parts of the plant were used or one variety provided a separate use, they were recorded separately. For instance, *Brassica oleracea* encompasses a variety of distinct food products, such as broccoli, collards, and kohlrabi, each of which were recorded separately. In contrast, yellow crookneck squash and zucchini (both *Cucurbita pepo*) were only recorded as a single species as this difference did not result in variation of plant parts. Proper taxonomic identification for unusual species was assured through photos and collection of voucher specimens for expert identification and archiving at the UC Riverside herbarium. We divided species into broad use categories based on whether the species provided provisioning or aesthetic/cultural ES. These 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 and cultural services. Other included plants used for spiritual purposes (e.g. *Tagetes erecta* used in *Dia de los muertos*), fiber plants, shade trees, and pest deterrents. Many plants had multiple uses, so the sum of edible, medicinal, ornamental, and other species was greater than total richness. The most common species in each use are included in Table S1 as part of the online supplement.

Data analysis

As many gardens were similar in production between years, biodiversity and abundance variables were averaged across the three sampled years to identify how patterns of biodiversity and ES production varied within a garden (14 points per analysis for all gardens, 8 points for individually-based gardens). We used both one-way ANOVA, for comparison of abundance of different uses across management styles and immigrant status, and linear regressions to examine controlling factors on ecological variables (SPSS 11.3).

To account for potential co-linearities between our hypothesized mechanisms, we conducted correlations and multiple regressions to determine which combinations of factors were influencing each biodiversity or abundance measurement. To do this, we first conducted a Pearson's product moment correlation to compare garden age (years since establishment), plot size (m^2), and median neighborhood income for all gardens and separately for individually based and farm managements. We found that for individually-based gardens, plot size was positively correlated with both age of garden and neighborhood income (Table 2). The age–size correlation is unsurprising, as gardens built before the 1980s were established before a major housing boom in Los Angeles and more open space was available for garden plots (Gottlieb 2006). In addition, income and population density are

Table 2 Pearson's product moment correlation for hypothesized biodiversity mechanisms

	Income	Size	Age
Income (ALL)	–	0.04	0.0.091
Size (ALL)	0.04	–	0.482**
Age (ALL)	0.0.091	0.482**	–
Income (IND)	–	0.703*	0.322
Size (IND)	0.703*	–	0.794**
Age (IND)	0.322	0.794**	–

Comparisons labeled (ALL) are for all gardens, while comparisons labeled (IND) are only for individually-based gardens

* $p < 0.05$, ** $p < 0.01$

negatively related across Los Angeles (Clarke et al. 2013; U.S. Census 2010).

Stepwise multiple regressions including garden age, income, and plot size were conducted to individually determine predictors of total number of plot species, average number of species per plot, and species abundance. These were repeated for each different use, immigrant status, and management style (individually-based or farm). When stepwise multiple regression models included a combination of two or more variables to explain biodiversity or abundance, we used a partial regression to separate individual variable effects. This additional analysis accounted for the established co-linearities identified between our explanatory variables. For the partial regression, each significant variable identified in the stepwise regression was regressed against the residuals of a simple linear regression on the biodiversity or abundance measure and the other identified variables. If the partial regression was significant, this was reported as the individual effect of that variable. If not, then the observed significant effect of that variable was due to correlations with the other noted variable.

We used the Jaccard's index to determine β diversity or turnover between plots in a single garden in a single year (Anderson et al. 2011). Matrices of species presence-absence were used to compare biodiversity across all plots in the same garden (EstimateS 9.0). Resulting values were inverted to create an average Jaccard's dissimilarity index for each garden. This analysis was repeated for edibles and ornamentals in each garden and then the combination of 3 years was compared between uses with an ANOVA. Average Jaccard's dissimilarity between gardens was also used to directly compare turnover between years in a single garden and similarities between composition of gardens in a single ethnicity and between ethnicities.

Non-metric multidimensional scaling ordination (NMDS) of the Jaccard's dissimilarity metric was used to analyze community assemblage differences between garden sites (Anderson 1971; Cilliers et al. 2012). This ordination is nonlinear, and creates a physical representation maximizing distance based on rank-order agreement with their dissimilarities in species composition (Austin 2005). The closer two gardens are in the ordination space, the more similar they are in species composition. A Jaccard's dissimilarity matrix was created from a species presence–absence matrix (EstimateS 9.0). This matrix compared each garden in each year to all other gardens in all other years. The ordination was then projected in two dimensions (PROXSCAL on SPSS). This analysis was repeated using only edible or ornamental matrices. We then divided gardens into ethnic groups (as labeled in Table 1) in order to determine whether ethnic differences and immigrant status influenced species similarity. For statistical significance, resulting garden locations on each ordination axis was compared between ethnic groups using a one-way ANOVA.

