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Why garden for wildlife? Social and ecological drivers, motivations and barriers for biodiversity management in residential landscapes

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ABSTRACT

Residential landscapes with private gardens are major land covers in cities and their sustainable management is paramount for achieving a resilient urban future. Here we focus on the value of residential ecosystems for biodiversity conservation and explore the social and ecological factors that influence wildlife-friendly garden management. Using a stratified sampling design across the UK city of Leeds, this interdisciplinary study develops and applies a mixed method approach, including questionnaires, interviews and ecological surveys across multiple spatial scales. We quantify wildlife-friendly gardening using two measures: (i) the number of wildlife-friendly features within gardens (the wildlife resources index, WRI); and (ii) the frequency of winter bird feeding. Wildlife-friendly gardening is influenced by a combination of garden characteristics and management intensity, householder demographics, wider environmental activity and landscape context. Residents reveal a range of motivations for wildlife-friendly gardening, notably personal well-being and a moral responsibility to nature. Respondents expressed a duty to maintain neighbourhood standards, revealing that social norms are a considerable barrier to uptake of wildlife-friendly activities, but also provide an opportunity where neighbour mimicry results in diffusion of wildlife-friendly practices. Community-driven initiatives that engage, educate and empower residents are better placed to encourage wildlife-friendly gardening than top-down financial incentives.

1. Introduction

In a context of increasing urbanisation (United Nations, 2010) and declining biodiversity, there is concern that people living in cities are becoming disconnected from the natural world (Miller, 2005; Turner et al., 2004), resulting in apathy towards wider conservation objectives (Dunn et al., 2006). This disconnect from nature is particularly worrying in light of evidence that interactions with urban wildlife are important for human health and well-being (Fuller et al., 2007; Luck et al., 2011). Since private gardens are one of the primary settings for interactions with wildlife in cities, they offer great opportunity for personal engagement with the natural world (Dunnett and Qasim, 2000; Freeman et al., 2012; Power, 2005).

Private gardens are a major component of cities in both developed and developing world countries (e.g. Gonzalez-Garcia and Sal, 2008; Loram et al., 2007) and the manner in which householders manage these spaces has a substantial impact on the provision of urban biodiversity. The benefits of activities by householders to encourage biodiversity through wildlife-friendly gardening have been recognised by policymakers and conservation NGOs alike (Goddard et al., 2010b). Ecologists have recently attempted to quantify the extent of wildlife-friendly gardening across UK cities (e.g. Davies et al., 2009; Gaston et al., 2007) and found that feeding birds is the most popular activity carried out by an estimated 12.6 million (48%) households. Similar levels of bird feeding occur in both the United States and Australia (Jones and Reynolds, 2008; Lepczyk et al., 2012). Research suggests that supplementary feeding can benefit bird populations at multiple scales (Daniels and Kirkpatrick, 2006; Fuller et al., 2008), although others have highlighted the adverse impacts of bird feeding, such as disease transmission and increased predation pressure (Robb et al., 2008). In general, the cumulative actions of many householder activities can combine to benefit biodiversity (Cooper et al., 2007). Equally, these impacts can be negative, such as from the application of lawn chemicals (Robbins et al., 2001), predation by domestic cats (Sims et al., 2008), or the enhancement of biological invasions (Niinemets and Penuelas, 2008).

Residential landscapes are complex socio-ecological systems that are best understood within an interdisciplinary framework (Cook et al., 2011; Grove et al., 2006). Initial interdisciplinary studies have shown that patterns of urban biodiversity are inherently linked with social stratification (Warren et al., 2010). For example, there is evidence of a ‘luxury effect’, whereby wealthier neighbourhoods support greater levels of vegetation cover or higher plant diversity (e.g. Hope

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et al., 2003; Lubbe et al., 2010; Martin et al., 2004). Socio-economic status also correlates with the richness of various vertebrate taxa (Kinzig et al., 2005; Melles, 2005; Smallbone et al., 2011; Strohbach et al., 2009). At finer scales within neighbourhoods, householder landscaping decisions are influenced by the desire to conform to prevailing social or cultural norms (Kurz and Baudains, 2010; Marco et al., 2010; Nassauer et al., 2009). Research in Baltimore, US, has shown an 'ecology of prestige' whereby vegetation cover in private gardens is predicted by lifestyle behaviour and a need to show membership of a given lifestyle group (Grove et al., 2006; Troy et al., 2007). The presence of a shared social ideal often results in spatial autocorrelation of gardening practices in suburbia (Hunter and Brown, 2012; Warren et al., 2008; Zmyslony and Gagnon, 1998), although these findings are not universal (Kirkpatrick et al., 2009).

