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Local terrestrial biodiversity impacts in life cycle assessment

A case study of sedum roofs in London, UK

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Abstract

Urban development is a key driver of global biodiversity loss. “Green” infrastructure is integrated to offset some impacts of development on ecosystem quality by supporting urban biodiversity, a prominent example being green roofs. The effects of green infrastructures on urban biodiversity are not well understood and poorly included in life cycle assessment (LCA) methodology. Here, we present a novel methodology that quantifies the local impact of green infrastructures on terrestrial biodiversity—demonstrated here for sedum roofs in London, UK—and integrates within LCA. It relates energy provision by plants to the metabolic requirements of animals to determine what species richness (number of species) and species abundance (number of individuals) are supported. We demonstrate this methodology using a case study, comparing the life cycle impact of developing 18 buildings, with either asphalt concrete or sedum roofs, on ecosystem quality. We found the sedum roofs (0.018 km^2) support 53 species (673 individuals), equivalent to 1.3% of the development's life cycle impacts on ecosystem quality. Complete offsetting requires considerable reduction in transport use throughout the development's lifetime, and lower environmental impact material selection during construction (contributing 98% and 2%, respectively). The results indicate sedum roofs offer minor impact mitigation capacities in the context of urban development, and this capacity is limited for all green infrastructures by species richness in local species pools. This paper demonstrates the potential and limitations of quantifying terrestrial biodiversity offsets offered by green infrastructures alongside urbanization, and the need for realistic expectations of what role it might play in sustainable urban design.

KEYWORDS
biodiversity, green roofs, industrial ecology, life cycle assessment, nature-based solutions, urbanization

1 INTRODUCTION

Urbanization is driving global land-use change, and through this threatening biodiversity (Hasan et al., 2020) and the provision of natural resources (Cardinale et al., 2012). Over 50% of the global population now live in urban areas (Watson, 1993); and this is expected to increase to 86% by 2050 in the Global North (Nations et al., 2018). As urban populations and their affluence (Chertow, 2000) grow, so too will their demands for urban development (Rao & Baer, 2012): the construction of gray (i.e., residential buildings) and supporting infrastructure (e.g., roads) needed for urban living, and green (vegetation-containing) and blue (aquatic) spaces that often enhance it (Erell et al., 2012; Wu et al., 2019). In the context of urbanization, we define land-use change as a change from one land-use category to another (e.g., "natural" to "residential" area) and changes within the same land-use category (e.g., "slum" to "residential buildings" in a residential area). In both cases, land-use change can have a considerable effect on terrestrial biodiversity (Haines-Young, 2009; Hansen et al., 2012). In addition to its inherent value, biodiversity contributes to the provision of natural resources (e.g., timber, minerals, and water) and services (e.g., flood defense and air purification) on which socioeconomic systems depend (Mace et al., 2011; Grizzetti et al., 2018; Smith et al., 2013); as such, urbanization threatens the resources upon which humans rely (Arnold et al., 2011; Kang et al., 2022). With urbanization expected to rise, these problems are set to worsen unless the needs that underpin biodiversity—from archaea to animals—are satisfied alongside those of humans.

The concept of "sustainable urban design" aims to meet the growing demand for urban infrastructure in ways that benefit humans and biodiversity simultaneously. The construction of urban green infrastructures is at the fore of sustainable urban design. They employ ecosystem components—plants, water, and geological materials like soil and rock—to address challenges relating to climate change, food and water security, and human health, benefiting humans and nature simultaneously (Stafford et al., 2021; Stange et al., 2022); challenges that drive their application (Mitsch, 2012). A familiar example is green roofs, but other examples include urban wetlands (e.g., as sustainable urban drainage systems, SuDS), urban forests, and hedge-based ecological corridors; each of which confer concurrent benefits across socioeconomic and ecological systems (Escobedo et al., 2019; Frantzeskaki, 2019; Stefanakis, 2019). Green roofs are now common features in urban landscapes, with government and legislative bodies encouraging their use (Davenport et al., 2021). Amongst their many reported benefits, green roofs support biodiversity including in densely populated urban areas (Arnold et al., 2011). However, the extent of this support beyond their constituent plants remains poorly understood, as do the mechanisms underlying this benefit. Studies have focused on plants and arthropods (e.g., invertebrates) (Wang et al., 2022), with limited research on higher trophic levels (Williams et al., 2014). This knowledge gap undermines our ability to assess the biodiversity impacts of green roofs or their efficacy in contributing to sustainable urban design (Curran et al., 2016).

Life cycle assessments (LCAs) inform the design and application of nature-based solutions (NbS) technologies, but the impact pathways between interventions and species loss are not yet fully described. The problem is compounded by poor transference of ecological data into LCA, as the complexity of ecological systems makes responses to human intervention difficult to generalize in ways compatible with LCA. There is a need to more completely characterize biodiversity impacts in the LCA framework, including more diverse anthropogenic pressures like different urban land cover types (Souza et al., 2015). Winter et al. (2017) encourage the inclusion of additional environmental pressures and the development of new midpoint and endpoint categories pertaining to species and ecosystem diversity. In their critical review of LCA literature, Larrey-Lassalle et al. (2022) identify the importance of distinguishing local (in situ) and global (ex situ) life cycle impacts. This is critical to determine the effect of technology-specific resource provision on biodiversity when assessing NbS. This aspect is largely omitted in LCA literature. An exception being Brachet et al. (2019), whose attempt to analyze in situ impacts specifically revealed ReCiPe 2016 and IMPACT World+ do not link resource provision, through land use (habitat creation) at the midpoint, to biodiversity loss (or gain) at the endpoint.

A secondary issue is that estimates of biodiversity loss due to land use are traditionally based on species–area relationships (SARs). Many issues have been identified with using SARs (e.g., (Dengler, 2008; Dolnik & Breuer, 2008; Fattorini & Borges, 2012; Triantis et al., 2003; Turner & Tjørve, 2005)), and many authors have proposed alternative approaches (Adler et al., 2005; Chaudhary et al., 2015; Geyer et al., 2010; Koh & Ghazoul, 2010; Triantis et al., 2012), but the use of SARs in LCA persists. Alejandre et al (2022) demonstrate a pertinent advancement. They relate land cover type to pollinator abundance, establishing a new midpoint indicator for LCA. In doing so, ecological expertise—here data (e.g., IPBES, 2016) and opinion on pollinator abundances—is leveraged to resolve the issues discussed. However, their restricted scope, although providing a tractable means for estimating wider biodiversity impacts, does not consider the impact of land occupation on biodiversity at the system level. Furthermore, their predictions offer less robust estimates of biodiversity change than, for example, energetics.

There remains a need to characterize impacts of land use on biodiversity at the endpoint across, particularly in the urban context. The types of urban land use described in LCA methods (e.g., ReCiPe 2016, Huijbregts et al., 2017; IMPACT World+, Bulle et al., 2019) are limited. Characterization should reflect the varied design of NbS as urban infrastructure and describe their effects across more comprehensive trophic systems. By modeling local ecosystems explicitly, the in situ impacts of urban land use on terrestrial biodiversity can be quantitatively characterized and distinguished from ex situ impacts for analysis. The impacts of biodiversity-supporting urban infrastructure on local ecosystem quality can then be more comprehensively assessed.