Results

Biodiversity patterns

Across all garden plots, we found 707 species identified in garden plots across the 3 years of our study (Table 3). Over half the species were ornamental, with the four non-immigrant individual gardens containing the highest ornamental richness (185 species) and highest overall species richness (349 species) (Table 3). Though ornamentals had a higher

Table 3 Descriptive biodiversity across garden immigrant status and management styles

	Total	Immigrant garden	Immigrant farm	Non-immigrant garden	Non-immigrant farm
# of gardens	14	6	3	4	2
# of species	707	299	197	349	238
Edibles	229	160	135	152	105
Medicinals	44	26	19	27	16
Ornamentals	442	124	47	189	128

Garden indicates individually-based gardens and farm indicates communally-based. # of species is the number found in plots. Includes overall garden (n) and γ biodiversity for all species and each major species use biodiversity than edible species when combined across multiple gardens, a *t* test indicated that edibles outnumbered ornamentals in each garden (γ) by a factor of three (Fig. 2a; $p < 0.001$) and by a factor of four for plot (α) diversity (Fig. 2b; $p < 0.05$). The exception to the pattern was a single non-immigrant farm in the highest income neighborhood, which had more ornamentals than edibles at the α and γ scale (NIMM5). The number of species per plot in a specific year was correlated with the number of species in that garden for that year ($r^2 = 0.53$, $p < 0.001$; Fig. 3), a pattern repeated for edible and ornamental species. In addition, we found no consistent temporal pattern across sample years for abundance or species richness, with individual gardens increasing, decreasing, or having consistent biodiversity (Fig. 2).

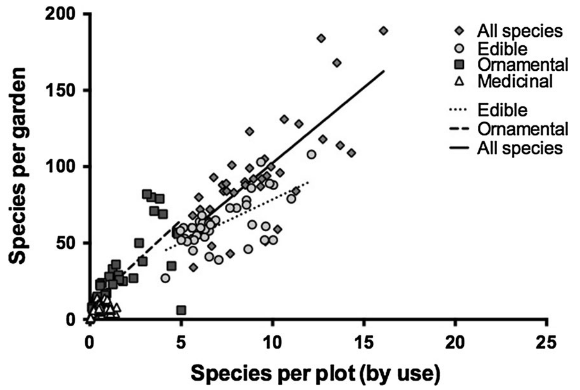

Fig. 3 Relationship of species per garden (γ) to species per plot (α), divided into all species (*diamonds*; $r^2 = 0.578$, $p < 0.001$), edible (*circles*; $r^2 = 0.339$, $p < 0.001$), medicinal (*triangles*; non-significant), and ornamental species (*squares*; $r^2 = 0.279$, $p < 0.001$). γ diversity of each use is compared to average α diversity of each use. *Each point* represents a single community garden in a single year (∼3 points per garden)

Socioeconomics and cultural background

Stepwise multiple regressions indicated that neighborhood income was variable the most related to overall species richness for all gardens, but plot size was the most related in individually-based gardens. Species biodiversity and cover were significantly related to neighborhood income in partial regressions controlling for the effect of plot size, though patterns

