Quantifying the natural resource requirements of terrestrial ecosystems for managing species diversity alongside human development

by

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Abstract

Socioeconomic demand for natural resources (e.g., land, water, sunlight) is causing considerable losses of biodiversity and ecosystem functionality, most notably in regions where socioeconomic (i.e., human) systems and ecosystems compete for the finite resources. However, a poor quantitative understanding of the natural capital requirements required to support biodiversity, while at the same time satisfy socioeconomic (i.e., human) needs, undermines our capacity to secure sustainable human development. It is not yet clear what resources (area, food, water, infrastructure for nesting…) must be provided to support biodiversity alongside humans under different contexts, nor what impacts different interventions may have on terrestrial species diversity.

This thesis represents the first step what is envisaged to be a considerable body of work, contributing to ecological understanding and LCA method, and of multidisciplinary value. A set of equations for estimating the land use, energy requirements, and rates of water consumption of terrestrial animals are presented. These are the first quantitative set of ecological needs for terrestrial mammals, birds, reptiles, and invertebrates, analogous to human wellbeing needs, which represent common physiological requirements of most organisms that underpin ecosystem functionality and sustainable human development.

This work culminates in a case study of the life cycle impacts of a residential development in London, UK. The local trophic system is empirically modelled, the resource requirements of its constituent species are quantified, and the potential life cycle impacts of including green infrastructure on-site on local species diversity and species abundance are calculated. The causal relationship between land cover type and local terrestrial species diversity is also demonstrated. Applied in the urban context here, the works in this thesis are applicable to any 'biodiversity-supporting components': ecological (e.g., soil, water) or man-made (e.g., shelter: nest boxes) elements that can support the physiological (or social) requirements of organisms and hence ecological systems.

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Declaration

I declare that this thesis was composed by myself, that the work contained herein is my own except where explicitly stated otherwise in the text, and that this work has not been submitted for any other degree or processional qualification except as specified.

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Preface

The work in this thesis constitutes the first step in understanding how natural resources are used by ecological systems concurrent to their use by humans within socioeconomic systems. The thesis represents a bringing together of ecological and engineering understanding and practices, the breadth of which are demonstrated in the literature review (Chapter 2), which motivates the thesis and research questions. The thesis comprises three main parts: the development of a means of quantifying the natural resource requirements of terrestrial animals (Chapter 3), the development of a method to characterise local biodiversity impacts of land use change on terrestrial biodiversity, for inclusion within life cycle assessment (LCA) (Chapter 4); and the development of an improved method for explicitly modelling local ecological systems and quantifying the natural resource requirements and population dynamics (species richness, species abundance) at the individual, species, and system level. Fig. P.1 illustrates the thesis structure and the relationships between its chapters.

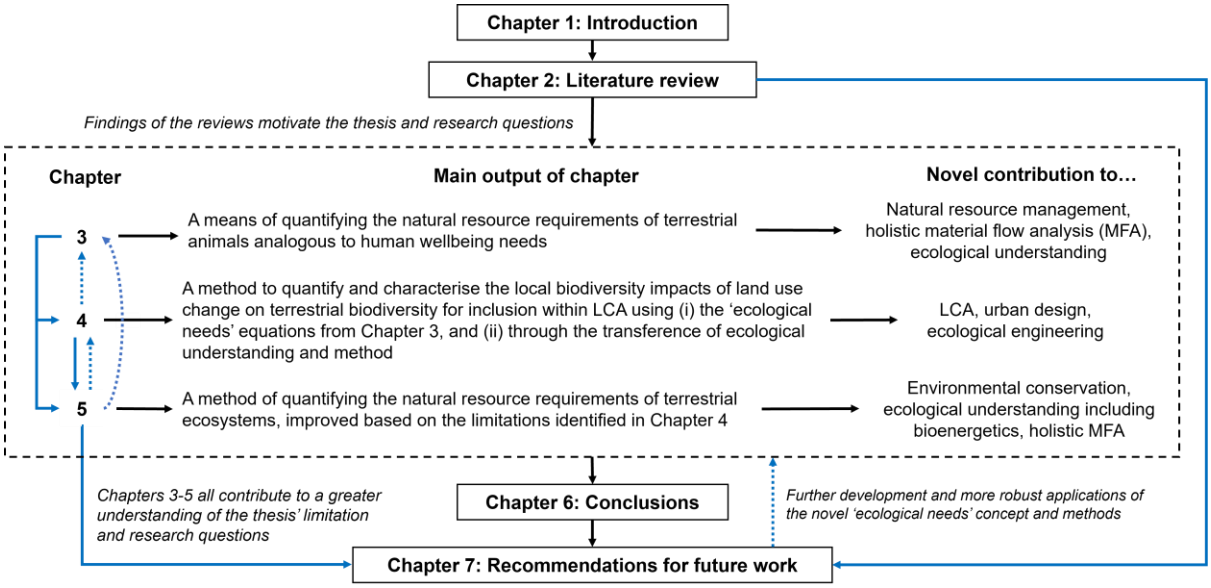


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This thesis represents the first steps of a considerable undertaking. As such, an important aspect is identifying current limitations in quantifying ecological systems and supporting biodiversity alongside human development, including through urban design, and means of improving our current capacity to do both. The multidisciplinary value of this research is demonstrated in this thesis. It is hoped that readers will also be inspired to consider the needs of ecological systems, concurrent to humans, wherever possible; and if a means is not yet described, to take the first step.

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Nomenclature

Glossary of symbols

$_i$	Denotes ‘of one individual of a species’
$_{p,i}$	Denotes ‘of one population of species *i*’
$_s$	Denotes ‘of one ecological system’
$_j$	Denotes the measure relates to a consumed species (i.e., prey) where *i* is a species acting as a consumer (i.e., predator).
L_i	Land use requirement of an individual of species *i* (km^2 $individual^{-1}$)
L_j	Land use requirement of an individual of a species as a consumer (km^2 $individual^{-1}$)
L_{BP}	Bio-productive land area (km^2)
L	Unit: Litres
L	Used to represent ‘land use requirement’ in a general sense (km^2). Distinguished from the unit ‘L, litres’ using italics.
$_{LUa,b}$	The subscript $LC_{a,b}$ denotes land cover of type *a*, in location (e.g., biome) *b*
ρ_i	Population density of an individual of species *i* (individuals km^{-2})
W	Used to represent ‘water consumption’ in a general sense (m^3 day^{-1})
W_i	Water consumption of an individual of species *i* over time (m^3 $individual^{-1}$ $[time]^{-1}$)
$W_{i,mammals}$	Water consumption of an individual species *i* of the biological class ‘mammals’ over time (m^3 $individual^{-1}$ $[time]^{-1}$)
$W_{i,birds}$	Water consumption of an individual species *i* of the biological class ‘birds’ over time (m^3 $individual^{-1}$ $[time]^{-1}$)
$W_{p,i}$	Water consumption of a population of species *i* over time (m^3 $population^{-1}$ $[time]^{-1}$)
W_s	Water consumption of an ecological system over time (m^3 $system^{-1}$ $[time]^{-1}$)
E	Used to represent ‘energy requirements’ in a general sense
E_i	Energy requirement of an individual of species *i* over time (kJ $individual^{-1}$ $[time]^{-1}$)
$E_{i,mammals,herbivore}$	Energy requirement of an individual herbivorous species *i* of the biological class ‘mammals’ over time (kJ $individual^{-1}$ $[time]^{-1}$)
$E_{i,mammals,omnivore}$	Energy requirement of an individual omnivorous species *i* of the biological class ‘mammals’ over time (kJ $individual^{-1}$ $[time]^{-1}$)
$E_{i,mammals,carnivore}$	Energy requirement of an individual carnivorous species *i* of the biological class ‘mammals’ over time (kJ $individual^{-1}$ $[time]^{-1}$)
$E_{i,birds,herbivore}$	Energy requirement of an individual herbivorous species *i* of the biological class ‘birds’ over time (kJ $individual^{-1}$ $[time]^{-1}$)

