Land Use Policy 64 (2017) 114–123

Contents lists available at ScienceDirect

Land Use Policy

journal homepage: www.elsevier.com/locate/landusepol

The value of green walls to urban biodiversity

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ARTICLE INFO

Article history:
Received 6 October 2016
Received in revised form 9 February 2017
Accepted 19 February 2017
Available online 6 March 2017

Keywords:
Green infrastructure
Green wall
Living wall
Green façade
Choice experiment

ABSTRACT

Despite increased interest in the implementation of green walls in urban areas and the recognised benefits of monetary valuation of ecosystem services, no studies have been undertaken to estimate the economic value of biodiversity they provide. The valuation of natural resources allows policy makers to justify resource allocation. Using the Southampton, UK, as a case study, this paper estimates the public's perceived value of green walls to urban biodiversity, in the form of their willingness to pay (WTP). Estimates were derived using a random parameter model that accounted for socio-economic and attitudinal determinants of choice, using choice experiment data. Three green infrastructure policies were tested; two green wall designs ('living wall' and 'green façade') and an 'alternative green policy'; and compared against 'no green policy'. Results indicated a WTP associated with green infrastructure that increases biodiversity. Attitudinal characteristics such as knowledge of biodiversity and aesthetic opinion were significant, providing an indication of identifiable preferences between green policies and green wall designs. A higher level of utility was associated with the living wall, followed by the green façade. In both cases, the value of the green wall policies exceeds the estimated investment cost; so our results suggest that implementation would provide net economic benefits.

1. Introduction

Biodiversity has multiple roles in the delivery of ecosystem services: as a supporting service of other ecosystem services, as a final regulating ecosystem service in itself, and as a good such as the existence of an iconic species (Atkinson et al., 2012; Mace et al., 2012). Conserving biodiversity ensures the provision of ecosystem services and the multi-layered benefits that underpin human health and wellbeing (Bolund and Hunhammer, 1999; MEA, 2005; NEA, 2011). With increasing urbanisation, the role of urban biodiversity in delivering ecosystem services has been studied widely (Botzat et al., 2016). Ecosystem services are recognised to add value to urban environments in economic, social and environmental terms (Bolund and Hunhammer, 1999; Gómez-Baggethun and Barton, 2013; Natural England, 2013).

Biodiversity in cities is concentrated mostly within a limited network of green infrastructure (Finlay, 2010; Tzoulas et al., 2007). Conventionally, green infrastructure includes a combination of parks, gardens, green corridors and rivers, strategically planned and linked to protect biodiversity (Tzoulas et al., 2007). With current rates of urban development unlikely to decrease, existing green infrastructure is insufficient to prevent predicted declines in biodiversity in urban areas (Mcdonald et al., 2008; Rosenzweig, 2003). The adoption of additional green infrastructure is needed to ensure the continued provision of ecosystem services and safeguard the health and wellbeing of city dwellers (Francis and Lorimer, 2011; Tzoulas et al., 2007).

In cities and urban environments, where space is costly, an increasingly common approach to enhance green infrastructure is to integrate vegetation into vertical structures as 'green walls' (Chiquet et al., 2013; Francis and Lorimer, 2011; Manso and Castro-Gomes, 2015). The term green wall refers to all forms of vegetated vertical surfaces (Manso and Castro-Gomes, 2015; Weinmaster, 2009). Traditional green wall methods are historically known, dating back to the Hanging Gardens of Babylon, and the Roman and Greek Empires (Köhler, 2008; Weinmaster, 2009). New engineering and technological advances have resulted in a variety of designs that can be incorporated into new or existing infrastructure (Manso and Castro-Gomes, 2015; Weinmaster, 2009). At a local scale, green walls have proven benefits for biodiversity, with even simplistic flora ensemblages providing a habitat for invetebrates (e.g. Francis and Lorimer, 2011) and nesting, food and shelter resources for

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http://dx.doi.org/10.1016/j.landusepol.2017.02.025
0264-8377/

urban ornithology (e.g. Chiquet et al., 2013). Theoretically, advances in technology mean that living walls can be engineered to replicate natural habitats and create wider possibilities for biodiversity enhancement (Francis and Lorimer, 2011). Green walls can support biodiversity in cities at a landscape scale by acting as a “corridor” or “stepping stone” to facilitate movement and despersal (Angold et al., 2006). A well connected network, managed at a landscape scale, will increase the stability of urban biodiversity in the face of increased disturbances and stochastic changes (Goddard et al., 2010). The EU Green Infrastructure Policy, which is linked to the EU 2020 Biodiversity Strategy, recognises that connectivity is key for biodiversity resilience against change and further highlights green walls as an important, and cost effective, element of green infrastructure in the urban environment (EEA, 2011; European Commission, 2013).

The United Kingdom (UK) Government have formally recognised the importance of green infrastructure in the provision of biodiversity through the publication of the Natural Environment White Paper; The Natural Choice: Securing the Value of Nature (DEFRA, 2011). Informed by the findings of the UK National Ecosystem Assessment (NEA, 2011), the White Paper aims to halt biodiversity loss by 2020, support ‘healthy and functioning ecosystems’, and establish ‘coherent ecological networks’ (DEFRA, 2011). In the UK, decisions regarding the implementation of green walls, and other elements of green infrastructure, are made at a local and neighbourhood level,[2] typically from an economic perspective (Vandermeulen et al., 2011). The monetary valuation of ecosystem services enables local authorities to quantify and recognise the benefits of ecosystem services, and justify the allocation of limited public resources (Natural England, 2013). The use of monetary indicators enables the direct comparison of alternative green policies as well as the costs based on a common unit of comparison, which is not always possible when using biological or descriptive indices (Natural England, 2013; Nijkamp et al., 2008; Nunes and Van den Bergh, 2001). Consequently, local authorities and decision makers are calling for the assessment of green policies and infrastructure in economic terms (Natural England, 2013; PUSH, 2010).