The ability to estimate natural resource use within a local species population provides a basis for this advancement. Natural resource use is typically modeled and assessed according to its ability to satisfy human needs. Yet like humans, plants and animals also have physiological (and sometimes social) needs (Mason et al., 2022). Until recently (Mason et al., 2022), a set of natural resource-based well-being needs, equivalent to those for humans (Rao & Baer, 2012; Rao & Min, 2018), had not been described for nature. This concept, which we termed “ecological needs” (Mason et al., 2022), has not yet been incorporated into system analysis methodologies like LCA. Here, we build upon the concept of ecological needs to addresses several deficiencies identified in LCA in a novel way.

In this paper, we model the local terrestrial biodiversity impacts associated with the life cycle of an urban development in London, UK. We approach the problem in three steps: (1) Demonstrate through novel methodology how the natural resource use of terrestrial species can be modeled to produce a more comprehensive inventory of socioeconomic and ecosystem activities. (2) Integrate the results into the LCA framework, focusing on how sedum roofs affect the life cycle impact of developing the study on ecosystem quality. (3) Discuss how our methodology can be used to develop a new characterization factor linking urban land occupation as “urban, sedum roof, London” to local terrestrial biodiversity change. Together, these improvements represent a means of a better understanding of how urban green infrastructures can support biodiversity alongside humans, advancing our ability to design and create more sustainable urban environments (Tanguay et al., 2010).

2 | METHODOLOGY

2.1 | Case study area: Meridian Water Development in London, UK

The Meridian Water Development (MWD) is a four-stage development plan for the construction of 10,000 homes and a train station across a 0.85 km^2 site in Enfield, London, UK (Figure 1a,b). The total roofed area is 0.018 km^2. The expected construction duration of the MWD is 20–25 years, with work on the first of its four stages, Meridian One, underway since 2021. Meridian One is the case study area in this paper. As the first stage to be developed, building designs for Meridian One are available for assessment and less likely to be altered than later development stages.

The MWD site is currently classified as brownfield land, having previously been developed for industrial use (Alker et al., 2000). The first development stage, Meridian One, will comprise 18 residential buildings encompassing 725 apartments (Figure 1c), and mixed-use infrastructure: retail, leisure, community, parking, and paving. Green roofs feature prominently in current site designs. Since designs are often more ambitious than the realized product, we consider two scenarios for the construction of Meridian One: (1) the construction of 18 residential buildings with asphalt concrete, and (2) the construction of 18 residential buildings with extensive sedum roofs consisting of sedums and shallow-depth substrate (Fig. A1, Supporting Information S1).

Two other main types of green roofs exist: semi-intensive and intensive, which feature greater substrate depths and more diverse planting; with intensive green roofs able to facilitate more substantial planting, including small shrubs (Jusselme et al., 2019; Passaseo et al., 2020). Of these designs, sedum roofs are the least bio-productive and constitute the poorest habitat. However, extensive sedum roofs are chosen for this case study as they satisfy building requirements for “biodiverse roofs” at lowest cost, making them more likely to be installed at Meridian One than the other green roof designs (Brachet et al., 2019).

2.2 | Scope of life cycle assessment

The goal of this LCA study is to compare the life cycle impacts of 18 residential buildings occupying Meridian One and the transformation and occupation of land surrounding them (within the bounds of Meridian One) on ecosystem quality under two design scenarios: (1) construction with asphalt concrete roofs, and (2) construction with (extensive) sedum roofs. The functional unit is 18 residential buildings (725 apartments), including the foundations, superstructure, and roofed areas, but not the contents of the apartments (furniture, etc.). We assess the lifetime impacts of Meridian One from cradle-to-grave, across the following life cycle stages: construction, use, and end-of-life. Figure 2, a LCA process flow diagram for this case study, describes the processes that make-up our assessment across its three life cycle stages.

We use structural design schematics, describing the planned residential infrastructures, collected from the developers alongside expert knowledge on structural design; and data from ecoinvent v.3.8 to produce an inventory of materials needed to develop Meridian One. LCA modeling was carried out in OpenLCA v.1.11.0, using endpoints in the IMPACT World+ method (Bulle et al., 2019). Endpoints are chosen since we are interested in biodiversity loss and gain, which is not reported at the midpoint level. Sixteen impact categories contribute to the area of protection “ecosystem quality” in IMPACT World+. Therein, the PDF m^2 year indicators quantify the disappearance of species (PDF) over a given surface (m^2) during a certain time (year) (Bulle et al., 2019; Huijbregts et al., 2017; Jolliet et al., 2003). Finally, as species loss is being described, a larger impact value represents a greater loss. Conversely, a negative value represents a gain. We considered but did not use ReCiPe (2016) methodology; it does not capture land transformation to urban land-use types in OpenLCA with ecoinvent v.3.8 (it is limited to the following land-use transformations: to/from primary and forest, natural grassland, scrubland, and inland wetland; National Institute for Public Health and the Environment, 2017).