Glossary of symbols continued

$E_{i,birds,omnivore}$	Energy requirement of an individual omnivorous species *i* of the biological class 'birds' over time (kJ individual^{-1} [time]$^{-1}$)
$E_{i,birds,carnivore}$	Energy requirement of an individual carnivorous species *i* of the biological class 'birds' over time (kJ individual^{-1} [time]$^{-1}$)
$E_{i,reptiles}$	Energy requirement of an individual species *i* of the biological class 'reptiles' over time (kJ individual^{-1} [time]$^{-1}$)
$E_{i,invertebrates}$	Energy requirement of an individual species *i* of the biological class 'invertebrates over time (kJ individual^{-1} [time]$^{-1}$)
$E_{i,j}$	Portion of an individual consumer's energy requirement (i.e., E_i) sourced through the consumption of consumed species *j*
$E_{p,i}$	Energy requirement of a population of species *i* over time (kJ population^{-1} [time]$^{-1}$)
$E_{p,i,veg}$	Energy requirement from vegetation of a population of species *i* over time (kJ population^{-1} [time]$^{-1}$)
$E_{p,i,nectar}$	Energy requirement from nectar of a population of species *i* over time (kJ population^{-1} [time]$^{-1}$)
$E_{p,i,inv}$	Energy requirement from invertebrates of a population of species *i* over time (kJ population^{-1} [time]$^{-1}$)
$E_{p,i,meat}$	Energy requirement from 'meat' (mammals and birds) of a population of species *i* over time (kJ population^{-1} [time]$^{-1}$)
E_s	Energy requirement of an ecological system over time (kJ system^{-1} [time]$^{-1}$)
$E_{c,s}$	Energy requirement of an ecological system over time as 'consumed', i.e., accounting for $E_{var,i}$, (kJ system^{-1} [time]$^{-1}$)
$E_{veg,s}$	Energy requirement of an ecological system from vegetation over time (kJ system^{-1} [time]$^{-1}$)
$E_{veg,c,s}$	Energy requirement of an ecological system from vegetation over time as 'consumed', i.e., accounting for $E_{var,i}$, (kJ system^{-1} [time]$^{-1}$)
$E_{c,i}$	Energy consumed by an individual of consumer species *j* over time (kJ individual^{-1} [time]$^{-1}$)
$E_{c,i,j}$	Portion of an individual consumer's 'consumed energy' (i.e., $E_{c,i}$) sourced through the consumption of consumed species *j*
$E_{c,p,i}$	Energy consumed by a population of consumer species *i* over time (kJ population^{-1} [time]$^{-1}$)
$E_{c,s}$	Energy consumed by all consumer species, system-wide, over time (kJ system^{-1} [time]$^{-1}$)
$E_{p,veg}$	The total energy use of individual herbivores and omnivores satisfied through the consumption of vegetation, i.e., primary producer species (kJ population^{-1} day^{-1})
$E_{var,i}$	Seasonal variations in species' metabolic rates, e.g., for moulting (of feathers) and for breeding (%)

Glossary of symbols continued

η_i	*Eta* – metabolic/assimilation efficiency of an individual of a species (%), i.e., conversion of consumed energy into usable energy
M_i	Adult body mass of an individual of a species (kg individual^{-1})
$M_{c,j}$	Adult body mass of an individual as the consumer species j (kg individual^{-1})
c_j	Average calorific value of an individual of a consumed species, j (kJ kg^{-1})
c_{veg}	Average calorific value for vegetation (kJ kg^{-1})
N_i	Species abundance, i.e., number of individuals, of species i (individuals)
$N_{p,i}$	Species abundance of a population of species i (individuals)
N_s	Species abundance of an ecological system (individuals)
$N_{e,j}$	'Energy as species richness (number of individuals) of consumed species j equivalent', i.e., the flow of energy from the consumed species, j, to the consumer species, i, as 'individuals' based on the calorific value of that consumed species: c_j (kJ individual^{-1})
n	Number of observations or data points
Δ	*Delta* – denotes a change in the variable that precedes it. E.g., ΔN is change in species abundance (individuals)
A	Area (e.g., km^2) – variations denoted by subscripts include
$A_{o,LU,a,b}$	Area of land occupied as land use type a in location b (km^2)
$A_{t,LU,a,b}$	Area of land transformed as land use type a in location b (km^2)
S	Species richness (number of species)
S_{rel}	Relative species richness
$S_{rel,R,b}$	Relative (to species richness in a defined reference state at location b) species richness. Subscript R denotes 'reference state'
$S_{rel,LU,a,b}$	Relative (to species richness with land use type a at location b) species richness
S_{ES} , S_{SE}	Ecological system and socioeconomic system, respectively (in Fig. 4.1)
$I_{occupation,LU,a,b}$	Potential impact of land occupation as land use type a in location (e.g., biome) b (PDF.m^2.year)
$I_{transformation,LU,a,b}$	Potential impact of land transformation as land use type a in location b (PDF.m^2.year)
t	Unit: time
$t_{occupation,LU,a,b}$	The duration of occupation
$t_{relax,LU,a,b}$	The time required for relaxation from the transformed land cover to that of the site's reference state (years)
T	The process of land transformation
O	The process of land occupation
$CF_{occupation,LU,a,b}$	Characterisation factor for land occupation as land use type a in location b
$CF_{transformation,LU,a,b}$	Characterisation factor for land transformation to/from land use type a in location b