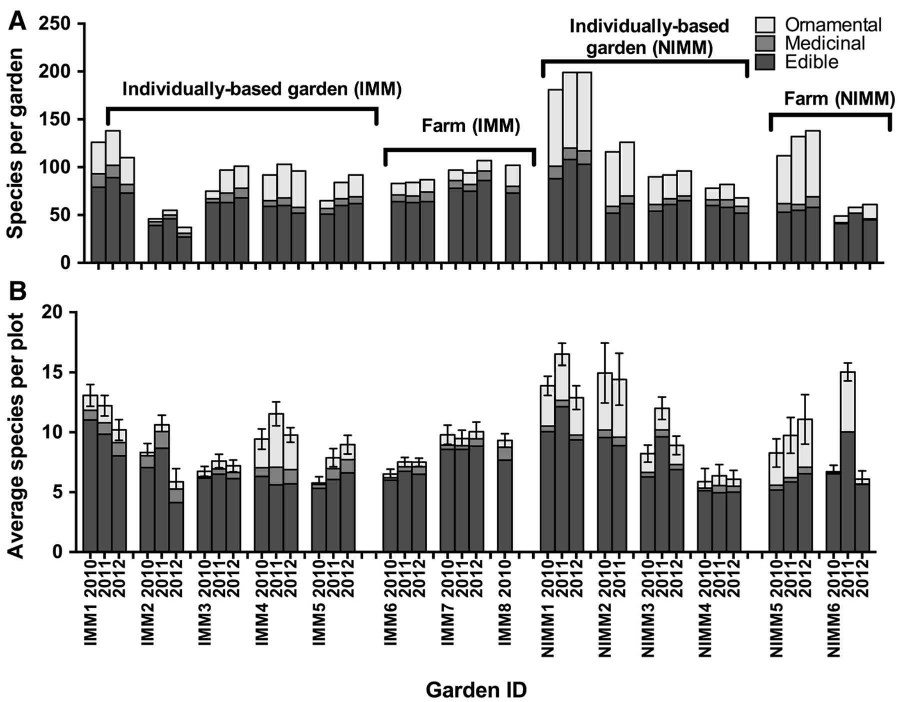

Fig. 2 Descriptive garden scale (**a**) and plot scale (**b**) plant biodiversity according to major use categories (ornamental, medicinal, edible). *Error bars* in **b** indicate standard error for overall biodiversity of plots within a single garden. For both garden and plot biodiversity, a t-test indicated edible species in each garden were more bio-diverse than ornamental or medicinal species (garden: $p < 0.001$; plot: $p < 0.001$)

differed between uses (Fig. 4a, b). Overall species richness was related to income ($r^2 = 0.553$, $p = 0.001$), but between different ES classes, only ornamentals increased with income ($r^2 = 0.719$, $p < 0.001$). Ornamental cover was also positively related to income ($r^2 = 0.530$, $p < 0.001$). Edible and medicinal species richness and cover showed no significant relationship with income ($p > 0.05$). When separated into ethnic groups, non-immigrant gardens were the only ones with a significant income-ornamental diversity relationship ($r^2 = 0.906$, $p < 0.01$). Immigrant gardens were located primarily in low-income neighborhoods, making it challenging to interpret whether immigrant status had a real influence over biodiversity.

Ornamental α and γ biodiversity were lower than edible species within and between gardens, but had a consistently higher turnover rate (β) than edibles (Fig. 5; Table 4). In each sample year, about 60 % of identified ornamentals were found in <1 % of garden plots, and no ornamental species were planted in more than 10 % of garden plots. In contrast, while 40 % of edibles found in each year were also found in <1 % of garden plots, they were more evenly distributed across plots. Between 10 and 15 edible species each year were found in 20–35 % of all plots (species identity of these common edibles shown in Table S1). β diversity varied greatly between uses (Table 4). While overall β was high between individual plots within a garden (Jaccard's dissimilarity >0.8), an ANOVA indicated ornamental β was the highest across all gardens ($p < 0.01$; Table 4).

Individual versus communal-based (farm) management style and immigrant status of community gardens affected the overall cover patterns (Fig. 6). While individual based garden plots had similar edible cover in both immigrant and non-immigrant locations, an ANOVA indicated that immigrant farms had the highest edible cover (Fig. 6; $p < 0.01$). Ornamental cover was highest in non-immigrant gardens and conversely lowest in immigrant farms ($p < 0.001$), while medicinal cover was the highest in immigrant gardens. In addition, edible cover was higher than ornamental across all gardens, ranging from 40 to

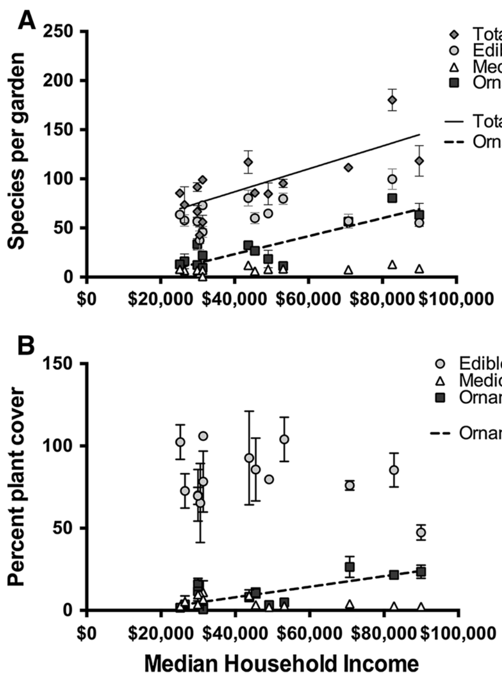

Fig. 4 Relationships between neighborhood median income and biodiversity (**a**) and vegetation cover (**b**) for each of the major species uses. All species *diamonds*, edible *circles*, medicinal *triangles*, and ornamental *squares*. *Error bars* represent standard deviation between 3 survey years. All regressions reported are based on stepwise regression models controlled for effect of plot size. Neighborhood income was related to total ($r^2 = 0.553$; $p = 0.001$) and ornamental biodiversity ($r^2 = 0.719$; $p < 0.001$) and to ornamental abundance ($r^2 = 0.530$; $p < 0.001$). Edible and medicinal richness and cover were not related to income