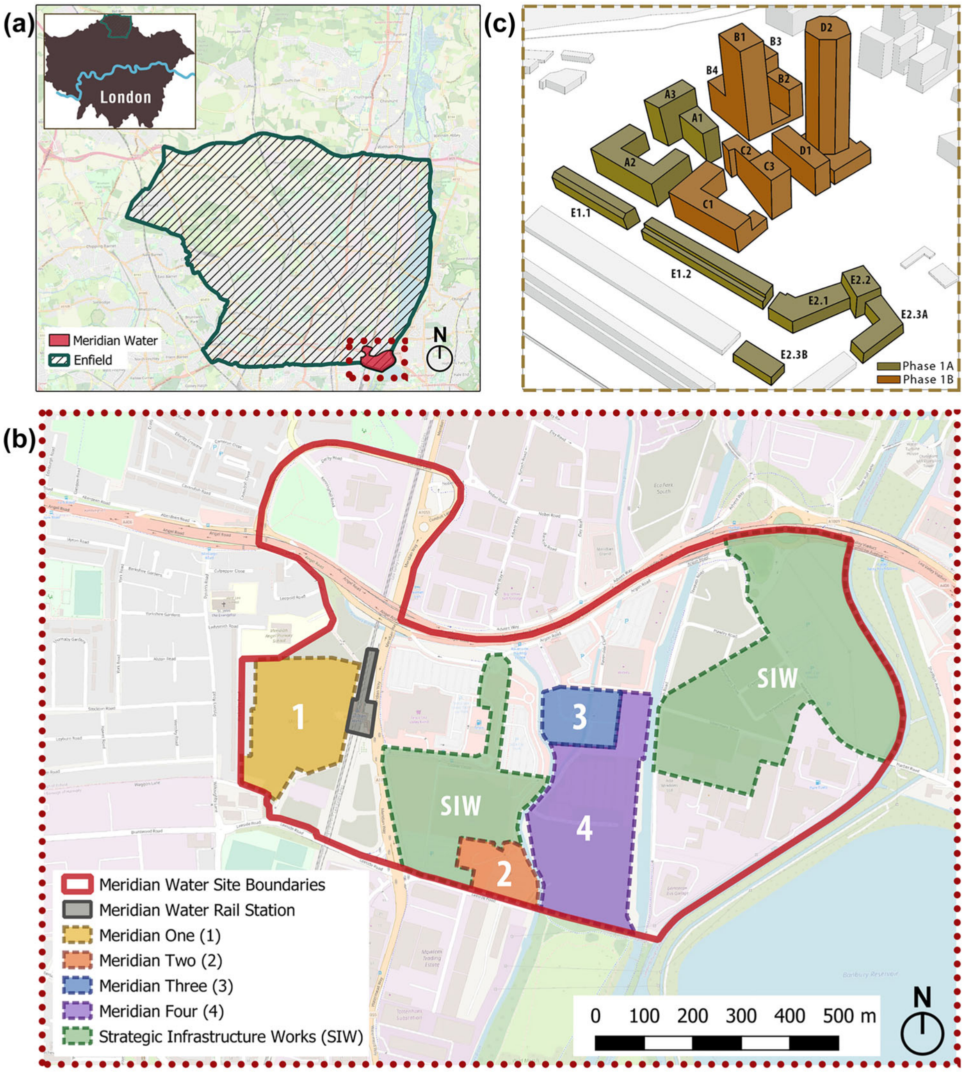

FIGURE 1 Site of the Meridian Water Development (MWD) in Enfield, London, UK, showing (a) the location of Enfield (borough) within the Greater Metropolitan London area, (b) the development stages of the MWD, and (c) the 18 residential buildings being constructed in Meridian One, each assigned an individual "building ID." These IDs are used to disaggregate material use for construction in Table S1, Supporting Information S2. Parts (a) and (b) are drawn using QGIS software; (c) is adapted from a visualization by Hawkins\Brown and HTA design LLP (Ing, 2022).

2.2.1 | Site-specific considerations

Owing to its location (London, UK), the local climate at Meridian One is oceanic temperate and the reference land-use type is "European broadleaf woodland" (Hickler et al., 2012; Kottek et al., 2006). Three land-use types will exist throughout the site's lifetime: "urban, fallow" (brownfield), "urban, continuously built" (non-roofed and cement roofed areas), and "urban, sedum roof, London," which we seek to describe in this work. Fig. A5 (Appendix S8, Supporting Information S1) details the chronology of these land-use types at Meridian One. As the study seeks to quantify the impacts of installing sedum roofs (rather than concrete roofs) at Meridian One, we assume that the areas surrounding the 18 residential buildings (Figure 1c) are "urban, continuously built" in both design scenarios. That is, the roofs' design is the independent variable, and the sedum roofs are the only green infrastructure present on site.

2.3 | Life cycle inventory

Table 1 describes the quantities of materials used during the 100-year lifetime of our case study at Meridian One with either asphalt concrete or sedum roofs (see Figure 1c and Fig. A1, Appendix S1, Supporting Information S1), including the production of vegetation per Figure 2. All underlying data, calculations, and assumptions are presented in the Appendix S1, Supporting Information S1, and Supporting Information S2.

TABLE 1 Quantities of materials used for the construction and maintenance of 18 residential buildings across the 100-year lifetime of Meridian One under one of two design scenarios: concrete (gravel and tar) roofs, or extensive sedum roofs.

Materials	Mass (kilotonne, kt)											
	Construction of residential buildings					Construction of concrete roofs	Production of vegetation	Construction of sedum roofs	Maintenance(use phase)		Total	
	Concrete frame	Wall	Floor	Ceiling	Total				Concrete roofs	Sedum roofs	Concrete roofs[a]	Sedum roofs[b]
Aluminum	–	0.6	–	–	0.6	–	–	–	–	–	**0.6**	**0.6**
Asphalt	–	–	–	–	–	0.8	–	–	3.1	–	**3.9**	–
Brick	–	10.8	–	–	10.8	–	–	–	–	–	**10.8**	**10.8**
Brick (crushed)	–	–	–	–	–	–	–	0.1	–	0.1	–	**0.1**
Cement	38.8	0.3	79.3	–	118.4	8.8	–	8.8	–	–	**127.1**	**127.1**
Clay	–	–	–	–	–	–	–	0.1	–	0.1	–	**0.1**
Compost	–	–	–	–	–	–	0.6	0.1	–	0.0	–	**0.7**
Concrete brick	–	9.8	–	–	9.8	–	–	–	–	–	**9.8**	**9.8**
Flashing membrane	–	0.0	–	–	0.0	–	–	–	–	–	**0.0**	**0.0**
Flooring	-	-	122.2	-	122.2	-	-	-	-	-	**122.2**	**122.2**
Glass	–	0.4	–	–	0.4	–	–	–	–	–	**0.4**	**0.4**
Gravel	64.7	–	251.3	–	316.0	32.6	–	4.1	130.5	–	**479.1**	**320.1**
Plasterboard	–	–	–	0.1	0.1	–	–	–	–	–	**0.1**	**0.1**
Polyethylene	–	–	0.1	–	0.1	0.0	–	0.0	0.1	0.0	**0.1**	**0.1**
Polypropylene	–	–	–	–	–	–	–	0.0	–	0.0	–	**0.0**
Polystyrene	–	–	–	–	–	–	–	0.0	–	0.0	–	**0.0**
Rock wool	–	1.0	–	63.6	64.6	0.2	–	0.0	–	0.0	**64.8**	**64.6**
Sand	43.1	1.0	176.8	–	221.0	19.2	–	19.2	–	–	**240.2**	**240.2**
Vegetation	–	–	–	–	–	–	–	0.2	–	0.2	–	**0.3**
Water	27.0	0.3	47.7	–	74.9	5.2	0.9	5.2	–	–	**80.2**	**81.1**

[a]Excludes quantities in "production of vegetation" and "construction of sedum roofs."
[b]Excludes quantities in "construction of concrete roofs."

