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The value of green walls to urban biodiversity

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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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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.

monetary valuation when respondents have no clear conceptual understanding of the complexity of the good they are asked to value (e.g. Farnsworth et al., 2015), or are uncertain of their response (e.g. Ready et al., 2001). Likewise, when respondents do not consider the scenario to be consequential, stated WTP results may be biased (Carson and Groves, 2007; Jones-Walters and Mulder, 2009). Respondents who concentrate on the payment vehicle rather than on the desirable quality of the service provided may provide conservative estimates (e.g. Carson and Groves, 2007). Responding to concerns about hypothetical bias, many studies have compared revealed and stated preference data and found similarities in preference structures (Adamowicz et al., 1994; Boxall et al., 1996; Jones-Walters and Mulder, 2009). Finally, some authors raise an ethical argument against the reduction of the multiple values into a single numeraire that may ignore the intrinsic or moral value of a species existence (e.g. Edwards and Abivardi 1998; Hanley and Milne 1996). We take a pragmatic approach here, and argue that the benefits of valuation outweigh those of the objections (Nijkamp et al., 2008) and seek to develop policy-relevant monetary values for cost-benefit analysis, using a broader definition of biodiversity (focusing only on a single attribute level of biodiversity), with the qualification that the results should be interpreted as providing only one, and an incomplete, account of public values for biodiversity (Nunes and Van den Bergh, 2001).

2.2. Choice modelling theory

The analysis of choices made in the CE is based on McFadden's (1974) random utility model; where the utility function U can be separated into an observable systematic component V, and an unobservable error component ε (Hoyos, 2010). Hence;

$$U_{int} = V_{int} + \varepsilon_{int} \quad (1)$$

where, U_{int} is respondent n's utility of choosing option $i\left(i = 1, 2, \ldots, i^{th}\right)$ in choice situation $t\left(t = 1, 2, 3, \ldots t^{th}\right)$. It is assumed that alternative i is chosen over alternative j if $U_{int} > U_{jnt}$. The error term ε includes all unobserved variables that impact the utility of choosing a specific alternative. The observable component of utility can be expressed as two additive parts;

$$V_{int} = \alpha_{in} + \beta_{kn} X_{int} \quad (2)$$

where α_{in} is the alternative specific constant capturing the intrinsic preference for a choice alternative over and above the baseline. The β-terms are parameters to be estimated; β_{kn} reflects the preference parameters related to the attributes (k) in set X, therefore, X_{int} is a vector of observed attributes. To account for explanatory variables such as respondent and attitudinal characteristics, the observable component can be further expressed as;

$$V_{int} = \alpha_{in} + \beta_{kn} X_{int} + \gamma_i s_n \quad (3)$$

in this case, $\gamma_i s_n$ captures systematic preference heterogeneity as a function of individual characteristics (s_n). Thus creating a linear combination of observed and explanatory variables, specified as follows:

$$U_{int} = \alpha_{in} + \beta_n X + \gamma_i s_n + \varepsilon_{int} \quad (4)$$

Eq. (4) represents a basic conditional logit model with socio-economic characteristics.

However, estimates derived by the conditional logit model are criticised for not reflecting realistic situations (Hoyos, 2010; Train, 2003), because they do not represent random taste variations (i.e. those that cannot be linked to observed characteristics of the respondents). Moreover, conditional logit models are constrained by the independence of irrelevant alternatives (IIA) hypothesis which states that the relative probabilities of two options being selected is unaffected by the introduction of removal of other alternatives; i.e. probability of choosing alternative i over alternative j should not depend on whether some other alternative k is present or absent. In cases where the IIA hypothesis is violated, mixed logit models including random parameters can be estimated to account for taste heterogeneity in a population (Hoyos, 2010). Under the mixed model approach, the unobserved portion of utility is partitioned into two additive terms:

$$U_{int} = \alpha_{in} + \beta_n X + \gamma_i s_n + \varepsilon_{int} + \eta_{in} \quad (5)$$

The first term, ε is an unobservable error component of utility that is assumed to be an identically and independently distributed standard Gumbel distribution term. The second, η is the idiosyncratic random error-component, which enters the utility function of the hypothetical alternatives. The integral of Eq. (5) cannot be evaluated analytically and simulation maximum likelihood methods are used to estimate the mixed models (Train, 1999).