Glossary of symbols continued

$CF_{occupation,\ urban,\ sedum\ roof,\ London\ (UK)}$	Characterisation factor for occupation of urban area in London, UK, as sedum roof (as described in 'Section 5.3.4', Chapter 5)
$CF_{endpoint}$	Abbreviation: endpoint characterisation factor
S_{ES}	Ecological system (used in Fig. 4.1(a))
S_{SE}	Socioeconomic system (used in Fig. 4.1(a))
S_1, S_2, S_3, S_4	Scenarios 1-4 (used in Fig. 4.1(b))
NPP	Abbreviation: net primary productivity (kJ km^{-2} day^{-1})
α_{veg}	A coefficient that accounts for the suitability of vegetation comprising NPP for context-specific – depending on climate, location, ecological system composition, species' diets – herbivory. For example, if 10% of the NPP is used for herbivory (i.e., it is suitable and accessible), $\alpha_{veg} = 0.1$.
e	Root mean squared error
R^2	R^2 value in linear regression
a_i	Intercept of the line of best fit determined through linear regression analysis
b_i	Gradient of the line of best fit determined through linear regression analysis
$e_{a,i}$	Standard error term for a_i
$e_{b,i}$	Standard error term for b_i
p_j	Consumable portion of a prey species' body mass ([kg 'meat'] [kg prey]$^{-1}$)
R_E	Rate of (local) energy production over a defined area (kJ [area]$^{-1}$ day^{-1})
R	The process of 'natural relaxation' or 'ecological succession'
Q	Potential ecosystem quality – various subscripts indicate the land use type: including Q_0, Q_d, Q_1, Q_R, $Q_{reference}$, $Q_{industrial}$, $Q_{sedum\ roof}$, and Q_{urban} which are defined in each case.
kg	Unit: kilogram
m^3	Unit: cubic metres
t.kg	Unit: tonne-kilogram
PDF.m^2.year	Unit: Potential disappeared fraction (PDF)-square metre-year

Glossary of terms

LCA	Life cycle assessment
LCIA	Life cycle impact assessment
MFA	Material flow analysis
CF	Characterisation factors
NbS	Nature-based solutions
SAR	Species-area relationship
S1, S2, S3, S4…	Scenario one, scenario two, scenario three, scenario four…

1 Introduction

The field of industrial ecology was developed to understand, analyse, and assess the environmental impacts of socioeconomic (human) actions [1]. It draws parallels between socioeconomic systems and ecological systems [2] and argues that, given their inherent interdependence, the two must be studied simultaneously [1], [3]. This is particularly important when considering the provision and use of natural resource (stocks of natural resources such as fossil fuels, timber, and minerals [4], [5], [6]), which act as a key interface between socioeconomic and ecological systems. Within industrial ecology, natural resource use is empirically modelled to systematically and quantitatively analyse the environmental impacts associated with human development [7], and what effects human development will have on the quality and availability of natural resource in the future [8], [9].

The IPAT equation (Eq. (1.1)) is a key principle in industrial ecology. It describes the environmental impact (I) of human activity as the combined effect of population (P), the impacts related to an increasing global human population (e.g., the more populous an area, the more resources are used [10]); affluence (A), the average per-capita resource consumption of the population (this gives further insight into resource use); and technology (T), the intensity of the processes associated with populations' needs. With a rising global population and increasing individual affluence, particularly in the Global North, and a societal dependence on contemporary technologies, our capacity to minimise the environmental impacts of human activities falls on technological improvements (i.e., reducing T) [1]. For example, through the adoption of less environmentally impactful industrial practices [4], [5], [6].

$$I = P \times A \times T \tag{1.1}$$

Empirical models are used to identify where technological improvements can and ought to be made. Two prominent examples are material flow analysis (MFA) and life cycle assessment (LCA). MFA is used to describe resource use systematically, as 'stocks' (reserves of resources) and 'flows' (the mass of the resource at a given time), within defined spatial and temporary boundaries [11]. MFAs are usually detailed and quantitative to facilitate analysis. Best practice is to describe a whole system and all flows therein (e.g., physical state), quantifying at least all significant flows [12]. Outputs are usually both numerical and figurative, as shown in Fig. 1.1, and accompanied by a discussion of the system and data. MFAs are either static or dynamic. Static MFAs describing a process under a single condition, like Fig. 1.1, whereas dynamic MFAs employ data on historic and projected flows to facilitate long-term (often scenario-based) analyses (e.g., [13]).

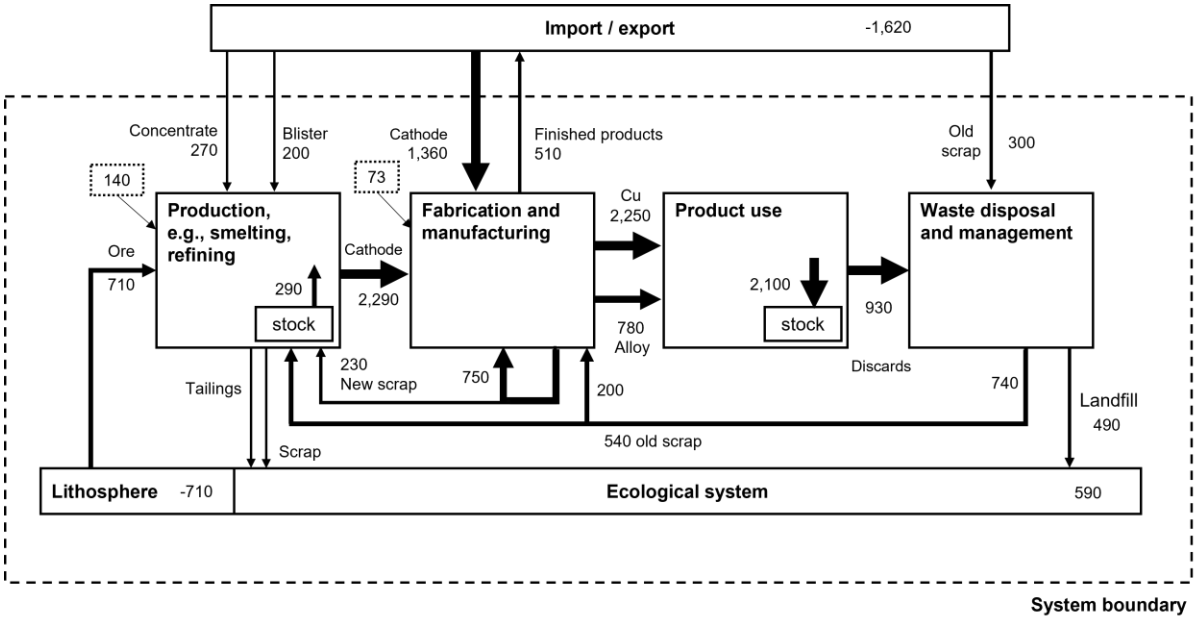


Figure 1.1. Detailed (static) material flow analysis (MFA) for gigagrams of copper in Europe, 1994. Figure adapted from Graedel (2019) [12].

While MFA allow resource use within a system to be scrutinised, it cannot be used to assess its (e.g., environmental) impacts. This requires the use of a secondary assessment method, such as LCA. LCA is a rigorous [14] method for quantifying the impacts of human activities on three 'areas of protection': human health, ecosystem quality, and natural resources (or equivalent); the commodities society seeks to protect and nurture [15]. As with MFA, LCA is applied within defined spatial and temporary boundaries. It has four stages: (1) goal and scope definition, where the parameters of the assessment are set; (2) life cycle inventory (LCI), where data on the processes within the defined parameters are compiled; (3) life cycle impact assessment (LCIA), where the potential impacts of those processes are calculated; and (4) interpretation, where the LCIA results are contextualised. The general method is described in Fig. 1.2.