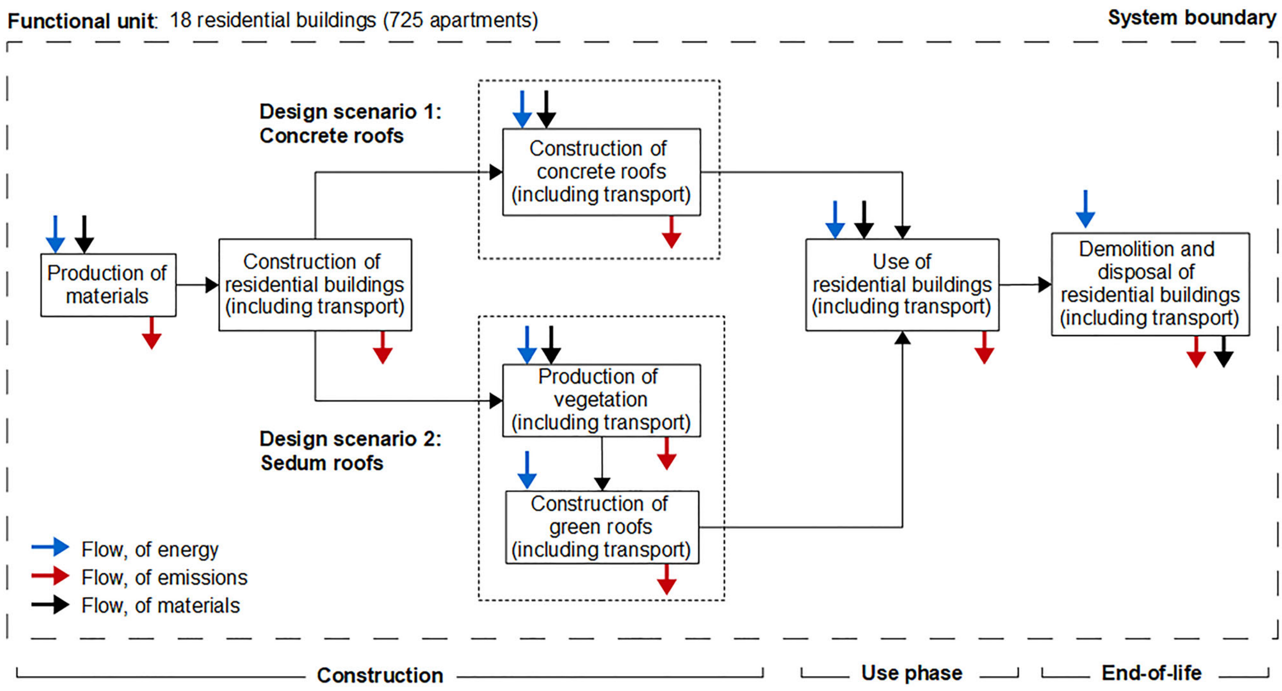

FIGURE 2 Flow diagram describing the processes and flows (thick arrows) involved in the development of Meridian One, from cradle-to-grave, as modeled in OpenLCA v.1.11.0 (with ecoinvent v.3.8). The thick, colored arrows are the flows (energy in blue, emissions in red, and materials in black). The thin, black arrows signify the direction of flow across three life cycle stages: Construction, use, and end-of-life.

Regarding land use, the development's life cycle biodiversity impacts are cumulatively affected by artificial habitats at two elevations: ground and roof levels. Green infrastructures offer potential gains in local species richness depending on their design (i.e., composition) and size (i.e., surface area they occupy). The potential biodiversity impacts of elevated land occupation by sedum roofs are separate and concurrent to the impacts of land occupation at ground level, so they should be inventoried separately (Tables S19, S20 Supporting Information S2). The need to distinguish between ground and other surface coverage when quantifying life cycle biodiversity impacts includes vertical greening (e.g., green walls), where bio-productive (and hence biodiversity-supporting) surface areas are both elevated and perpendicular to land cover.

2.4 | Biodiversity at Meridian One

2.4.1 | Constructing a food web to describe energy flows at Meridian One

There is a lack of data describing biodiversity at the MWD site. The MWD site has also undergone development and hence disturbance preceding the development of Meridian One. For these reasons, and for the purpose of this study, it is assumed that no terrestrial species are actively supported at Meridian One prior to development. In lieu of on-site data, we compile a list of terrestrial species likely to use the site using ecological literature, expert opinion, local priority species designation, and geo-referenced data from the online data repository iNaturalist (Biodiversity Reporting and Information Group [BRIG], 2007; Ealing Council, 2022; iNaturalist, 2022). Our goal here is to develop a reasonable estimate of a local, terrestrial species population, which could make use of the sedum roofs. We aim to represent the case study's location and account for seasonal changes. To ensure the population is representative of the case study's location, only species likely observed at the site (Enfield, London) are included. To ensure temporal consistency, we restrict the population to a fixed period: late spring (April–June). This is when sedums flower and nectar—its limiting resource—is produced, making it the period of maximum resource provision. Here, we assume that the population is static, with no variation in individual species' diets during this period. We acknowledge that the population would demonstrate considerable change over a year, owing to the seasonality of vegetation and prey abundance (e.g., migration, metamorphosis of invertebrates), and reflecting changing dietary needs for gestation, malting, and overwintering (e.g., hibernation). However, it is not possible or practical to capture this variation in our study at this stage. A species list is compiled, and a trophic web illustrated in line with these assumptions (Table A1 and Figure A2, respectively, Supporting Information S1); all underlying assumptions are discussed in Appendix S2.

2.4.2 | Quantifying the biodiversity supported by the sedum roofs

To calculate the species richness supported by green roofs at Meridian One, two things must next be determined: the production of energy by the sedum roofs and the way energy is distributed across the population. Local species richness, quantified here, is a measure of alpha (bio)diversity (Večeřa et al., 2019). Native and non-native species contribute to alpha biodiversity, so we consider both in our paper. This includes invasive non-native species such as the ring-necked parakeet (Heald et al., 2020).

The production of plant biomass and nectar by the green roofs underpins its ability to support biodiversity. Of the animals species expected to use Meridian One: 15 consume plant biomass, and only some invertebrates consume nectar (see Appendix S2). Based on our assumptions (see Appendix S6, Supporting Information S1), the sedum roofs produce 8.6×10^5 kJ plant biomass day^{-1} and 12,458 kJ nectar day^{-1} (Bosch et al., 1997; Lechantre et al., 2021; Rodney & Purdy, 2020).

A bottom-up approach is used to describe the distribution of energy, from the sedum roofs, through the food web. Data on interspecies predation based on observation or from stomach contents, for example, are conventionally used to construct trophic models of this nature (Benke & Huryn, 2017; Benke et al., 2001; Woodward et al., 2005). Without comprehensive data on how species would compete for energy sources a Meridian One, it is assumed that invertebrates make use of energy stocks before higher trophic levels and that each animal has the same opportunity as its competitors to access energy (i.e., energy stocks are distributed equally between consumers). Assumptions underlying this distribution are also presented in Appendix S6, Supporting Information S1.

2.5 | Developing an endpoint characterization factor for extensive green roofs

When calculating biodiversity loss, transformation and occupation of land are causally linked to species loss, based on observed relationships between species richness and habitat size (MacArthur & Wilson, 2001). Our ecological model describes a similar relationship for sedum roofs at Meridian One, facilitating novel characterization based on existing LCIA methods. We demonstrate this using the IMPACT World+ life cycle impact assessment method, as shown here, and discussed further in Appendix S8, Supporting Information S1.

We develop an endpoint characterization factor for urban sedum roofs using the results of our ecological model (life cycle ecological impact in number of species), converting them into the integrated units PDF m^2 year, consistent with the IMPACT World+ framework. Our biodiversity methodology and results can also be applied to calculate CFs consistent with other LCIA methods.