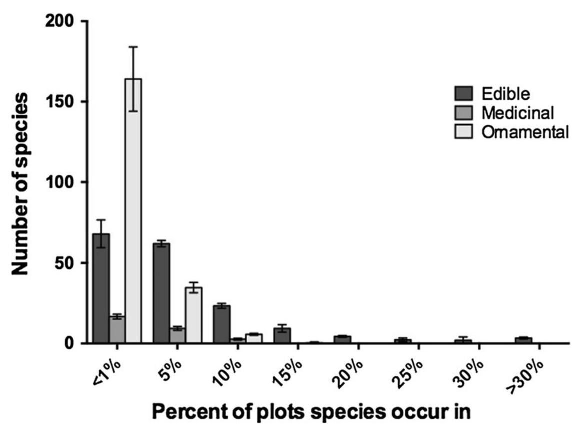

Fig. 5 Frequency distribution of edible, medicinal, and ornamental species. The *X-axis* represents the percentage of plots across all gardens that contain a specific species and the *Y-axis* indicates how many species are present at that frequency. *Error bars* represents standard deviation between the 3 study years. No ornamental species were found in more than 10 % of plots and the majority were found in <1 % of plots. In contrast, there are many edibles found in 10–30 % of all plots

Table 4 Average Jaccard's dissimilarity index, divided into use categories, between plots in each specific garden (representative of plot turnover and β diversity)

Garden ID		Species use		
		All	Edible	Ornamental
IMM1		0.862^{A}	0.873^{B}	0.985^{C}
	SE	0.002	0.002	0.002
IMM2		0.865^{A}	0.851^{B}	0.939^{C}
	SE	0.006	0.006	0.010
IMM3		0.874^{A}	0.863^{B}	0.994^{C}
	SE	0.002	0.002	0.001
IMM4		0.909^{A}	0.903^{A}	0.941^{B}
	SE	0.002	0.003	0.004
IMM5		0.916^{A}	0.908^{B}	0.977^{C}
	SE	0.002	0.002	0.003
IMM6		0.845^{A}	0.828^{B}	0.990^{C}
	SE	0.001	0.001	0.001
IMM7		0.867^{A}	0.858^{B}	0.940^{C}
	SE	0.002	0.002	0.004
IMM8		0.864^{A}	0.850^{B}	0.993^{C}
	SE	0.002	0.003	0.001
NIMM1		0.871^{A}	0.850^{B}	0.943^{C}
	SE	0.001	0.001	0.001
NIMM2		0.869^{A}	0.859^{B}	0.950^{C}
	SE	0.002	0.003	0.004
NIMM3		0.928^{A}	0.921^{A}	0.987^{B}
	SE	0.004	0.004	0.003
NIMM4		0.889^{A}	0.855^{B}	0.966^{C}
	SE	0.005	0.006	0.004
NIMM5		0.930^{A}	0.938^{A}	0.985^{B}
	SE	0.003	0.003	0.002
NIMM6		0.824^{A}	0.818^{A}	1.000^{B}
	SE	0.003	0.003	0.000

The higher the index, the more dissimilar garden plots are within that use. Different letters represent significant differences ($p < 0.01$) between use types in a single garden. For all gardens, ornamental species were the most dissimilar within each garden

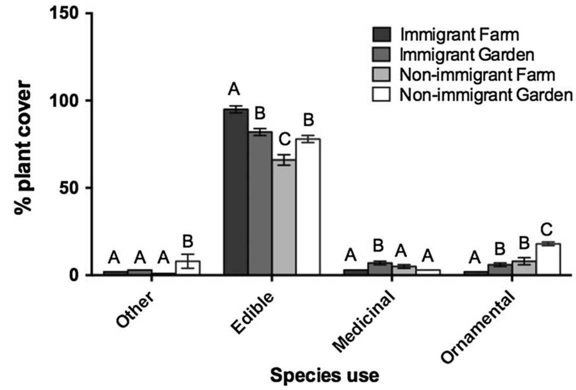

Fig. 6 Average vegetative cover of species across uses, immigrant status, and garden management style. *Error bars* represent standard error across plots in specific garden categories in all 3 years. *Different letters* represent significant differences between cover of a specific use between garden management categories

140 % in each plot, while ornamentals ranged from 1 to 30 % (Fig. 6). Ornamental and edible cover increased with their respective species richness, though explanatory value for edible species was low (Ornamental: $r^2 = 0.68$, $p < 0.01$; Edible: $r^2 = 0.14$, $p < 0.05$).