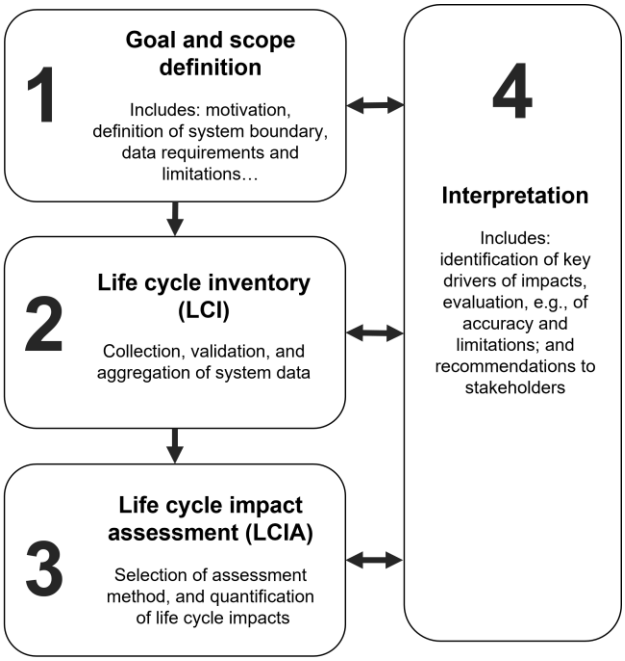


Figure 1.2 The four general stages of life cycle assessment (LCA) as described in the ISO 14040 standard [14]. A summary of each stage is provided. Detailed descriptions are provided in Chapter 2.

In the LCIA stage, activities (e.g., the emission of CO_2) are causally linked to a measure of damage (e.g., climate change) through defined impact pathways. The topic of LCIA method is substantial and beyond the scope of the introduction – the reader is referred to the detailed ISO standards: [14]. Together, MFA and LCA are two of the most important tools in an industrial ecologist's toolkit, but a critical deficiency undermines their application to model natural resource use and assess its impacts: they do not yet capture the natural resource demands of species or populations of species within ecological systems. That is, species other than humans.

1.1 Wellbeing needs as drivers of natural resource use

At the most fundamental level, human development requires the use of natural resource to satisfy 'human wellbeing needs'. These are a set of essential (e.g., shelter, food, and water) and non-essential (e.g., access to technology) social, economic, and physiological requirements for good physical and mental health [16], [17]. A robust, quantitative understanding of natural resource use in socioeconomic systems allows it to be modelled empirically [18]. The associated environmental impacts can then be assessed [19]. Within the last two decades, a quantitative set of human wellbeing needs has been developed, which describe the resource use needed to achieve a decent standard of living in humans [20], [21]. This advancement has made it possible to infer the minimum level of natural resource use needed to support socioeconomic systems [22], and assess the environmental impacts of human

development more accurately. Importantly, this permits greater scrutiny of contemporary human development targets – the most prominent example being the Sustainable Development Goals (SDGs) [23] – and our capacity to achieve them.

All living organisms (plants, animals, fungi, etc.) demonstrate physiological needs. Like humans, organisms in ecological systems require water, energy, and nutrients. In some cases, namely plant and animal species, they also demonstrate social needs [24], [25]. Intuitively, these requirements constitute the 'wellbeing needs' of organisms comprising ecological systems, analogous to human wellbeing needs. However, quantitative ecological wellbeing needs are not yet established, prior to this thesis. A method to quantify ecological wellbeing needs has now been described and applied (see Chapter 2 [26] and Chapter 3 [27]). Given the concept's immaturity, it remains unclear what type, quantity, or quality of natural resources are needed to support individual species, and by extension populations of species, within ecological systems; nor is it known what natural resource must be allocated to ecological systems, concurrent to their use in socioeconomic systems, to stimulate and sustain specific ecosystem functions on which contemporary socioeconomic systems depend [28], [29]. The most relevant example here is the provision of natural resources.

The lack of quantitative ecological wellbeing needs undermines our capacity to manage natural resources in a sustainable way. Where socioeconomic and ecological systems make use of the same, finite stocks of natural resources to satisfy their wellbeing needs – namely land area, water, and energy (from food, etc.) [30], [31], [32] – the two systems compete. Intuitively, demand for natural resources cannot exceed their rate of production. Where the combined demand of socioeconomic and ecological systems would exceed the rate of natural resource production, a trade-off is necessary. This trade-off manifests as a compromise between satisfying human and ecological wellbeing needs, and between human development and environmental conservation, as illustrated in Fig. 1.3.

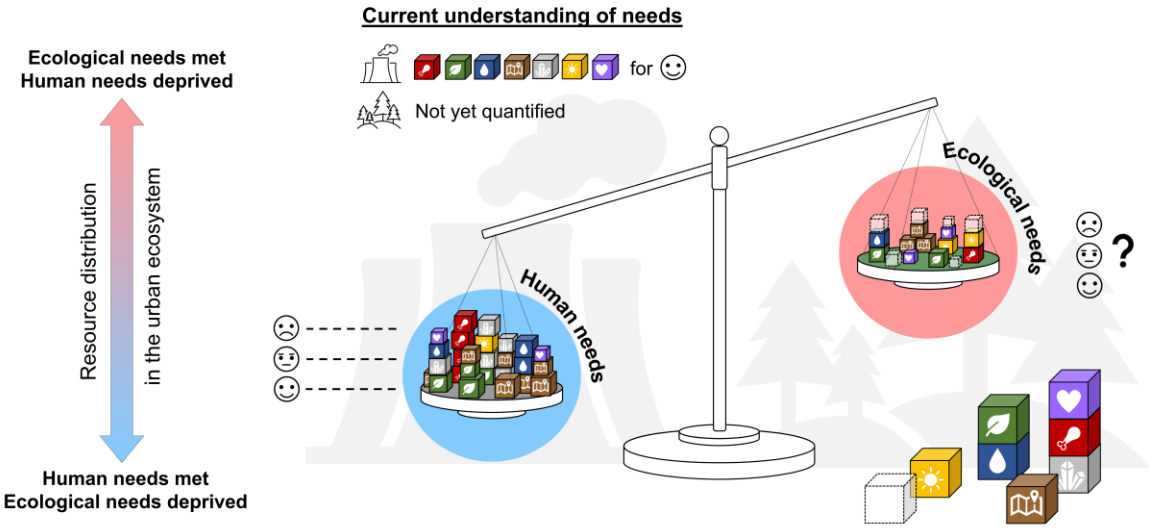


Figure 1.3. The implications of natural resource (mis)management between human and ecological systems, where the needs of humans and/or nature are met or deprived. Seven 'needs' are illustrated: access to sunlight (yellow with sun image (☼)), energy from vegetation (green with leaf image (🍃)), water (blue with raindrop image (💧)), land area (brown with map image (🗺)), energy form meat/animals (red with drumstick image (🍗)), access to minerals (grey with crystal image (💎)), and social interaction (purple with heart image (♡)). The clear, dashed cube illustrates unfulfilled needs. These represent both social and physiological needs. Sustainable resource distribution – balancing the scales – requires a quantitative understanding of needs in both systems. In which case, natural resource provision and ongoing intergenerational opportunities, for humans and nature, are secured. Three general states of wellbeing (needs satisfaction) are illustrated: good, (☺), adequate (😐), and poor (☹). Currently human needs are well defined quantitatively, but ecological needs are not. Here 'ecosystem' describes a combined socioeconomic-ecological ('human-nature') system.