In IMPACT World+, the potential impact of occupying land of type LU$_i$ in ecoregion j on the quality of that ecosystem, $I_{o,LUi,j}$ (PDF m^2 year), is calculated using Equation (1) (Bulle et al., 2019; de Baan et al., 2013):

$$I_{o,LU_{i,j}} = CF_{o,LU_{i,j}} \times A_o \times t_o \tag{1}$$

where CF$_{o,LUi,j}$ (PDF) is the endpoint characterization factor for land occupation of type LU$_i$ in ecoregion j. A_o (m^2) is the area of land being occupied, and t_o (years) is the duration of that occupation. By rearranging Equation (1), the endpoint characterization factor for sedum roofs in this case study can be calculated using a known $I_{o,LUi,j}$ value (Equation 2):

$$CF_{o,LU_{i,j}} = \frac{I_{o,LU_{i,j}}}{A_o \times t_o} \tag{2}$$

3 | RESULTS AND DISCUSSION

3.1 | Biodiversity supported by green roofs at Meridian One

We found that primary production by the green roofs supports 53 terrestrial species comprising 673 individuals (see Table S18, Supporting Information S2). These 53 species and the flow of energy between them, across four trophic levels, is described in Figure 3. While not included in the IMPACT World+ framework, species abundance is a useful additional output of our modeling. It confers information on the population that should inform decision-making in urban design. In this case, the population has a low species diversity (Appendix S9) (Willis & Martin, 2022). This local biodiversity impact—a gain of 53 species—is attributed to the use phase of Meridian One and maintained throughout the site's occupation (see Appendix S8 and Fig. A6, Supporting Information S1). Local biodiversity impacts of the sedum roofs are restricted to the use phase for two reasons: First, species, particularly those in high taxa, are unlikely to be supported until construction is complete. Second, when the sedum roofs are removed at end-of-life, the resources that support terrestrial biodiversity will be lost and offsetting will not continue thereafter.

The reference species richness value used in this study corresponds to the ecoregion "Western European broadleaf forest" as this is the ecoregion (subscript j in Equations 1 and 2) that is being transformed and occupied. The reference species richness for Western European broadleaf forest

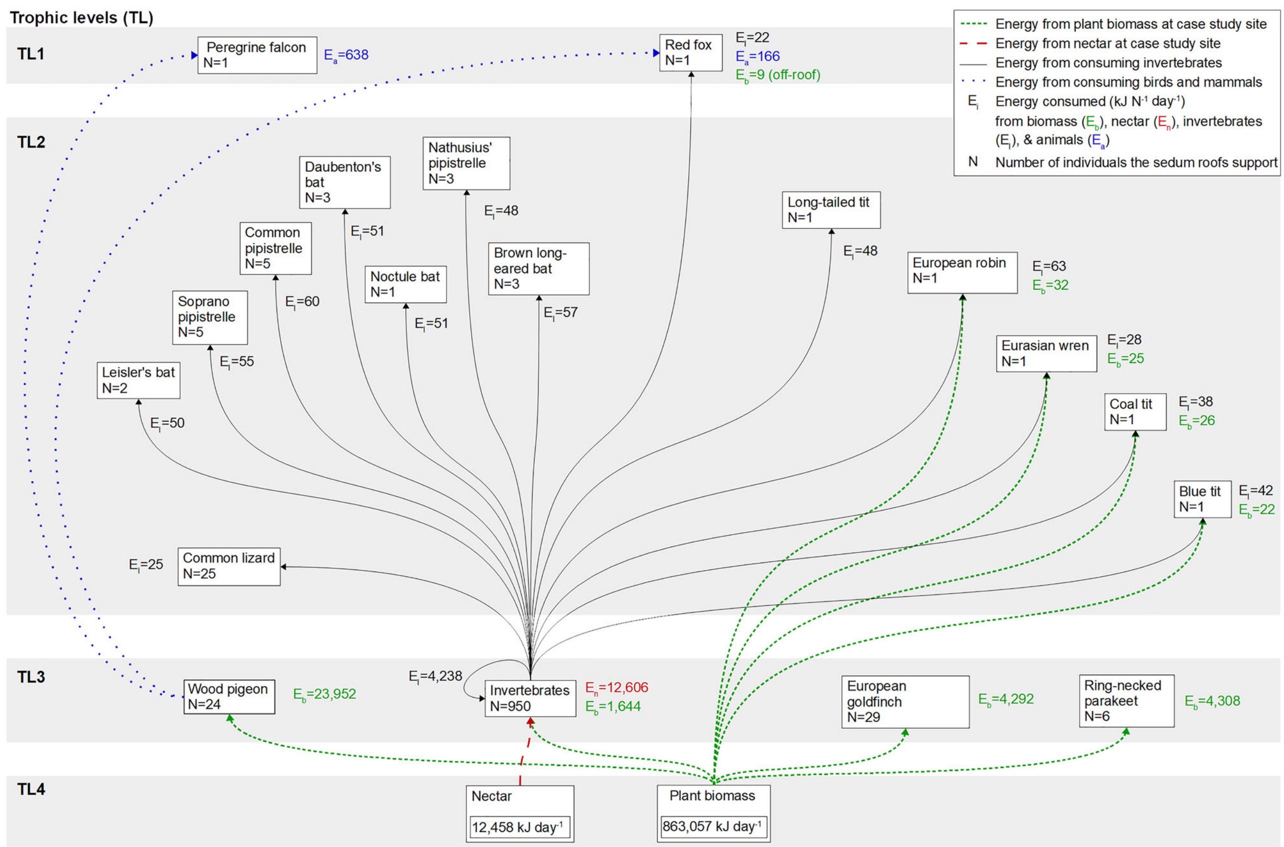

FIGURE 3 Energy flow diagram describing the number of individuals supported by the sedum roof (*N*, individuals) and the interactions taking place between them. The energy consumed per predator-prey interaction per species of population size "*N*" (E_i, [individuals (prey)] [individuals (predator)]$^{-1}$ day^{-1}) are also described. E_n corresponds to the consumption of nectar (dashed red line); E_b corresponds to the consumption of plant biomass (dashed green lines); E_I corresponds to the consumption of invertebrates (solid black lines); and E_a corresponds to the consumption of birds and mammals (dotted blue lines). A key is provided (top right). The gray areas (background) indicate trophic level grouping, numbered (left) from the highest trophic level (TL1) to the lowest (TL4). Data used to produce this figure is found in Tables S13-S14, Supporting Information S2.

that is used in IMPACT World+, as reported by De Baan et al. (2013), is 3279 terrestrial species (Bulle et al., 2019; Lammerant et al., 2019); a value is derived from the WWF WildFinder database and Keir (2005) (Table S50, Supporting Information S2) (Kier et al., 2005; World Wildlife Fund, 2006). The sedum roofs' in situ life cycle ecological impact of −53 species (negative as it describes a gain) is 1.6% of this reference species richness value. The occupation of roof areas (0.018 km^2) as "sedum roof" at Meridian One therefore supports 1.6% of the reference terrestrial species richness over 1.8×10^4 m^2 (total roof area) over 100 years (total development lifetime) (De Baan et al., 2013). This description is then condensed; the units are combined into the form PDF m^2 year, as required: $(0.016) \times (1.8 \times 10^4) \times (100)$. The modeled local terrestrial biodiversity impact is: -2.91×10^4 PDF m^2 year.

Interspecies competition will influence the way energy is distributed within the local species population, but this is difficult to predict owing to a lack of data, making assumptions surrounding species' access to energy inherently speculative. However, given that the green roofs comprise vegetation, and hence produce energy as nectar and plant biomass, it is expected that invertebrates are preferentially supported. Nectivorous invertebrates in particular face less competition for nectar, given that most species in the local population consume other forms of energy. The estimated invertebrate species abundance is higher than reported field data (Appendix S3, Supporting Information S1), with a lower species richness than perhaps might be observed (Jones, 2002). Field studies report only what is observed, whereas our model describes potential species richness based on energetics. In this sense, a larger species richness is expected. The discrepancy in species richness is attributed to limited data relating to uncharismatic species near the case-study site.