NMDS for all gardens indicated that dominant garden ethnicity influenced species composition within and across species uses. For all species (Fig. 7A.1), predominantly Hispanic/Asian gardens were grouped in ordination space and were located in a unique location in axis 1 (Fig. 7A.2). For edible species (Fig. 7B.1), Hispanic gardens were close to each other in ordination space and had a different set of species than all other gardens, as indicated by their unique location on axis 1 (Fig. 7B.2), and African-American food species were located in a unique area along axis 2. Finally, for ornamental species, plant distributions were more variable, though Hispanic gardens included significantly different species than non-immigrant gardens (7C.1, C.2). As edible species were the most grouped by ethnicity, we list the most commonly planted culturally specific food species in Table S2.

An ANOVA showed that gardens of a specific ethnicity were most similar in food species (according to Jaccard's dissimilarity) and most dissimilar in ornamental species (Fig. 8). Individually, gardens were self-similar across the 3 years of the study (average Jaccard's dissimilarity: 0.5, $p < 0.05$), indicating consistency of garden composition. The highest dissimilarity was observed between gardens of different cultural backgrounds in the same years (Fig. 8; average Jaccard's dissimilarity = 0.7; $p < 0.05$).

Species–area relationships and legacies

Garden scale species richness was positively related to size of individual plots ($r^2 = 0.785$; $p < 0.01$;