3. Methods

3.1. Study area

The research reported in this paper is based on data obtained from the City of Southampton, Hampshire, on the south coast of England. Southampton is the largest city in Hampshire with a population of approximately 250,000 people (SCC, 2015), and a population density of 4,858 people per square kilometre (ONS, 2013), the second highest population density outside of London (ONS, 2013). Characteristically for urban areas, green infrastructure within Southampton is limited and scattered throughout the city, with several elements of high natural environment quality, such as; Southampton Common, Central Parks, the Greenways Network, Waterside, and the internationally important River Itchen and River Test (SCC, 2008). All of these types of green infrastructure provide habitats for a diverse range of species, and are proven to support urban populations of protected species such as; bats, badgers, great crested newts and slow worms (Hand and Barker, n.d.). Southampton City Council owns and manages the majority of open spaces with special biodiversity interests, granting access to nature for urban residents (ibid.).

As part of the Partnership for Urban South Hampshire (PUSH, 2010), Southampton City Council (SCC) officially recognised the importance of biodiversity and is considering implementing green walls into the city centre to improve connectivity of the city's existing green infrastructure network (SCC, 2008). At present, there are few green walls scattered throughout the city, including; modular wall designs at University of Southampton halls of residence City Gate Way, and at a SCC building One Guildhall Square. The green wall at One Guildhall Square forms part of the Southampton City Centre Master Plan as a strategy to create a greener city centre (David Lock Associates, 2013). Southampton's high population density and characteristic green infrastructure network combined with SCC's interest in the adoption of green walls makes the city an ideal case study for this CE as a representation of other urban environments.

3.2. Attribute selection and choice design

The first step of a CE is to identify choice alternatives and their attributes. Four policies were identified and were described in the simplest terms possible (Table 1). Two green wall policies, a living wall and a green façade, were both deemed necessary inclusions in the choice experiment due to their contrasting aesthetic difference that may influence preferences. Green façades are based on the traditional approach to vertical greening, and involve the colonisation of woody or herbaceous climbing plants, rooted either to the

Table 1
Policy choice alternatives and descriptions presented to respondents in the CE.

Policy	Policy description
Living wall	Walls contain a soil like substance and can support a large number (more than three) of different plants growing at any height.
Green façade	Made up of one or two climbing plants growing up an existing vertical surface.
Alternative green policy	Investment into existing green and open spaces. Limit the enhancement of biodiversity to areas already supporting some level of biodiversity.
No green policy	No implementation of a new green policy to enhance urban biodiversity. Allow existing policies and/or strategies to continue and urban biodiversity to continue to decline.

ground or to elevated planter boxes, on a vertical surface such as a wall or trellis (Chiquet et al., 2013). Façades are therefore restricted in the variety of species they can support (Köhler, 2008; Manso and Castro-Gomes, 2015). Living walls, also referred to as modular green walls, are a more recent innovation and represent a more complex approach to vertical greening (Köhler, 2008). Formed from a modular system of planter boxes (or a similar structure containing a growing medium), often with an integrated water delivery and drainage system, living walls can support a wide variety of plants that can be anchored at any height (Francis and Lorimer, 2011). This enables the rapid and uniform coverage of a vertical surface (Köhler, 2008; Manso and Castro-Gomes, 2015).

Despite potential implications on biodiversity at a landscape scale, the descriptions presented to respondents for the green wall policies focussed on the impact the two designs had on urban plant biodiversity (Table 1). This follows findings from previous research that city dwellers generally have poor biodiversity identification skills and are better able to estimate plant biodiversity than the biodiversity of other taxa (Fuller et al., 2007; Shwartz et al., 2014). An alternative green policy was formulated and included as an opportunity for respondents to reject the two green wall policies but still choose to enhance urban biodiversity. The fourth option, labelled 'no green policy', enabled respondents to not fund any of the three additional 'green policies'.

To ensure the simplicity of choice options, only two attributes were included; biodiversity and cost. The biodiversity attribute level varied according to plausible levels (Table 2), i.e. the three 'green' policies do not result in a decline in biodiversity and 'no green wall' does not result in an increase in biodiversity.