3.2 | Combined LCA and biodiversity results

The lifetime impacts of Meridian One on ecosystem quality under both design scenarios are presented in Figure 4. In both scenarios, ex situ impacts dominate. They represent 99% of the net total impact of both scenarios. The overall impact of developing Meridian One with sedum roofs

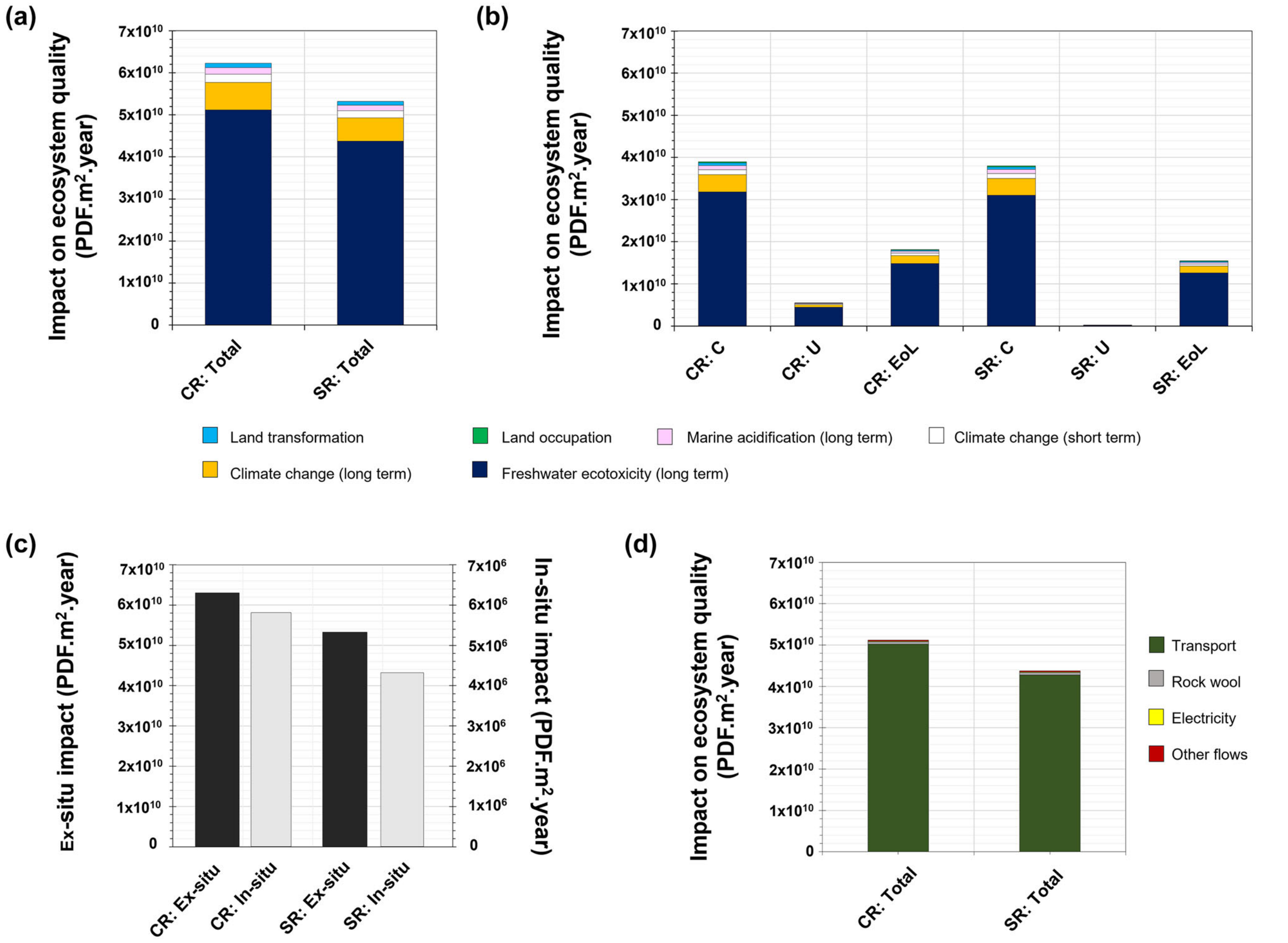

FIGURE 4 Endpoint impacts on ecosystem quality associated with the development of 18 residential buildings at Meridian One with either concrete roofs ("CR") or sedum roofs("SR") over its 100-year lifetime. In (a), the total life cycle impacts of both roof scenarios (concrete and sedum roofs) on ecosystem quality are shown. To improve readability, only the six most important impact indicators are described: "Land transformation" (light blue), "Land occupation" (green), "Marine acidification (long term)" (pink), "Climate change (short term)" (white), "Climate change (long term)" (amber), and "Freshwater ecotoxicity (long term)" (dark blue). In (b), the results are disaggregated by life cycle stage: Construction ("C"), Use ("U"), and End-of-Life ("EoL"). (c) Illustrates the contribution of ex situ (black, primary [left] axis) and in situ (gray, secondary [right] axis) impacts. Note the different scales of the primary ($\times 10^6$) and secondary ($\times 10^{10}$) *y*-axes. (d) Shows the main contributors to total life cycle impacts on ecosystem quality by process. The processes with the three highest contributions are shown: transport (dark green), rock wool production (gray), electricity generation (yellow); other flows are shown as combined (dark red). Positive impacts values represent decreased species richness (number of species); negative values represent increased terrestrial species richness. Data used to produce this figure are presented in Tables S21-S28, Supporting Information S2.

(5.46×10^{10} PDF m^2 year) is 15% less than developing Meridian One with concrete roofs (6.39×10^{10} PDF m^2 year). This result is not unexpected, so we also consider the significance of the sedum roofs and whether developing Meridian One with sedum roofs is more beneficial than leaving the site undeveloped, from a conservation perspective. The sedum roofs support 53 terrestrial species. This offsets 1.3% of the in situ biodiversity loss caused by the development (occupation of the roofs as well as other urban area on-site, e.g., roads and paths: 2.94×10^6 PDF m^2 year). Compared to the development's total (in situ plus ex situ) impact, the biodiversity impact of the sedum roofs is found to be negligible. The 15% decrease observed is instead the result of less impactful material and transport use in the sedum roofs scenario.

The results suggest it is more beneficial, from a conservation (ecosystem quality) perspective, to leave the site undeveloped than either concrete or sedum roof scenarios. If this were the case, the site would undergo natural relaxation toward its reference state (De Baan et al., 2013). Developing the site prevents this, delaying relaxation by 100 years. Occupying the roof areas as sedum roofs, while the rest of the site is "urban, continuously built," is insufficient to offset the in situ loss of terrestrial biodiversity caused by the site's development. The impact in the use phase (SR:U) is positive (Figure 4).