Given that some level of ecological wellbeing must be sustained to achieve ongoing human development [33], the distribution of natural resources between socioeconomic and ecological systems must be carefully managed. However, the unavailability of quantitative ecological wellbeing needs makes this difficult. The sustainability of different natural resource use cannot be distinguished until this deficiency is addressed. The prioritisation of human needs in industrial ecology further compounds this problem. This prioritisation is a consequence of anthropocentrism within the field [34], which seeks to optimise human development. Even so, biased intentions to satisfy human needs in the present may, limit our ability to satisfy those same human needs in the future. If ecological needs are sufficiently deprived [35], the principles of sustainable human development, which aims to achieve *ongoing* human development [36]. This requires intergenerational access to natural resources and thus, intergenerational ecological wellbeing. Fig. 1.4 illustrates this cause-effect mechanism. With ecological wellbeing needs not yet quantitatively described nor fully understood, our intergenerational opportunities may already be undermined.

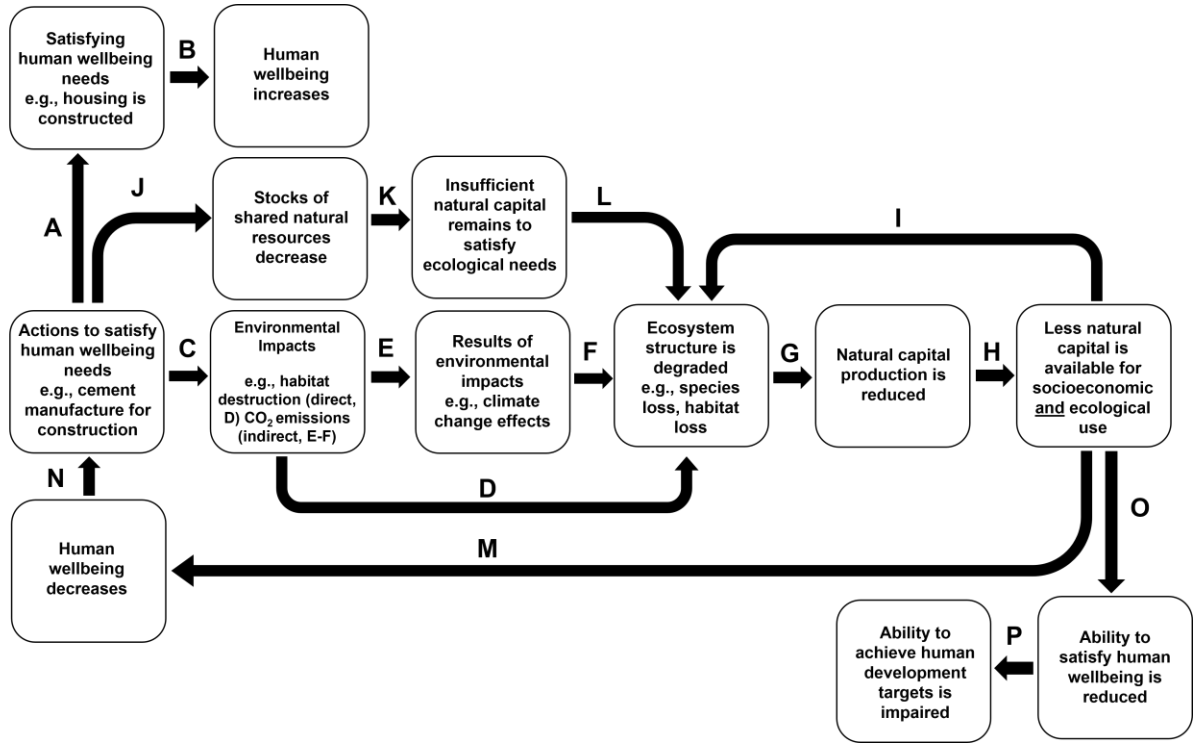


Figure 1.4 A cause-effect mechanism where ecological systems are degraded following two main causal pathways. First, due to the environmental impacts associated with meeting human wellbeing needs (**C**), either directly (**D**) or indirectly (**E-F**); and second, due to a trade-off between socioeconomic and ecological systems driven by limited natural resource availability (**J-L**). This degradation reduces the production (**G**) and hence the availability (**H**) of natural resource, which further degrades ecological systems (**I**). This exacerbates the impacts of socio-economic activities. In response, human wellbeing decreases (**M**) and our ability to satisfy intergenerational human wellbeing needs is reduced (**O**). Our ability to achieve human development targets is thus impaired (**P**)

1.2 The urgency of biodiversity loss

The Convention on Biodiversity (CBD) defines biodiversity as, 'the variability among living organisms… [including] diversity within species, between species and of ecosystems' [37]. Here, 'ecosystem' describes a combined socioeconomic-ecological ('human-nature') system [38], [39], [40]. This is maintained throughout this thesis. Despite global efforts to conserve biodiversity (e.g., [41]), global rates of biodiversity loss are at a critical level, occurring at 100 (sometimes 1000) times the natural background rate [42], [43]. There has been a 68% decline in wildlife populations since 1970 – a change largely attributed to human activity [44], and most pronounced in the Global North [45]. This issue is pertinent as biodiversity contributes to ecosystem function and hence underpins human development. Biodiversity loss and the associated degradation of natural habitats therefore reduces the capacity of ecological systems to function. These losses then enact a cascade effect resulting in further degradation [46], [47]. Socioeconomic systems will be severely impacted and though difficult to

predict, the loss of ecosystem functions will be detrimental to human (physiological and economic) wellbeing.

Ecosystem dynamics, interactions between different species and their metabolism of energy and other resources, drive ecosystem functions that underpin contemporary socioeconomic systems and strongly influence human wellbeing. The composition of species within an ecosystem and their interactions with one another, and their surroundings determine its stability, resilience, and productivity [48]. Metabolism describes the transformation or use of energy and nutrients, typically through processes such as predation, respiration, decomposition, and photosynthesis, which sustain critical ecological processes [49]. These processes directly impact human wellbeing by providing essential ecosystem services like climate regulation, and the provision of clean air and water, and fertile soils vital for agriculture. Disrupting ecosystem dynamics, particularly the metabolism of energy by organisms within ecosystems, can threaten these services, posing risks to human health, wealth, and wellbeing [33]. Therefore, understanding ecosystem dynamics and the metabolisms of species is critical to social and economic development, making them primary interests in urban ecology.