The associated cost levels were defined according to the estimated cost to implement a green wall; approximately £5 to £13 per household per year for a living wall and £2 per household for a green façade.[3] The highest cost level of £25 represents any additional benefits obtained from the green wall policy. The survey included two versions of the fourth 'No green policy' alternative: either a zero payment in which biodiversity would decline, or a small payment of £2 in which the decline of biodiversity would be slowed down.

Table 2
Policy attributes and their levels.

Choice alternative	Attribute	Levels
Living wall	Urban Biodiversity	Increase biodiversity Maintain biodiversity
	Cost	£2 per household per year £10 per household per year £25 per household per year
Green façade	Urban Biodiversity	Increase biodiversity Maintain biodiversity
	Cost	£2 per household per year £10 per household per year £25 per household per year
Alternative green policy	Urban Biodiversity	Increase biodiversity Maintain biodiversity
	Cost	£2 per household per year £10 per household per year £25 per household per year
No green policy	Urban Biodiversity	Slow biodiversity decline Allow biodiversity decline
	Cost	£0 per household per year £2 per household per year

Attributes and levels were combined using SPSS Orthoplan to construct an orthogonal, fractional factorial main-effects design. Fractional factorial designs permit the statistical testing of several factors, without testing every combination of factor levels (Hoyos, 2010). The final design resulted in the formation of 16 unique choice sets split over two blocks, i.e. each respondent was asked to make eight choices. The text of an example choice card is included in Table 3–the original choice cards contained example photographs of the structures.

3.3. Data collection and analysis

The survey was conducted via face-to-face interviews between November 2014 and January 2015 at different street locations in Southampton city centre by four trained interviewers. The sampling strategy targeted adults visiting the city centre, because they were assumed to envision the implications of the proposed policies to establish a green wall in the city centre more easily than respondents outside of the centre. A convenience sampling strategy was employed. The interviewers approached people sequentially, so after each survey the next person who passed would be approached with an invitation to participate in the survey that did not mention the topic. Upon agreeing to participate respondents were asked to read an information sheet outlining the purpose of the CE, the importance and anonymity of their participation, and that the proposed location of the green wall was in the city centre. Although the potential for self-selection bias (Hudson et al., 2004) and non-response bias (Whitehead et al., 1993) is recognised, we minimised the risk of selecting mainly pro-environment respondents by sampling in the city centre in the busy shopping period leading up to Christmas, and using a short and neutral invitation. No record of

[3] These cost estimates are based on the assumptions that living walls costs between £750 to £2000 per square metre, and green façades approximately £280 per square metre (Growing Green Guide, 2013), that the green wall proposed is 120 m long and 5 m high, and a population of 91,217 households in Southampton (SCC, 2008). The proposed dimensions of the green wall were not used for survey purposes following feedback from the pilot survey.

Table 3
An example choice set.

Policy option	Living wall	Green façade	Alternative green policy	No green policy
Biodiversity	Increase level of urban biodiversity	Maintain level of urban biodiversity	Maintain urban biodiversity at current level	Allow urban biodiversity to decline
Cost Check only one box:	£25	£10	£10	£0

non-response was kept, but we estimate that the response rate was around 20%, and choice to decline participation was made before the topic of the survey was mentioned.

The questionnaire administered to respondents consisted of three sections.[4] The first section included open and closed questions to establish opinions, attitudes and knowledge of green walls and their benefits to the urban environment. It was also used to introduce and define green walls, biodiversity and subsequent benefits, all of which may have been unfamiliar to some respondents. Visual aids in the form of photographs and diagrams were used to support definitions. Biodiversity was explained as 'the biological variety in an environment, indicated by the number of different species of plants and animals and habitats present. A high level of biodiversity is proven to benefit human health'. For the purpose of the CE all respondents were asked to consider this definition when making their decision. No reference to other ecosystem service benefits of green walls was made. At the end of the section, respondents were asked whether they would be willing to pay for green walls within their local area.