In all cases, the greatest contributor to ex situ impacts at Meridian One is the endpoint impact category "freshwater ecotoxicity, long term" (Figure 4). Freshwater ecotoxicity describes biodiversity loss caused by the pollution and subsequent degradation of aquatic ecosystems (Owsianiak et al., 2023), often resulting from stressors like pesticides and metals entering freshwater ecosystems (Gandhi & Diamond, 2018; Schuijt et al., 2021). Over the lifetime of Meridian One, 98% of freshwater ecotoxicity is attributed to transport, predominantly via the emission of metals (including aluminum, copper, and iron) to water (Fig. A4, Supporting Information). Transport contributes most to freshwater ecotoxicity in all life cycle stages except the use phase in the green roofs scenario (Fig. A4g, Supporting Information). There, 92% of freshwater ecotoxicity is attributed to electricity, owing to vehicle use in the green roofs use phase (2.04×10^7 t km) being two orders of magnitude less than in the concrete roofs use phase (6.68×10^9 t km).

The fact that most of the development's life cycle impacts relate to freshwater ecotoxicity raises concerns surrounding ecological equivalence in biodiversity offsetting (Kate et al., 2004; Pope et al., 2021). To achieve net-zero biodiversity loss, biodiversity gains must not only be at least equal to biodiversity losses, but like-for-like. This means that while the green roofs offer terrestrial biodiversity offsets, offsetting the life cycle ecological impact of freshwater ecotoxicity requires freshwater biodiversity offsets. By broadening the scope of this work, to include the indirect effect of resource provision on site to freshwater ecosystems, the effect of the sedum roofs on freshwater biodiversity could also be modeled. However, this is beyond the research scope at this stage. It is also worth noting that theoretical reference states would comprise native species. While any change in species richness constitutes a biodiversity impact, the support of non-native species—the ring-necked parakeet in this case—would not count toward the restoration of reference-state conditions.

3.3 | Sensitivity analysis

With transport responsible for most endpoint impacts associated with the development of Meridian One, the site's proximity to distributors and disposal sites significantly influences the ex situ impacts. For this reason, we perform a sensitivity analysis on transport in both design scenarios (concrete and sedum roofs) varying two parameters: transport distance and vehicle type (Fig. A4). We model a ±25% change in transportation distances, and the use of either "7.5–16 tonne" or ">32 tonne" lorries.

3.3.1 | Effects of transportation distances and vehicle type

In the concrete roofs scenario, the life cycle impact (PDF m^2 year) demonstrates a 22% increase with a 25% increase in transport distances and a 22% decrease with a 25% reduction in transport distances. The same responses are seen for life cycle freshwater ecotoxicity (PDF m^2 year), with a 22% increase with a 25% increase in transport distances and a 22% decrease with a 25% reduction in transport distances. More favorable responses were seen in the sedum roofs scenario: the life cycle ecological impact shows a 6% increase at +25% transport distances and a 35% reduction with a 25% reduction in transportation distances. Again, the same responses are seen for life cycle freshwater ecotoxicity (PDF m^2 year): a 6% increase with a 25% increase in transport distances and a 35% decrease with a 25% reduction in transport distances.

By using larger vehicles with larger load capacities, fewer vehicles and associated infrastructures are needed over the lifetime of Meridian One. Conversely, the use of smaller vehicles can require more resources. The benefit of larger load capacities is found to be more pronounced in the sedum roof scenario, with a smaller increase (less than half) incurred when the smaller vehicle type (7.5–16 tonne lorry) is used. In the concrete roofs scenario, the use of 7–16 tonne vehicles increases the life cycle ecological impact and life cycle freshwater ecotoxicity by 38%. The use of >32 tonne vehicles decreases the life cycle impact on ecosystem quality by 42% and life cycle freshwater ecotoxicity by 55%. In the sedum roofs scenario, the use of 7–16 tonne vehicles increases the life cycle impact on ecosystem quality and life cycle freshwater ecotoxicity by 18%. The use of >32 tonne vehicles decreases the life cycle impact on ecosystem quality by 50% and life cycle freshwater ecotoxicity by 62%.

A combined sensitivity analysis is performed based on these findings (Figure 5). The best-case being: >32 tonne vehicles and −25% transportation distance. This produces a 55% and 62% reduction in the overall (life cycle) impact on ecosystem quality for the concrete and sedum roofs scenarios, respectively, and a 99.9% reduction in life cycle freshwater ecotoxicity in both cases.

3.3.2 | Practical limit of offsetting terrestrial biodiversity loss in this case study

For the sedum roofs to completely offset ex situ terrestrial biodiversity loss, transportation impacts must be reduced by at least one order of magnitude. Using the ">32 tonne" vehicle option, we find that ex situ impacts are not offset until transportation distances are reduced to 8% their original values: 4 km for material supply (4.8 km for vegetation), and 1.7 km for disposal. These distances are not feasible; material supply and disposal requirements cannot be satisfied so close to the London site. It is therefore not possible to completely offset the terrestrial biodiversity loss incurred when developing Meridian One through the proposed use of sedum roofs.

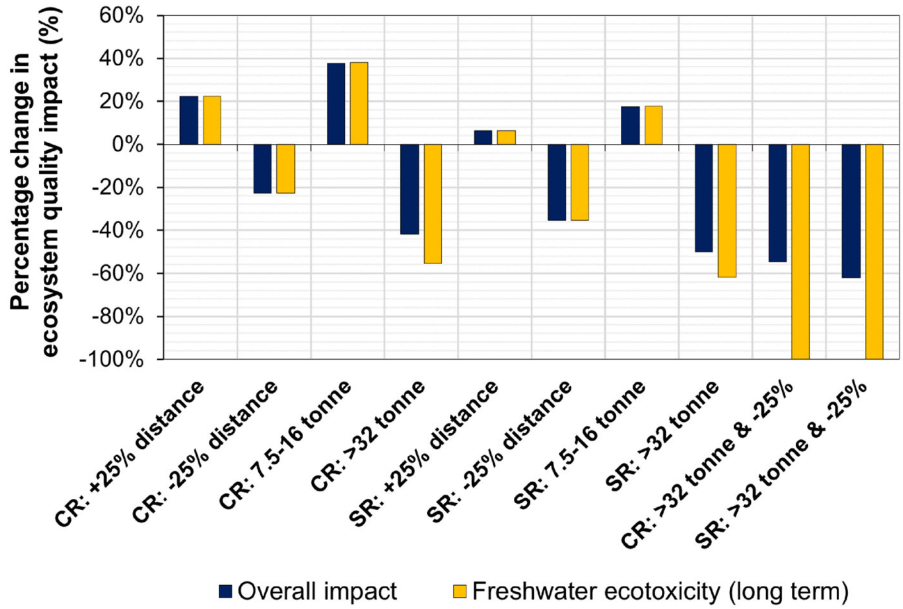

FIGURE 5 Sensitivity analysis of life cycle impacts on ecosystem quality when developing Meridian One with concrete roofs (CR) or sedum roofs (SR), varying the transportation distance (±25%) and vehicle type, 7.5–16 tonne and >32 tonne. A combined, best-case scenario is also presented: >32 tonne vehicle and −25% distance. Results are presented in terms of percentage change (%) across two impact categories: "Overall impact" (dark blue) and "Freshwater ecotoxicity, long term," the greatest contributor to the overall impact of the case study on ecosystem quality. Data used to produce this figure are presented in Tables S37-S44, Supporting Information S2.