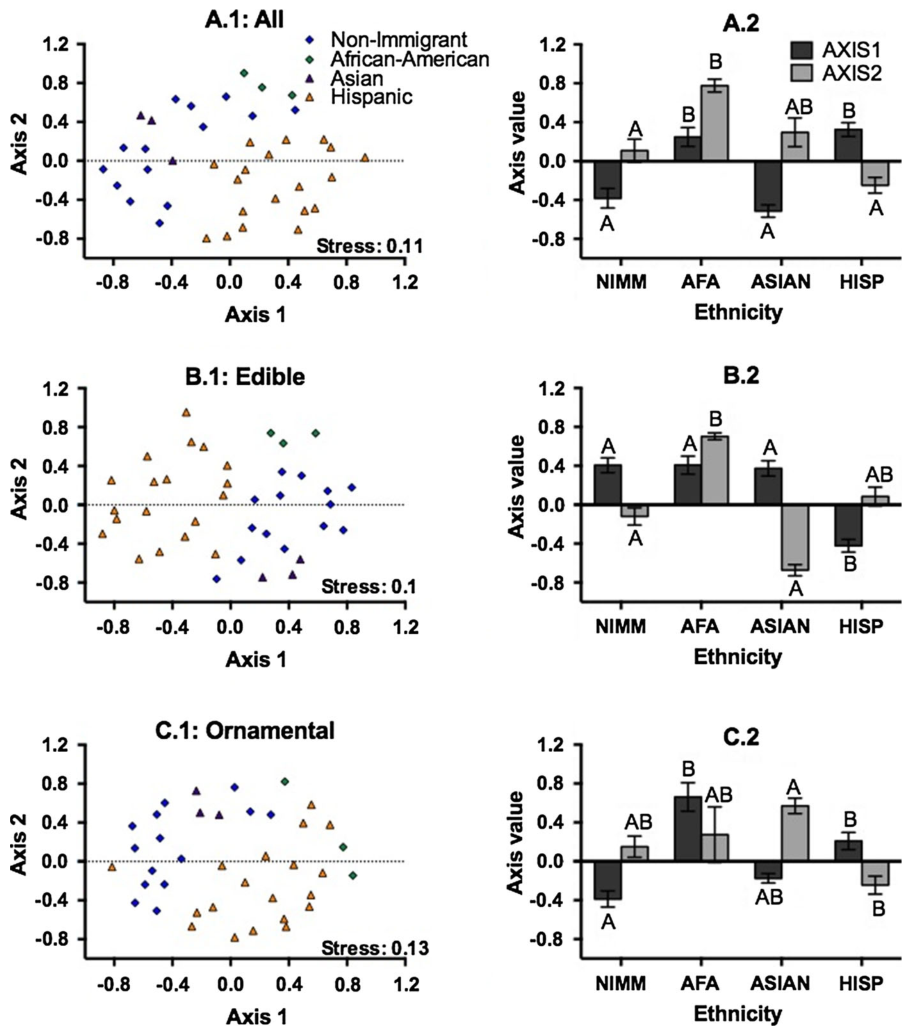

Fig. 7 *Column 1* non-metric multidimensional scaling (NMDS) ordination based on Jaccard's dissimilarity matrices for all species (**A.1**), edible species (**B.1**), and ornamental species (**C.1**). *Each point* represents a single garden in a single year. Gardens closer to each other are more similar in species composition. Stress levels in each plot indicate proportion of variance unaccounted for. *Column 2* ANOVA comparing location of culturally distinct gardens on each ordination axis. *Different letters* indicate significant differences ($p < 0.05$) between gardens of different ethnicities (*AFA* African-American, *ASIAN* Asian, *HISP* Hispanic, *NIMM* non-immigrant) on that axis and indicate unique groupings. *Error bars* represent standard error

Fig. 9a), but only in individually-based gardens, not farms (Fig. 9b). The species–area relationship was the most evident for both edible ($r^2 = 0.810$; $p = 0.001$) and medicinal species ($r^2 = 0.882$; $p < 0.001$) in individually-based gardens. As income and garden establishment dates were not identified as significant factors in total, edible, and medicinal stepwise models, we did not complete partial regressions for this analysis. Ornamental species richness was unrelated to size in stepwise models. Farm-style gardens had low variation in the number of species found within gardens, regardless of plot size, a pattern that remained the same across all species uses. For individually-based gardens, garden establishment date was not identified as a significant factor in any stepwise models for abundance or biodiversity, even in individual comparisons of species uses, immigrant status, and garden management.

Discussion

Los Angeles community gardens contain extensive plant biodiversity, with over 700 managed species in a total area of only 6.5 ha, or nearly 100 species per hectare across 3 years. Though 95 % of the species found in gardens are non-native exotics, managed species have been shown to contribute functional traits that are beneficial to humans and the environment (Hooper et al. 2005; Matteson et al. 2008; Pataki et al. 2013).

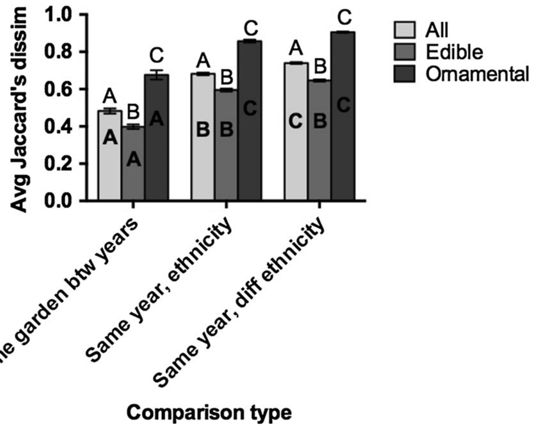

Fig. 8 Average Jaccard's dissimilarity between gardens for major species uses (all, edible, ornamental). Comparisons include a single garden across each of 3 years, gardens in the same year and ethnicity, and gardens in the same year with different ethnicities. *Different bold letters* within columns represent significant differences between Jaccard's dissimilarity in a single use across comparison types. *Different letters* above columns represent significant differences between uses in a single comparison type. *Error bars* indicate standard error