The second section was the CE exercise; the policy options, the attributes, and the cost of policy implementation were described to the respondents. It was explained that the cost would be a reoccurring annual payment covered through a budget reallocation decision made by the council, which would amount to X amount of their annual Council Tax bill. During the pilot study, this payment vehicle was tested using various descriptions to ensure understanding was clear. It was emphasised to participants that as a budget reallocation other policies currently receiving funding through their Council Tax would undergo a cut to account for their decision, and that choosing a lower cost scheme would translate into them wanting less of their Council Tax allocated to the proposed policies. Participants were presented with examples of services funded by SCC and asked to consider these services, or services they know to be SCC funded, when making their decision. This "bundle" approach to budget reallocation is consistent with the payment vehicles adopted in previous studies (e.g. Bergstrom et al., 2004; Swallow and McGonagle, 2006). By bundling all services together respondents are making a trade-off between more green policies, and less of all other public services. Respondents were then randomly allocated a version (block) of the choice sets and asked to select their preferred policy from each of the eight choice cards, where they were reminded to consider the policy design, impact on biodiversity and differing cost when making their choice. The third and final section included questions to collect socioeconomic data. All definitions and descriptions used in the final questionnaire were refined based upon feedback from nine respondents in a pilot study. Feedback indicated that the structure of the questionnaire was clear, but definitions for both biodiversity and green walls needed to be simplified and more pictorial evidence was recommended.

Data was analysed using the R software (version 3.1.2). Both mixed and conditional logit models were estimated. Mixed logit models were simulated using Halton draws with 1000 replications. The parameter estimates from the models were used to obtain marginal WTP for selected attributes using the following formula;

$$\mathrm{WTP} = \frac{\beta_{\mathrm{Attribute}}}{\beta_{\mathrm{M}}} \quad (6)$$

where $\beta_{\mathrm{Attribute}}$ is the coefficient on the attribute of interest and βM is the negative coefficient of the cost variable. The ninety-five percent confidence intervals around the mean WTP values were then estimated using Krinsky and Robb (1986) bootstrapping procedures with 2000 draws.

[4] Questionnaire provided in the appendix

Table 4
Socio-demographic characteristics of respondents/household.

Characteristic	Percentage of sample population	Percentage of Southampton population
Age:		
18–24	28	21
25–34	28	21
35–44	14	16
45–54	14	14
55–64	2	11
65+	14	17
Gender:		
Male	44	50
Qualification attainment:[a]		
No qualifications	0	7
Level 1	12	10
Level 2	17	16
Level 3	14	25
≥Level 4	56	34
Other	1	8
Employment status:		
Full time employment	43	40
Part time employment	14	17
Unemployed	4	15
Student	35	11
Retired	4	16
Annual household income: [b]		
<£20,000	26	N/A
£20,000–£40,000	20	N/A
£40,000–£60,000	26	N/A
£60,000–£80,000	12	N/A
£80,000–£100,000	6	N/A
>£100,000	2	N/A
Prefer not to say	8	N/A

[a] Southampton population figures from 2014 data. All other figures from 2011 data.
[b] Categorical data for household income was not available from the census. Median gross weekly earnings to be £401 per resident in employment (CI% = ∓ 3.9) (ONS, 2014). Extrapolating this across the year provides gross annual earnings of £20,852 per resident in employment. Although this value falls within the median category of the sample population (median = 2.5, S.E = ∓ 0.10), the two cannot be accurately compared as underlying assumptions regarding household structure would have to be made.

4. Results

4.1. Sample statistics

A sample totalling 127 respondents completed the survey, each representing a different household. There were an additional 23 partially completed surveys (because of time constraints or aversion to provide socio-economic data), all of which were disregarded. None of the participants objected to the payment vehicle or any other part of the hypothetical scenario.

The sample population was made up of 44% males, and had modal education level of greater than or equal to level 4 of the European Qualification Framework (European Commission, 2016). The largest proportion of respondents were in full time employment (43%). The socio-demographic characteristics of the sample population were compared to the Southampton population (Table 4), using both 2011 census data (ONS, 2011) and 2014 annual survey data (ONS, 2014), where available. Analysis suggests that the sample is mostly representative of the Southampton population, conforming to the census population for both age (Chi-squared test; $\chi^2 = 10.275$, d.f. = 5, P = 0.068) and gender (Chi-squared test; $\chi^2 = 2.036$, d.f. = 1, P = 0.154). However, the sample population has a greater representation of people with qualifications greater than level 4 compared to the census data, and a lower representation

Table 5
Attitudinal characteristics of respondents/respondents household.