Given the threat biodiversity loss poses to humans and the wider ecosystem, the ability to accurately assess and reduce the impacts of human activities on biodiversity has become a priority. In LCA, biodiversity (species) loss is used to describe the environmental impacts of human activities in a tractable way [50], [51]. Current deficiencies of LCA method regarding biodiversity and ecological wellbeing needs are discussed critically reviewed in ‘Section 2.4.7’, Chapter 2.

1.3 Land use and urbanisation as a driver of biodiversity loss

Land use change – the transformation and occupation of (natural) habitats to support human activities – is a priority driver of terrestrial biodiversity loss [52], [53]. Urbanisation is amongst the most significant driver of this land use, alongside agricultural and forestry systems, all of which are driven by human needs [54], [55]. For example, the need for housing, crops, and timber. Urbanisation requires the manufacture, transport, use, and disposal of many resources (cement, glass, steel), generating a considerable environmental stress as a result. With population and affluence (Eq. (1.1)) increasing, natural habitats continue to be transformed and occupied to satisfy growing human demands for food and resources, and to accommodate urban living. Without action, biodiversity loss will worsen. This prospect is driving innovation, with sustainable technologies at the fore. This thesis focusses on urbanisation. There are two main reasons for this: urbanisation constitutes a range and intensity of

human activity necessitating innovation, while also demonstrating unique potential for the integration of socioeconomic and ecological systems in a way that benefits both [56].

Urbanisation is driving global land-use change, threatening biodiversity [57] and the provision of natural resources [58]. Over 50% of the global population now live in urban areas [59]; a figure expected to increase to 86% by 2050 in the Global North [60]. As urban populations and their affluence [61] grow, so too will their demands for urban development [62]: the construction of grey (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 [63], [64]. In the context of urbanisation, land-use change is defined 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 [65], [66]. In addition to its inherent value, biodiversity contributes to the provision of natural resources (e.g., timber, minerals, water) and services (e.g., flood defence, air purification) on which socioeconomic systems depend [20], [67], [68]; as such, urbanisation threatens the resources upon which humans rely [69], [70]. With urbanisation 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.

1.4. Responses to biodiversity loss in the urban context

The concept of 'sustainable urban design' aims to meet the growing demand for urban infrastructure in ways that benefit humans and biodiversity simultaneously. This includes the reduction of environmental impacts. The construction of biodiversity-supporting urban infrastructures is at the fore. These infrastructures employ ecological components – plants, water, and geological materials like soil and rock – to address challenges relating to climate change, food & water security, and human health [71], [72]; challenges that drive their application [56]. 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 [73], [74], [75]. Green roofs have become common in urban landscapes, with government and legislative bodies encouraging their use [76]. Amongst their many reported benefits, green roofs support biodiversity including in densely populated urban areas [15]. However, the extent of this support beyond their constituent plants remains poorly understood, as do the mechanisms underlying this benefit. Studies have focussed on plants and arthropods (e.g., invertebrates) [77], with limited research on higher trophic levels [78]. This knowledge gap undermines society's ability to

assess the biodiversity impacts of green roofs or their efficacy in contributing to sustainable urban design [79].

LCA informs the design and application of biodiversity-supporting urban infrastructures, but the impact pathways between human interventions (the action) and species loss (the impact) are not yet fully described. Poor transference of ecological data into LCA compounds the problem. The complexity of ecological systems makes responses to human intervention difficult to generalise in ways compatible with LCA. There is a need for greater integration of biodiversity into the LCA method [80]. Winter et al. (2017) [81] proposed the development of new midpoint and endpoint categories pertaining to biodiversity. In their critical review of LCA literature, Larrey-Lassalle et al. (2022) [82] identify the need to distinguish 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) [83], whose attempt to analyse 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.

There remains a need to characterise 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 [84], IMPACT World+ [19]) are limited. Characterisation should reflect the varied design of NbS as urban infrastructure and describe their effects across more comprehensive trophic systems. By modelling local ecological systems explicitly, the in-situ impacts of urban land use on terrestrial biodiversity can be quantitatively characterised 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.

1.5. Central hypothesis in this thesis

The central hypothesis of this research is that the ability to quantify ecological wellbeing needs, analogous to human wellbeing needs, could improve our capacity to model natural resource use comprehensively across entire ecosystems (socioeconomic-ecological systems); facilitate more accurate assessment of the environmental impacts associated with socioeconomic natural resource use; and thus, manage global natural resource stocks in a more sustainable, and intergenerationally secure, way.

Therefore, this thesis aims to develop a concept and accompanying method for quantifying the wellbeing needs of ecological systems through which the life cycle impact of biodiversity-supporting urban infrastructure on biodiversity can be quantified more comprehensively and then integrated into the LCA method. This has excellent potential to improve both the design of biodiversity-supporting urban infrastructure, to ensure adequate and appropriate provision for desirable ecological output, and the assessment of any urban land occupation type with capacity for natural resource provision: gardens, ponds, hedgerows, and so on. This work addresses critical knowledge gaps within industrial ecology in a novel way, while providing a preliminary means to address deficiencies within the other fields – environmental engineering, ecological engineering, and conservation biology (Fig. 1.5) – whose work it ought to underpin. Finally, this work acts to integrate ecological understanding into engineering, demonstrating the value of interdisciplinary knowledge transference and encouraging others to do the same.

The concept of 'ecological needs' [26] represents a critical deficiency across ecological engineering, conservation biology, environmental engineering, and industrial ecology (Fig. 1.5).

- In ecological and environmental engineering, ecological needs offer a means to understand quantitatively what resources are needed for appropriate (as needs demand) urban ecosystems to establish and persist, ideally with minimum human intervention [56], [85].
- In conservation biology, ecological needs may improve our understanding of best practice in environmental management, including biomanipulation, habitat restoration, and species (re)introduction [86].
- In industrial ecology, as discussed, natural resource modelling can be improved alongside LCA, to achieve holistic, ecosystem-wide inventory for more sustainable natural resource management and greater technological development (see Fig. 1.2).