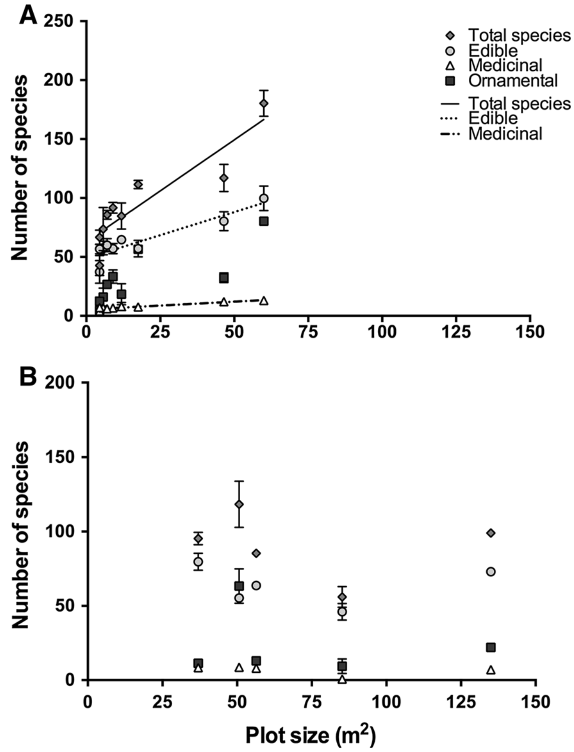

Fig. 9 Relationship between plot size and species richness in individually based gardens (**a**) and farms (**b**). Total number of species (*diamonds*) is then divided into edible (*circles*), medicinal (*triangles*), and ornamental (*square*) species. *Error bars* represent standard deviation between years. *Regression lines* are based on stepwise regression models. Plot size in individually based gardens (**a**) is positively related to all species ($r^2 = 0.785$; $p < 0.01$), edibles ($r^2 = 0.810$; $p = 0.001$), and medicinals ($r^2 = 0.882$; $p < 0.001$), but not ornamentals. Plot size and biodiversity were not related in farms (**b**)

Since this subsample of community gardens represents <20 % of the 100 gardens in Los Angeles County, the number of managed species in LA gardens may be higher than previous studies of entire metropolises (Walker et al. 2009; Wang et al. 2012). This high biodiversity and the ES provided in LA community gardens are driven by a combination of garden management, income, cultural identity, and area. Scale-specific variation of α and β diversity are linked to ES provided and garden management style (Figs. 5, 6), and our results indicate high plot (α) biodiversity influenced larger scale garden (γ) biodiversity (Fig. 3). Older gardens showed no legacy effect on biodiversity, and gardens remained relatively similar in species composition over multiple sampling years (Figs. 2, 8). Our findings support our hypothesis of a hierarchy of need coupled with cultural preferences, indicating that gardens in impoverished regions produce culturally important food species (Table S2, online supplement; Figs. 7, 9), while high-income gardens invest more heavily in ornamental diversity (Fig. 4), possibly due to increased financial resources. We also found that species–area relationships exist only at the plot scale in individually-based gardens, primarily influencing edible species (Fig. 9a, b), thus indicating management style and ES influence space demands.

Socioeconomics and the hierarchy of need

Species uses and ES production in community gardens are related to median family income (Fig. 4), supporting a hierarchy of need hypothesis (Wu 2013). There is a lack of resident access to culturally appropriate and healthy food in Los Angeles (Shaffer et al. 2002; Azuma et al. 2010; Jackson et al. 2013). Our results are consistent with low-income garden participants responding to reduced access to resources by selecting crops that provide edible ES, and not investing in ornamentals (Figs. 4, 7), though individual participant motivations were not quantified. Food crops may improve gardener livelihoods through providing basic food needs and promoting cultural expression (Alaimo et al. 2008; Davis et al. 2011; Clarke et al. 2014b).

High ornamental richness in affluent neighborhoods may be due to luxury investments in aesthetic and cultural ES. Heterogeneity of ornamentals (Table 4) was high, and may result from affluent gardeners expressing preferences through unique ornamentals (Marco et al. 2010). This shift from provisioning to cultural and aesthetic ES with increasing socioeconomic status has been observed in cities across the world (Hanna and Oh 2000; Kinzig et al. 2005; Loram et al. 2008). While edible species richness does not decrease with increasing income, higher income may give gardeners resources to invest in flowering species (Cilliers et al. 2012) and intensively manage more extensive plant assemblages (Walker et al. 2009; Lowry et al. 2012).

Patterns of scale-specific landscape variation may also be interpreted using a hierarchy of need. Regional and garden scale richness display different patterns in allocation of species providing ES. Though ornamentals outnumber edibles regionally, each garden has proportionally higher edible richness (Table 1) associated with differences in β diversity. Gardener valuation of provisioning and aesthetic ES may explain the proportional difference. Specific food needs may be fulfilled by each edible species, not by overall diversity, and gardeners may value a few food species to sustain their family (Galluzzi et al. 2010; Hale et al. 2011).

Though our finding of higher edible abundance in low-income neighborhoods supports the hypothesis of a hierarchy of need, little is known about individual motivations and garden scale contributions to food security. Follow-up qualitative surveys of urban gardeners will better identify individual desires and the role of gardens in alleviating food security.

Ethnic gardener preferences

Ethnically distinct groups of gardeners grow distinctly different sets of garden species (Fig. 7). In particular, edible species were more similar within specific ethnicities than other uses (Fig. 8), and contained unique culturally relevant species (Table S2). Consistent with these landscape patterns, individual gardens were also similar in species biodiversity, especially edibles, across multiple years (Fig. 8). Both spatial and temporal patterns are consistent with valuation of increased food sovereignty.