Attitudinal characteristic	Number of respondents	Percentage of respondents
Know what a green wall is:		
Yes	70	55
Find green walls aesthetically pleasing:		
Yes	115	91
Design preference:		
Modular green wall	86	68
Green façade	36	28
Neither	5	4
Aware of benefits green walls may provide:		
Yes	68	54
Familiar with the term biodiversity:		
Yes	97	76

of people with no qualifications (Chi-squared test; $\chi^2 = 391$, d.f. = 5, $P < 0.001$). This is likely to a be a result of a slight overrepresentation of the student population within the sample, in addition to a slight underrepresentation of the unemployed (Chi-squared test; $\chi^2 = 42.775$, d.f. = 4, $P < 0.001$).

Examination of attitudinal characteristics (Table 5) indicated that just under half of the sample population (n = 70) knew what a green wall was prior to completing the CE. The remaining 57 respondents were unfamiliar with the design concept; despite this, approximately 86% of those unfamiliar with green walls still found them to be aesthetically pleasing when presented with the images used in the CE.

A total of 115 respondents found green walls to be aesthetically pleasing, the remaining 12 respondents were either unsure or did not find green walls aesthetically pleasing. In the CE, all 115 respondents who found green walls to be aesthetically pleasing did select a preferred green wall design; with 81 respondents selecting the living wall, and the remaining 34 preferring the green façade.

Only 6% of choices in the CE were for the no-change option (no payment into green infrastructure, continued biodiversity decline). Finally, 68 respondents were aware of at least one benefit of green walls in cities and 97 of all respondents said to be familiar with the term biodiversity, subsequently understanding of this term was generally not found to be a limiting factor in this CE.

4.2. Model estimates

Two generic model specifications were estimated: an attributes-only model (Model A) and a model including individual-specific variables interacted with the policy options (Model B). Results of the Hausman and McFadden test (1984) led to reject the IIA assumption for both specifications of the conditional logit models. Here we present the estimates of the best fit mixed logit models, according to the Log likelihood and Akaike Information Criterion (AIC) (Table 6).

Both models were significant according to the results of the Likelihood ratio test (P > 0.001, in both cases). As many of the attributes are categorical, the model includes dummy variables: three for the different policy options using no green wall as the baseline, and two for biodiversity. Because of the design restrictions, the level "maintain biodiversity" is the baseline for the green policies against which the parameter of biodiversity increase should be compared, and the "decline in biodiversity" is the baseline against which the parameter of the "slow decline" can be interpreted for the no-green policy options. For the two biodiversity variables and the living wall options, random parameters were estimated, indicating that there is significant unexplained preference heterogeneity among the respondents in the sample. For the other two options (green façade and alternative policy), taste heterogeneity was not statistically significant. The cost attribute is a numerical variable, for which a fixed parameter is estimated.

Table 6
Choice model estimates.

Attribute	Mixed logit Models	
	Model A[a]	Model B[b]
Policy characteristics: fixed effects		
Green façade	1.977*** (5.92)	1.084* (1.94)
Alternative policy	1.592*** (4.77)	1.608*** (4.88)
Cost	−0.061*** (−4.93)	−0.061*** (−4.93)
Policy characteristics: random effects		
Increase biodiversity	1.094*** (6.37)	0.588** (2.24)
(s.d. normal distribution)	1.071*** (6.00)	0.999*** (5.75)
Living wall	2.195*** (6.41)	1.309** (2.38)
(s.d. normal distribution)	0.696*** (4.57)	0.658*** (4.21)
Slowing biodiversity decline	−3.038** (2.47)	−2.925*** (2.60)
(s.d. normal distribution)	3.413*** (4.03)	3.188*** (4.34)
Respondent characteristics:		
Increase in biodiversity * knowledge biodiversity meaning (dummy: 1 if known, 0 otherwise)		0.672** (2.31)
Living wall/Green façade * aesthetically pleasing (dummy: 1 if pleasing, 0 otherwise)		1.011** (2.44)
Model statistics:		
Log-likelihood	−1151.61	−1141.42
Adjusted R2	0.18	0.18
AIC value	2321	2304
Number of observations	1016	1016

Asterisks indicate significance: ***for <0. 1%, **for <1% and *for <5%.
[a]Model A: attributes-only model.
[b]Model B: individual-specific variables interacted with the policy options.