3.4 | Describing an endpoint characterization factor

Using IMPACT World+ methodology (Section 2.5), we calculate the relative impact of land occupation on ecosystem quality, I_o, (PDF m^2.year) for the sedum roofs in this case study to be -2.91×10^4 PDF m^2.year. Therefore, using Equation (2) (Section 2.5), the endpoint characterization factor of occupation for the case study, which we term "urban, sedum roof, London," is -1.6×10^{-2} PDF.

3.5 | Perspectives

We applied our methodology, to quantify the local ecological impact of land occupation (Section 2.4), to an urban sedum roof in London. However, the methodology is based on energetics and hence is not restricted to a particular location, scale, or infrastructure type. This gives it excellent potential for further application, notably by offering a way to improve how terrestrial biodiversity change associated with green (or any other bio-productive) infrastructures are quantified in LCA. Within urban design, this advancement can further our understanding of how urban and peri-urban areas can support natural needs alongside those of humans. With a better understanding of the role green infrastructures may play in creating sustainable, multifunctional urban ecosystems, more robust guidance on can be produced to guide their design (United Nations Development Programme, 2016). For example: ensuring sufficient and suitable (type and quality) resources are available to support the animals urban areas are designed to support.

More ecological data is needed to apply this methodology across different habitat types more comprehensively at the ecoregion level. There is need for more ecological data: geo-referenced data describing local species populations, location, and season-specific data on the provision capabilities of plant species, and data on the assimilation efficiencies of different species. Without these data, assumptions must be made when modeling the local ecosystem, which imparts uncertainty and represents an important limitation of the proposed methodology at present. Contingent on sufficient data and with ongoing application, new characterization factors can be developed describing the potential impact of different biodiversity-supporting infrastructure (green, blue, or otherwise) on terrestrial biodiversity across any temporal–spatial condition. This will address an important limitation in how biodiversity is included in LCA, enabling more robust predictions of biodiversity loss (or gain) and more granular application of LCA without the need to model the local ecosystem in each case.

While the methodology can be applied in any circumstance, the results are restricted to equivalent spatial–temporal conditions; here: urban area, temperate climate, April–June. Achieving high granularity in LCA application therefore requires considerable work to model and quantify different land-use types, globally. As energy flows within the local ecosystem must be modeled explicitly in the first instance, characterization will likely be restricted to small spatial scales (i.e., meso-scale: community level within a forest, city, or lake; Jordán et al., 2019; Simmons et al., 2021) and limited gena until ecosystem dynamics are quantitatively described at the ecoregion level. While it is conceptually possible to develop these models under any temporal–spatial conditions, on regional or global scales using our proposed methodology, insufficient species data, pertaining to their metabolism of natural resources and inter/intraspecies interactions at the larger scale makes doing so unfeasible at present.

Attention must be paid to temporal and spatial conditions when applying the CF developed in this paper. However, some generalization is possible. First, we consider the temporal aspect. Our CF relates to the period April–June where provision is amongst its highest because sedum is flowering. Many flower-feeding invertebrates feed predominantly during April–June, entering diapause for the rest of the year, therefore green roofs provide resources in a critical part of the year. Applying this CF to winter periods, however, may produce an overestimate of terrestrial biodiversity impacts, as outside the flowering period sedum roofs will be restricted to supporting herbivory. Next, we consider the spatial aspect. Species richness is related to energy provision and demonstrates distinct, lateral boundaries at the global scale (Hillebrand, 2004). The CF developed here is applicable in comparable urban contexts, meaning with similar temperate climate and within the region where the European broadleaf forest is the reference habitat. This includes Dublin, Amsterdam, and Paris, since temperate climates are common across Northwestern Europe (Kottek et al., 2006). However, this is contingent on local species pools being comparable to that in this case study. If dissimilar, the local biodiversity impact should be modeled anew in each case. Consequently, the CF developed is unsuited to application in tropical climates where species richness is greater than temperate climates. A potentially greater terrestrial biodiversity impact could be seen where biome-equivalent green roof plants are used.

Next, there is an implication of using theoretical rather than contemporary reference data for potential species richness at the case-study site. The theoretical reference species richness value used (Western European broadleaf forest) assumes biodiversity potential on-site that is substantially greater than that which is typical in urban areas (De Baan et al., 2013). Unlike forests, urban areas are typically fragmented, and subject to high levels of disturbance, which limits access for many species. The potential biodiversity impact of the sedum roofs is calculated to be 1.6% of the reference species richness value. Prior to development, the site was urban brownfield, a land cover type which typically has much lower species richness than broadleaf forest (De Baan et al., 2013). Thus, a greater relative biodiversity impact would be experienced if a contemporary reference value was used. As such, supporting 1.6% broadleaf forest species richness constitutes conservative estimate of the potential impact for the site.

Finally, aggregating species data across an entire year would produce characterization factors that are less temporally restricted. In this study, a fixed period in spring was used. This corresponded to the assumed period of maximum provision and hence, the sedum roofs' maximum potential biodiversity impact. However, this case study represents an idealized scenario. Urban areas typically comprise many, heterogeneous habitats with varied and less temporally restricted provision. Going forward, the ideal would be to aggregate species data cross an entire year, in a way that captures seasonality in provision, species metabolism (e.g., for malting and hibernation), and the temporary absence or presence of migratory species.

4 | CONCLUSION

In this paper, we developed a methodology to quantify what terrestrial biodiversity can be supported using urban green infrastructure within the built environment. We demonstrated our methodology in a case study of the MWD in London, UK. To achieve this, we compared the life cycle impact of the development on ecosystem quality with either concrete roofs or sedum roofs through LCA. We quantified the in situ biodiversity impact achieved in the green roofs scenario by modeling energy flows across a local species population, and then integrated this biodiversity gain into the LCA result in a novel way. In the case study presented, the sedum roofs support 53 terrestrial species (673 individuals), equivalent to an endpoint impact of -2.91×10^4 PDF m^2.year. This is equal to 1.3% of the in situ impacts associated with Meridian One's development. However, its impact on the development's total life cycle impact was found to be negligible. The benefits of more richly planted green roofs than those considered here (sedum based) would be similarly limited without a more diverse local species population. Our study thus shows that sedum roofs may only serve as minor ecological impact mitigation measures in the context of urban development, with terrestrial biodiversity offsets confined to the use phase. More substantial life cycle impact reduction can be achieved by decreasing the amount of transport used throughout the development's lifetime and reducing energy consumption in the use phase. This paper demonstrates a method to quantify the local terrestrial biodiversity impact associated with green infrastructures in a way that integrates into the existing LCA framework, which can improve both (i) understanding of how ecological needs can be supported alongside those of humans in urban areas, and (ii) our ability to design green infrastructures that can offset the impacts of urban development on ecosystem quality.

AUTHOR CONTRIBUTIONS

Adam R. Mason: Conceptualization; methodology; formal analysis; investigation; resources; data curation; writing—original draft; writing—review and editing; visualization. **Pepe Puchol-Salort**: Conceptualization; resources; data curation; writing—review and editing; visualization. **Alfred Gathorne-Hardy**: Conceptualization; writing—review and editing; supervision. **Barbara Maria Smith**: Conceptualization; writing—review and editing. **Rupert J. Myers**: Conceptualization; writing—review and editing; supervision.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

The data that supports the findings of this study are available in the supporting information of this article.

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SUPPORTING INFORMATION

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