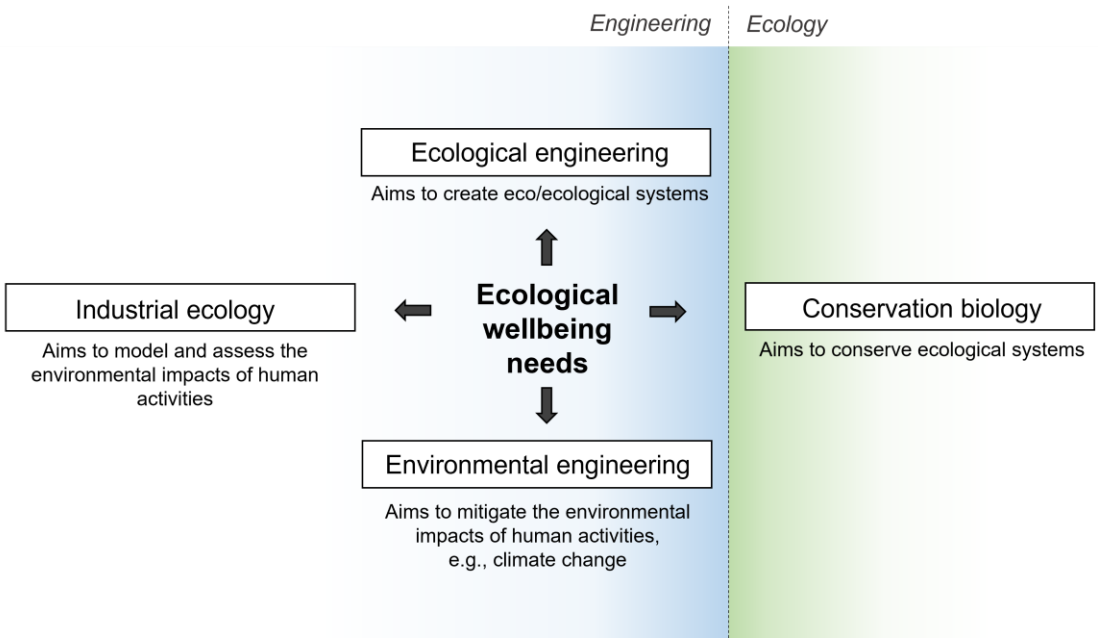


Figure 1.5 Illustrating the concept of ecological needs as a nexus between industrial ecology, ecological engineering, environmental engineering, and conservation biology; and hence between the disciplines of engineering (left, blue gradient) and ecology (right, green gradient).

As a nexus between different fields, and a bridge between engineering and ecology, the concept of ecological wellbeing needs represents myriad opportunities for novel, multidisciplinary advancement. However, such a task is too great for a single PhD and its complexity makes a perfectly linear development difficult. As such, this thesis represents only the first stage in the concept's development, within the context of industrial ecology: (i) the concepts development (Chapter 4), and (ii) its inclusion within LCA to advance the design and assessment of biodiversity-supporting urban infrastructure (Chapter 5).

The literature review in Chapter 2 identifies how the concept of ecological needs forms a multidisciplinary nexus between ecology and engineering (Fig. 1.5). The need to better include biodiversity in MFA and LCA is also discussed, and a critical overview of the state of the art in quantifying species' physiological needs and designing biodiversity-supporting infrastructure is provided. The literature review includes detailed discussions of LCA and LCIA, the application of which underpin this thesis ('Section 2.4.7').

In Chapter 3, equations for estimating the land area use, water consumption, and energy requirement of terrestrial mammals, birds, reptiles, and invertebrates are developed. These represent a starting point in quantifying the ecological needs of organisms under greater temporal (e.g., accounting for the effects

of habitat and climate) and phylogenetic (e.g., describing additional gena) disaggregation. The consequences of overlooking ecological needs in the context of global natural resource management and human development are also discussed. The Natural Resource Laboratory in Scotland is used as a case study at this initial stage, through which complications in interpreting species land use requirements are identified.

Chapter 4 presents a case study whereby the ecological needs concept is advanced. Here, the equations presented in Chapter 3 are used to address a deficiency in the LCA methodology: the inability to quantify the local impacts of land use change on biodiversity. An urban development in London, UK, is used as a case study to develop a new endpoint characterisation factor for land occupation as sedum roofs. This case study demonstrates the transference of ecological knowledge into engineering in a novel way, to quantify the local impact of human activities – in this case, the installation of sedum roofs – on terrestrial biodiversity. This work represents multidisciplinary advancement. This work presents a means of mapping trophic networks complimentary to dietary data (e.g., predation frequencies, stomach contents) or where that data is unavailable. It addresses a critical deficiency in LCA, facilitating more comprehensive inventories of natural resource use and more informed assessments of human activity. It also demonstrates, conceptually, how a quantitative understanding of ecological needs can inform urban design, to produce biodiversity-supporting infrastructure tailored to specific socioeconomic-ecological needs.

Chapter 5 presents a method to quantify the natural resource requirements of terrestrial ecosystems, informed by the findings of Chapter 4. Food web data for the American kestrel on St Martin Island, Caribbean, is used as a case study to demonstrate the method which includes more comprehensive data on interspecies interaction (acts of predation) [87] than was available for the case study in Chapter 4. Metabolic variability, $E_{var,i}$, is accounted for and a less restricted timeframe is used. The modelled ecological system and its natural resource requirements are inherently less speculative as a result. The capacity to scale natural resource use at the system level scales from that of the individual level in a generalisable way is also investigated. The results from Chapter 5 facilitate a greater understanding of ecological systems. The potential impact of different types of land use change on the modelled ecological system can also be inferred, demonstrating conservation value. This insight is of value in urban design, too, as it offers quantitative data on the provision required to support the American kestrel and the species underpinning it. This understanding can improve efforts to conserve or create equivalent ecological systems on St Martin Island.

The results from Chapters 3, 4, and 5 are then summarised in Chapter 6. This chapter therefore describes the improved capacity to assess and design biodiversity-supporting urban infrastructure. It also describes the wider potential of the developed concept and accompanying methodologies to manage and (re)create ecological systems in both urban and natural settings, and its potential in developing more rigorous policy and targets for sustainable human development alongside environmental conservation. Ultimately balancing the needs of socioeconomic and ecological systems – between human development and conservation – represents a complex, ethical dilemma far beyond the scope of this thesis. Here, the scope is restricted to developing the concept of ecological needs and an associated methodology for their application in the urban context.

Chapter 7 outlines recommendations for future work, based on the research in Chapters 3, 4, and 5. The identification of current limitations to the development of this the 'ecological needs' concept and methods, and the means of overcoming those limitations, are an important aspect of this thesis; a first step in a considerable undertaking. Finally, Chapter 10 contain the appendices for this thesis: Appendix 1, 'Supplementary information for Chapter 3' (10.1), Appendix 2, 'Supplementary information for Chapter 4' (10.2), Appendix 3, 'Supplementary information for Chapter 5' (10.3), Appendix 4, 'Supplementary data for Chapter 3' (10.4), Appendix 5, 'Supplementary information for Chapter 4' (10.5), and Appendix 6, 'Supplementary data for Appendix 5' (10.6)

2 Literature review

This thesis brings together concepts from the fields of ecology and engineering, producing a broad literature review. The topics covered are outlined here to prepare and guide the reader. Human wellbeing needs and ecological wellbeing needs (the lack thereof) form the basis of this thesis. These are discussed first in 'Section 2.1' and 'Section 2.2', respectively. 'Section 2.2' includes a definition of the term 'ecological wellbeing needs' followed by discussions on the state of the art in quantifying the land use, water consumption, and energy requirements of terrestrial animals. Next, the literature review focusses on urban biodiversity and how it is supported through urban design. The term 'biodiversity-supporting 'infrastructure', coined in this thesis, is defined in 'Section 2.4.1'. The following topics are then discussed: how urban biodiversity uses urban spaces ('Section 2.4.2') – this informs urban conservation and urban design, urban-biodiversity support in contemporary urban design ('Section 2.4.3'), and the concept of ecological engineering ('Section 2.4.4'). The need to supplement qualitative ecological understanding with quantitative species data is evidenced in 'Section 2.4.5'. Finally, the literature review focuses on life cycle assessment (LCA), which informs the use of ecological components (e.g., vegetation) in urban design and contributes to our understanding of the impacts of urban conservation efforts on local species diversity. Relevant deficiencies in life cycle inventory and life cycle impact assessment are identified and discussed in 'Section 2.5.'. The literature review is concluded in 'Section 2.6'.