The estimated β-coefficents display the expected signs for the attributes, with positive utility associated with the three policy options and 'increase biodiversity' and negative utility associated with increasing cost and 'slowing biodiversity decline'. The three policy options are significant, suggesting that a green policy design is a significant determinant of choice associated over the no green policy scenario. The relative weighting of the policy options remains the same in Model A and Model B. Respondents place a higher level of utility on the living wall design, followed by the green façade, and lastly the alternative green policy. The parameters of the living wall and green façade are expected to reflect other benefits associated with green infrastructure, including aesthetic benefits.

Policy characteristics in both Model A and Model B are significant. The positive β-coefficients attached to the policy characteristic 'increase biodiversity' indicates that respondents are more likely to select a policy that enhanced biodiversity compared to a policy that maintains the current level of biodiversity. The negative utility associated with the policy characteristic 'slowing biodiversity decline' suggests that for the 'no green policy' option, respondents attach a lower level of utility to slowing biodiversity decline against a £2 payment than to the current decline without payment. However, the standard deviation of the random parameter is larger than its mean, which indicates that for a small proportion of the respondents the option to pay £2 in order to slow down the decline in biodiversity loss has positive benefits. The negative β-coefficient for the cost parameter is as theoretically expected, and shows that the probability of selecting a policy decreases with an increase in policy cost.

The lower AIC value of Model B compared to Model A indicates that accounting for interactions with attitudinal characteristics within the model results in a better fit. The β-coefficients attached to the respondent characteristics in Model B are significant determinants of choice, capturing further respondent heterogeneity in fixed effects, and reflect the probability of a household choosing one

Table 7
Mean WTP values for policies and policy attributes.

Attribute	Model A[a]			Model B[b]		
	Mean WTP	95% CI		Mean WTP	95% CI	
		LB	UB		LB	UB
Policy options:						
Living Wall	37.88	23.47	61.66	22.49	4.6	45.8
Green façade	34.1	21	55.49	18.64	0.73	41.45
Alternative policy	27.52	15.39	46.69	27.46	15.43	46.12
Policy characteristic:						
Increase biodiversity	18.44	13.75	25.84	9.86	1.24	19.14
Slow biodiversity decline	-109.63	-52.78	-9.12	-50.14	-102.11	-10.98
Attitudinal characteristic:						
Know biodiversity				11.31	1.3	22.87
Aesthetically pleasing				17.14	3.64	33.43

Notes: CI confidence interval, LB lower bound, UB upper bound, WTP willingness to pay.
[a] Modal A: attributes-only model.
[b] Model B: individual-specific variables interacted with the policy options.

of the hypothetical scenarios relative to a no green policy option.[5] In Model B, the attitudinal characteristic 'know biodiversity' is significant and captures respondent heterogeneity. The 'know green wall' variable was included in the model as a dummy variable taking the value one for respondents who stated to know what biodiversity was. These respondents attach a higher value to the three green policies. Furthermore, respondents who stated that they find green walls aesthetically pleasing are more likely to choose the living green wall or green façade than other respondents.

4.3. Willingness to pay estimates

Using Eq. (6), WTP values were estimated for both models (Table 7). Model A's marginal mean values for policy options range from £27.52 per household per year for the alternative green policy, to £34.10 per household per year for the green façade, and £37.88 for the living wall. WTP based on Model B results are very similar at £27.46, £38.23 and £38.08 respectively, when using the sample mean of opinion on the aesthetics of green walls for the green façade and living wall estimates. The differences across these WTP estimates are not statistically significant. Model B results were used to obtain a value representative of the average household for three different scenarios. The first scenario is a living wall policy, the second a green façade and the final an alternative green policy; in all scenarios an increase in biodiversity was assumed. For all scenarios, the sample means for 'knowledge of biodiversity' of 0.76, and 'aesthetically pleasing' of 0.91 was used.

Aggregating these scenarios WTP estimates across the entire region of Southampton, of approximately 91,217 households, results in an aggregated annual WTP of £5.2 million for a living wall policy that increases biodiversity, and £4.8 million for a green façade (Table 8).