To date, there have been two studies that quantify benefits of green walls. The first, a study by Veisten et al. (2012), successfully provided an economic unit of acoustic and aesthetic benefits. The second, a study by Perini and Rosasco (2013), presents a cost-benefit anaylsis to determine the economic sustainability of green walls; the benefits of biodiversity where included within the scope of their analysis but were only considered at a qualitative level. Neither study specifically quantified the benefits of biodiversity provided by green walls and cannot reliably be used to justify the implementation of green walls as a means to enhance biodiversity. Therefore, the aim of this paper is to present a monetary valuation study of green infrastructure; in which we set out to economically quantify the value of biodiversity provided by green walls, and determine public preferences towards green wall design.

2. Environmental valuation methodology

2.1. Application of choice experiments to value biodiversity

For many benefits generated by biodiversity there is no formal market, i.e. the value is non-marketed (Jones-Walters and Mulder, 2009), and analysts wishing to value such benefits have to rely upon non-marketed valuation techniques (Bartkowski et al., 2015). Among the array of tools and methods to monetise non-market values, the recently more commonly adopted technique is choice experiments (CEs) (Bartkowski et al., 2015). Developed by Louviere and Hensher (1982) and Louviere and Woodworth (1983), CEs involve the application of characteristics theory of value (Lancaster, 1966), combined with random utility theory (Manski, 1977; Thurstone, 1994), where utility refers to the total amount of satisfaction received from consuming a good or service (Louviere et al., 2000). CEs rely on the generation and analysis of stated preference data; data are acquired through questionnaires (Hoyos, 2010). Respondents, usually the general public, are presented with choice sets containing mutually exclusive hypothetical alternatives and asked to choose their preferred option (ibid.). Alternative choices are defined and differentiated by a set of attributes, each attribute taking more than one level. The individual’s choice implies a trade-off between alternatives (Hanley et al., 2002; Hoyos, 2010). When cost or price is included as an attribute, marginal utility estimates can be obtained and converted into willingness to pay (WTP), thus providing a monetary value (Bartkowski et al., 2015; Jones-Walters and Mulder, 2009). The application of a CE also presents the opportunity to gauge public preferences for different policy designs, and assess whether these preferences vary with individual characteristics (Nijkamp et al., 2008; Vandermeulen et al., 2011).

Due to the non-market value of urban biodiversity, the use of hypothetical markets in CEs justifies the use of stated preference method in this study (Gómez-Baggethun and Barton, 2013). Other methods do not have the potential to capture non-use (Pascual et al., 2010) and indirect values, which are crucial value components of biodiversity (Bartkowski et al., 2015). Existing studies valuing the benefits of biodiversity include Christie et al. (2004, 2006), Morse-Jones et al. (2012) and Garrod and Willis (1997). There are also a number of studies utilising CEs to value the benefits of other elements of green infrastructure in urban areas including; urban forests (Bernath and Roschewitz, 2008; Kwak et al., 2003), wetlands (Boyer and Polasky, 2004), open spaces (Brander and Koetse, 2011) and urban greenways (Lindsey and Knaap, 1999).

Biodiversity is a complex and multi-level concept that can be broken down into many additional attributes (Bartkowski et al., 2015). One of the critiques of monetary valuation of biodiversity is that respondents may interpret this concept (and associated benefits) differently. Biodiversity can be defined at the level of genes, species, ecosystems or functions (Nunes and Van den Bergh, 2001), but the term is also used more broadly to refer to biological variety in the environment at all levels, indicated by the number of different species of plants and animals and habitats present. These multiple roles are associated with multiple benefits (and multiple beneficiaries), which makes monetary valuation of biodiversity challenging (Atkinson et al., 2012), especially when the processes and functions leading to benefits are interdependent and non-linear. One option would be to list all such benefits in non-monetary terms, but knowledge about the range and amount of such benefits may not exist and it would greatly complicate the decisions faced by respondents in the CE. In general, high complexity can negatively influence the validity and reliability of estimates (Hanley et al., 2002) and has been attributed to the current limited use of CEs in day-to-day decision making, particularly at a local planning level (Broekx et al., 2013).

The debate on the appropriateness and reliability of monetary valuation is on-going. Arguments in favour include practical requests for value estimates (e.g. Rudd et al., 2016), the need to demonstrate the importance of biodiversity for green economic development (e.g. Potschin et al., 2016), and the possibility of approximating of public support expressed in monetary terms and comparison with costs that can be valid in particular contexts (e.g. Lienhoop et al., 2015). However, others argue that stated preference techniques are not sufficiently reliable to be used for

[2] The Localism Act 2011, and the National Planning Policy Framework (DCLG, 2012), led to fundamental changes in the planning system; power has shifted to local and neighbourhood levels. Subsequently, it is now the decision of local authorities to implement a green infrastructure policy.