Immigrant gardeners may express social heritage and history through culturally important food sources (Fu et al. 2006; Hale et al. 2011). Cultivating culturally relevant crops helps immigrants and ethnic groups maintain cultural identity and agrarian traditions in an unfamiliar environment (Corlett et al. 2003; Peña 2006). Each identified ethnic group had a suite of edible species distinct to their cultural background (Table S2). Many immigrant participants in community gardens express desire for fresh, familiar produce in their gardens (Corlett et al. 2003; Taylor and Lovell 2014). Though ornamental composition is less segregated by ethnicity than edibles (Figs. 7C, 8), ornamental may also hold cultural value. For instance, *Tithonia rotundifolia* and *Tagetes erecta* are both used as ornamental species in Hispanic gardens (Table S1), but they also provide important cultural services, as they are used extensively in the *Dios de los Muertos* celebration throughout Central America. Americans, Europeans, Hispanics, and Asians can have very different preferences for decorative landscapes (Kaplan and Herbert 1987; Fraser and Kenney 2000; Kinzig et al. 2005), which may explain some of the ethnic preferences in ornamental choice.

Garden area and age

Garden management style affected species–area relationships across community gardens for plot size, not garden size, affecting edible and medicinal biodiversity only in individually-based gardens (Fig. 9a). Farms often share food communally, so there is less pressure for a single plot manager to grow all edibles necessary for sustenance (Pedro Barrera, farm manager, pers. comm). In individually-based gardens, participants who desire a certain suite of species must grow them all in a single plot. In contrast, ornamentals take up a much smaller area of the garden (Fig. 6) and our other results indicate they are valued for diversity, not cover (Fig. 5; Table 4). Species abundance patterns are also affected by both management and immigrant status (Fig. 6). Gardeners, who rely monetarily on garden success, such as farm participants, may be more likely to plant edible species because of their commercial value (Fu et al. 2006; Lubbe et al. 2011; Galluzzi 2012). This pattern is evident in immigrant farms, which have the highest abundance of edibles and conversely lowest ornamentals.

We did not observe a legacy effect of garden age on species biodiversity patterns. Previous studies showing a clear effect of development age on biodiversity were from surveys of trees or perennials, which are uncommon in community gardens (Boone et al. 2010; Clarke et al. 2013). We had initially posited that older gardens could indicate high land tenure and security for gardeners, encouraging crop legacies. While our analyses show no effect of garden age on species biodiversity or abundance, the age of gardens may be a poor proxy for gardener tenure and security. Qualitative surveys incorporating individual gardener decisions based on plot scale tenure or garden stability may better evaluate legacy effects.

Synthesis

The results of our intensive study provide comprehensive information for urban planners on the extent of community garden biodiversity, abundance, and the drivers of biodiversity and ES production in a large and diverse U.S. metropolis. Community garden biodiversity is influenced interactively by income, culture, management, and area. These highly diverse and dynamic crop repositories may be considered a secondary Vavilov center of global biodiversity (Vavilov 1949), where high genetic biodiversity in LA is being created and maintained by gardeners imposing selection pressure on crop species over multiple years (Soleri and Cleveland 2004; Heraty 2010). Our results also indicate that garden placement and planning by local government bodies should favor ethnic food production for impoverished minority communities (Lovell and Taylor 2013; Smith et al. 2013). In addition to the direct services, we expect that high biodiversity can also support indirect ES, such as pollination and pollution reduction. Further, potential disservices of urban agriculture, such as weed and pest proliferation (Mack and Erneberg 2002) should be evaluated to better understand and minimize ES tradeoffs associated with urban agriculture. A health tradeoff often observed in urban gardens is heavy metal contamination of urban soils (Schwarz et al. 2012; Clarke et al. 2014a) and reconciling food production with potential contamination is an important concern.

Our quantitative data helps "close the loop" in linking gardener and societal desires to ES production across complex urban landscapes (Lawson 2007; Chappell and LaValle 2011). As community gardens are proliferating across the country (Corrigan 2011; Lawson and Drake 2013), these results indicate demand for policy makers to create more secure, accessible gardens for minority participants in lower income neighborhoods. Community gardens in Los Angeles are a model for understanding of human–ecosystem functioning related to biodiversity and the production of ES and show how diverse drivers, including a hierarchy of need, cultural preferences, and size of plots, influence patterns of diversity and ES production. These causes of variation and their interaction may be broadly applicable in CHaNS where ecosystem services are regulated by both social and environmental heterogeneity.

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