To maximise the contribution that householders make to the biodiversity of residential ecosystems, a greater understanding of the myriad ecological and social factors that underlie wildlife gardening practices is required (Goddard et al., 2010a; Kendal et al., 2010). There have been very few investigations into patterns of wildlife-friendly gardening, but preliminary research in US cities has explored some of the socioeconomic and demographic correlates of householder activities that influence birds (Lepczyk et al., 2004, 2012). UK urban ecology studies have examined the spatial variation in wildlife gardening and bird feeding and related this to neighbourhood-scale socio-economic status, population density and landscape context (Fuller et al., 2008; Fuller et al., 2012; Gaston et al., 2007). As yet, we know little about what drives people to engage in wildlife-friendly gardening. Studies of motivations for gardening in general have found that observing nature is highly valued by gardeners (Clayton, 2007; Fuller and Irvine, 2010). A body of social research from Australia and New Zealand has illustrated gardeners' attitudes and practice regarding native plants (e.g. Doody et al., 2010; Head and Muir, 2006; Zagorski et al., 2004), whilst researchers in environmental psychology have investigated the association between wider environmental values and ecological gardening practices and found contrasting results (e.g. Kiesling and Manning, 2010; Larson et al., 2010). Here, we use an integrated, interdisciplinary research design to simultaneously explore the social and ecological drivers, motivations and barriers for biodiversity management in residential landscapes at multiple scales. In particular, the study objectives are to: (1) examine the spatial variation in activities to encourage wildlife in gardens and relate this to landscape context, socio-economic status, householder demographics, environmental values and garden characteristics and management; (2) assess whether wildlife gardening activities are correlated with bird richness, diversity or abundance; (3) determine the range of influences on, and underlying motivations behind, wildlife gardening; and (4) explore the potential of various mechanisms for incentivising greater participation in wildlife-friendly gardening.

2. Methods

2.1. Study Area and Sampling Design

We develop and apply a mixed-methods research design that incorporates householder questionnaires and interviews with ecological surveys across a stratified sample of urban neighbourhoods in the UK city of Leeds, West Yorkshire (53° 47′ 59″ N, 1° 32′ 57″ W). With a human population approaching 790,000, Leeds is the third largest municipality in the UK. The Leeds metropolitan district covers an area of c. 550 km^2, of which around two-thirds is farmland. Here we define the Leeds study area as the extent of the contiguous Leeds and Bradford urban area that falls within the Leeds District (Fig. 1). This urban area covers 133 km^2 and is typical of cities in developed, temperate countries in containing a wide range of residential areas.

We used a hierarchical sampling design whereby study households were located within neighbourhoods that were in turn nested within wards. Wards are UK administrative areas and 27 fall within the Leeds urban boundary. Wards are further divided into Output Areas, OAs (hereafter termed neighbourhoods), that are the finest scale for which census data are available, typically classified based on tenure and dwelling type with a target size of 125 households (Office for National Statistics, 2011). We selected wards and neighbourhoods using stratified random sampling to capture the range of variation in landscape and socio-demographic characteristics. Six wards were selected: Roundhay, Morley South, Pudsey South, Whinmoor, Armley and Hunslet (Fig. 1). Three neighbourhoods were selected within each ward, giving a total of 18 study neighbourhoods (Fig. 1; Table 1). The hierarchical sampling design allows us to ascertain the relative contribution of household-scale factors compared to neighbourhood- and landscape-scale drivers affecting the biodiversity of private gardens.

2.2. Household Questionnaire

A questionnaire was delivered by hand to all households in the 18 neighbourhoods. To maximise response rate we implemented several of the methods recommended by the Tailored Design Method (Dillman et al., 2009), such as the inclusion of a stamped return envelope and personalising correspondence by using a hand-addressed envelope along with a personally signed covering letter explaining the purpose of the survey. 2198 questionnaires were delivered and 533 were completed (24% response rate). There was a response bias across neighbourhoods, with the most affluent neighbourhood (R1) having a 49% response rate, compared to 14% in the least affluent neighbourhood (H3). An exploratory analysis that controlled for the effect of response rate using linear models showed that it had no significant effect on model fit, so was subsequently dropped from analyses as it indicated response rate did not affect the results.

The questionnaire was the most comprehensive survey of garden management and wildlife gardening practices to have been undertaken in a UK city. It contained 30 questions that covered main themes of: (i) garden use and management; (ii) current wildlife gardening practices and wildlife observations; and (iii) house and garden characteristics (e.g. house type, garden size) (Appendix A). In addition, respondents were asked socio-demographic questions relating to age, presence of young children, housing tenure, length of residency, occupation and education. Respondents' level of wider environmental commitment was assessed by asking about participation in other environmental activities and membership of garden or wildlife organisations/charities. Finally, respondents were asked how important they deemed six global environmental issues by scoring them on a Likert scale from 1 (not important) to 5 (very important). Four indices were calculated for data analysis based on questionnaire responses (Table 2). The wildlife resources index (WRI) was used as one of two response variables, and the management intensity index, environmental activity index and environmental concern index were used as explanatory variables. Ground-truthing was used to verify the accuracy of the WRI during garden ecological surveys (Goddard, 2012) and correlation between respondent-assessed WRI and that recorded by MG in a subset of 90 gardens was moderately high ($r_s = 0.72$).

Questionnaire responses were excluded from data analysis if the householder failed to complete the appropriate questionnaire section, or if ≤3 questions were answered in the management intensity index or environmental concern index. Where respondents failed to answer ≤2 questions in the above indices, missing values were imputed based on responses to other questions in the same index (after Luck et al., 2011). For the WRI and environmental activity index, where respondents were asked to tick boxes to indicate the presence of features or participation in activities respectively, blanks were interpreted as negative because it was deemed relatively easy