Table 8
Aggregate WTP values for policy and policy attributes across Southampton (million £).

Attribute	Mean WTP	95% CI	
		LB	UB
Living wall	5.16	3.64	7.70
Green façade	4.81	3.36	7.15
Alternative policy	3.40	2.07	5.32

Notes: CI confidence interval, LB lower bound, UB upper bound, WTP willingness to pay.

5. Discussion

5.1. Public value of green walls

All green policies increased utility significantly over the no-green policy scenario, which indicates a positive value associated with green infrastructure that increases biodiversity in Southampton. However, the difference between the welfare estimates is not significant. The statistical insignificant outcome may be due to lack of statistical power, associated with a relatively small sample, but it may also reflect true indifference toward the three green infrastructure policy options (Christie et al., 2006). The distinction between these potential explanations has important policy implications. The first suggests that the public do have identifiable, although similar, preferences towards the policies and therefore these preferences should be taken into account during the policy formation (Hanley et al., 2002). In the case of the second, it would appear the public does not have specific or identifiable preferences, and therefore policy decisions whether to implement urban biodiversity infrastructures can be made without further reference to public opinion (Christie et al., 2006). However, the significant β-coefficients (Table 6) for attitudinal and policy characteristics provide some indication of identifiable preferences between policy designs. Individuals who knew what biodiversity was prior to the CE display a significant preference towards the living wall. Willingness to pay also varies with opinion on the aesthetics of green walls. The presence of these identifiable preferences implies that local authorities such as SCC should continue consultation with the public throughout the planning and decision making process. Furthermore, the results show that the public attaches a significant utility to increasing biodiversity when compared to maintaining it or slowing down the on-going biodiversity decline, which amounts to approximately £10 per household (Table 7), up to £21 per household for those who are familiar with the concept of biodiversity. There is the possibility that positive attitudes to biodiversity are a result of self-selection bias; however, our results support those from previous studies (e.g. Christie et al., 2006; Spash et al., 2009). Potential bias due to a sense of inconsequentiality of the survey cannot be ruled out either as we did not include control questions to explore this aspect.

The aggregate value for both green wall policies across Southampton is substantial (Table 8). It was calculated that a proposed scenario of a 120 m by 5 m living wall would cost approx-

[5] To ensure this trend was not a result of the slight over-representation of the student population (Table 4), an additional model was run extending model B with a student dummy variable. The interaction of the student variable with policy options was insignificant. Results are available from the authors upon request.

imately £450,000 to £1,200,000, whilst the cheaper alternative of a green façade would cost approximately £168,000 (Growing Green Guide, 2013). In both cases, the aggregate WTP obtained surpasses the respective estimated investment costs, resulting in a net economic benefit. Under the assumption of this CE the WTP for more complex living wall designs is the same as more simplistic alternative. Consequently, a more complex and expensive living wall will produce a lower net benefit compared to a cheaper and simpler living wall design; approximately £3,960,000 and £4,710,000 respectively. The net benefit obtained from a green façade is approximately £4,642,000. The greatest net benefit is obtained from a simple living wall design, it costs approximately £470 less per square meter to implement a green façade (Growing Green Guide, 2013). In our proposed scenario this equates to an additional cost of £282,000 for a £68,000 gain in net benefits.

Aggregate estimates (Table 8) are based on the assumption that all households in Southampton will benefit from the proposed green wall policy. Hence, we assume that the sample is representative for the total population of Southampton, also in terms of preferences towards the policies and frequency of visiting the city centre, and that WTP also contains non-use values, especially for biodiversity, that exist alongside use values associated with the appreciation of the aesthetic quality of green walls through visiting the centre or other ecosystem services benefits. Although this was not discussed in the survey, green walls aim to create ecological and biodiversity networks, which would mean that a green wall in the centre would have wider benefits. However, further research into the impacts of green walls are needed to assess biodiversity impacts at the wider urban scale. At the same time, we ignore the benefits enjoyed by visitors to Southampton and living outside the city boundaries who may benefit but not incur any cost. Overall, this study highlights how the use of CE to obtain monetary indicators can aid informed decision making and, in the case of local authorities, justify the allocation of limited public resources.

When utilising monetary values of biodiversity, local authorities must recognise that there is an ongoing debate regarding whether biodiversity, along with other elements of the environment, should, and reliably can, be assessed in monetary terms. We therefore recommend, especially at local level decision-making, to combine these stated preference valuation results with other decision-criteria, including public consultations such as citizen juries and non-monetary methods (Turner and Schaafsma 2015; Lienhoop et al., 2015).

The payment vehicle adopted for this study, a tax reallocation, is less conventional than raised taxes. Bergstrom et al. (2004), Nunes and Travisi (2009) and Swallow and McGonagle (2006) have found that the use of budget reallocation results in higher WTP payments compared to an increase in taxation, whereas results from Kontoleon et al. (2005) show that these payment vehicles result in statistically equivalent WTP estimates. Secondly, by bundling all public services into one public good, we cannot be certain of the exact services respondents considered to focus on and alter funding to when making their decision. This may affect the precision of the WTP estimates. However, this is not too dissimilar to the limitations associated with using a more conventional payment vehicle such as an increase in taxation (Morrison and MacDonald, 2011), and provisional results from Nunes and Travisi (2009) suggest that WTP from a budget reallocation does not depend upon the budget sources.

It should be further noted, that for simplicity our CE included only biodiversity impacts and costs as design elements, but the size and location of the proposed green policies was not specified. Through combining biodiversity benefits we have addressed limitations that cognitive burden can have on result validity (Hanley et al., 2002). However, as a result we are unable to disentangle values that individuals may attach to green infrastructure elements, such as a specific element of the design or aesthetics. The valuation of other green wall design elements or alternative green infrastructure options is an important area for future research. For example, our biodiversity attribute could be further specified. Previous valuation studies have used anthropogenic qualifications of biodiversity such as: “cuteness”, “perceived rarity”, and “charisma” in the CE design (Christie et al., 2006). It has been recognised that charismatic and flagship species attract significantly higher WTP (Christie et al., 2006; White et al., 2001). Future studies can be used to assess preferences towards green walls with ‘beautiful, aesthetically pleasing’ plants, designs that support wildlife, or other anthropogenic qualifications and their influence on the level of utility attached to green urban policies.

Welfare estimates obtained from this study strengthen the case for the implementation of green walls within cities. However, green walls are by no means the silver bullet for biodiversity enhancement in an urbanising world, and thus far attempts to increase biodiversity in cities has been narrowly focused at a project scale addressing only the immediate issues rather than the wider social and ecological patterns and processes (Hostetler et al., 2011). Green walls should be implemented as a technique to complement and create a valuable network of green spaces (Francis and Lorimer 2011). The adoption of a landscape scale management plan will ensure the stability of biodiversity but the integration of green walls in such plans represents a new frontier in applied ecology (Francis and Lorimer 2011). Further investigation into the strategic planning of green walls needs to be undertaken to ensure habitat heterogeneity is maximised at a landscape scale (Goddard et al., 2010).

6. Conclusion

To our knowledge, this paper presents the first study to use a choice experiment to value the public benefits associated with urban green walls and their contribution to urban biodiversity, and only the third empirical monetary valuation study of green walls after Veisten et al. (2012) and Perini and Rosasco (2013). On the basis of this study, which sampled a relatively small subset of the population, the Southampton general public attach a net economic benefit to the implementation of a green wall, within the city centre. However, green walls are only a single element of a green infrastructure network, and further research is recommended to ensure that their design and implementation complements existing green infrastructure to create a functional and stable network of biodiversity, at both a site and landscape scale, essential for the long-term provision of ecosystem services (Francis and Lorimer 2011; Goddard et al., 2010). The welfare estimates obtained are likely to be specific to Southampton; however, the methodology and broader policy implications of this research are applicable to other locations.

Despite recognised limitations of the use of monetary valuation studies for decision-making on biodiversity, this study demonstrates that valuing biodiversity provides benefits and using such information is more transparent in a policy and decision making context than the absence of it. Therefore, it is advisable that local authorities stimulate further research into public preferences using stated preference methods to value the benefits of green walls as well as other elements of green infrastructure in their area. Such efforts may help to increase the reliability and validity of WTP estimates, and thereby inform alternative financing instruments for urban green infrastructure.

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