2.1 Human wellbeing needs

Wellbeing describes a sense of mental, physical, economic, and spiritual health or fulfilment [88]. What constitutes wellbeing is partially subjective, with different components prioritised across different cultures and belief systems [89], [90]. Maslow's 'Theory of Human Motivation' is amongst the most formative pieces of wellbeing research in the west [91], [92], [93]. Similar concepts include those of McClelland (1961) and later Doyal & Gough (1984) [94], [95]. Maslow described and categorised requisites of human wellbeing needs across three broad categories, relating to self-actualisation and spirituality, psychological wellbeing (i.e., self-esteem and sense of belonging), and basic wellbeing (i.e., safety and physiological needs). Though prominent, shortcomings are identified in Maslow's hierarchy [96], [97]. These include its use of social individualism, and the absence of interactivity between needs [98], [99], [100]. Notwithstanding calls to expand and restructure Maslow's hierarchy of needs (e.g., [101], [102], [103]), physiological needs (e.g., food, water, sleep) underpin all other human wellbeing needs as they are fundamental to human survival. This makes physiological needs the most objective and universal (shared by all humans) components of human wellbeing.

The physiological needs for human wellbeing can be generalised at the species (i.e., human) level to describe a minimum threshold for human wellbeing. An important and long-standing case for this is in its legal protection. Human wellbeing is intrinsically linked to human rights and the need to protect individuals' ability to achieve (at the very least some minimum level) human wellbeing. The Universal Declaration of Human Rights (UDHR) was produced by the UN General Assembly to protect this right [17]. It was created in response to human rights abuses during World War II, which exposed inadequacies within the cursory 1945 United Nations Charter [104]. The UDHR qualitatively describes fundamental human rights. This includes physiological needs pertaining to human wellbeing such as 'the right to water' and 'the right to food'. Physiological needs are covered by Article 25 which includes 'the human right to an adequate standard of living' [17]:

'Everyone has the right to a standard of living adequate for the health and wellbeing of him/herself and of his/her family, including food, clothing, housing and medical care and necessary social services, and the right to security in the event of unemployment, sickness, disability, widowhood, old age or other lack of livelihood in circumstances beyond [their] control.'

The UDHR has since served as the foundation for many subsequent international laws and agreements. It has formed an important basis for measuring human wellbeing and quantifying the resource use needed to achieve it.

Quantitative and qualitative measures of social and economic progress are used as indicators of human wellbeing. In much the same way our understanding of human wellbeing has evolved, these indicators have undergone substantial development over the last century. In 1967, Bauer proposed a set of social indicators to pre-existing economic indicators which, although widely used, did not measure quality of life. Economic indicators measure economic performance and activity [105]. Gross domestic product (GDP) is a prominent example which conveys 'the value of goods and services produced and sold by a country or countries' [106]. Bauer's work was pioneering. It redefined what ought to be used to measure human wellbeing at both the individual and (socioeconomic) system level. Social indicators have since undergone many refinements (e.g., [107], [108], [109], [110]) to better reflect the complex, multidimensional components of human wellbeing. This includes the inclusion of ecological components (e.g., [111], [112]). Over 100 human wellbeing indicators have since been described [113] and applied in different combinations as part of wellbeing measurement methods.

The method developed by the Organisation for Economic Co-operation and Development (OECD) is amongst the most prominent, to which 38 countries are subscribed [114]. The OECD method measures

wellbeing across 11 categories: ‘income and wealth’, ‘work and job quality’, ‘housing’, ‘health’, ‘knowledge and skills’; ‘environmental quality’, ‘subjective wellbeing’, ‘safety’, ‘work-life balance’, ‘social connections’, and ‘civic engagement’ [115], [116]. The method includes both objective and subjective indicators, capturing common disparities between material and psychological or psychosocial satisfaction [117]. The method measures inequalities across the 11 measured categories and facilitates intersectional analyses (e.g., by gender, by class, by ethnicity). By measuring the resilience of natural, human, economic, and social resources, it also aligns with the intergenerational aims of human development [118]. Prefabricated methods remove the need for individuals or organisations to select wellbeing indicators for assessment, for which little guidance is available [113], [119]. Their use is therefore widespread. The comprehensiveness and longevity of the OECD method places it at the fore.

In 2012, Rao & Baer [62] proposed a set of quantitative human wellbeing needs to be used and assessed concurrently. They quantified the resource use needed to achieve a decent standard of living in humans across 13 ‘basic goods’ (analogous to Maslow ‘basic needs’ [93]) categories: food, water and sanitation, shelter, health care, education, clothing, television, refrigerator, mobile phone, and mobility. This work built upon the UDHR [17] to understand how best to manage resources at the global scale to provide sufficient, intergenerational access for human wellbeing and development [120], [121], [122]. Rao expanded his concept in 2018 [123], asserting like many before that food (energy, nutrition) and water (for drinking, cooking, and cleaning) are essential components of physical wellbeing [124]. Understanding of food and water requirements continues to improve. Expedited by growing interest in health and fitness amongst individuals [125], [126], there now exists abundant, comprehensive data on metabolism (e.g., [127], [128]), nutrition (e.g., [129], [130]), and hydration (e.g., [131], [132]) in humans. As such, there is a robust, quantitative understanding of natural resource use for human wellbeing.

The impacts of achieving human wellbeing have been assessed at the global scale [133], [134]. This includes life cycle analyses of individual human consumption [135], as well as consumption-based scenarios of resource use and human development across different social [136], [137] and temporal-spatial scales [22], [138], [139], [140], [141]. These assessments often focus on environmental impacts. Ecological systems contribute to both objective and subjective components of wellbeing [20], [90]. Sufficient ecosystem quality must therefore be protected to secure intergenerational human wellbeing and development [142]. However, securing adequate environmental conservation concurrently remains a challenge. Two important reasons include: (i) complex ethical, economic, and political issues hinder global resource management [143], [144], [145], [146], and (ii) there is a need to better understand the

connection between human wellbeing and ecosystem function, and the impacts of human development on ecological systems [147], [148], [149], [150]. This thesis focuses on the second. It is the consequence of persistent and restrictive anthropocentrism which has obscured activities beyond the socioeconomic domain [34]. Securing intergenerational human development requires an (ecological) system-wide perspective, where the needs of other species are considered alongside those of human [142].

2.2 Ecological wellbeing needs: the natural resource requirements of animals

2.2.1 Proposed terminology: 'ecological wellbeing needs'

This thesis establishes the term 'ecological wellbeing needs' to describe the physiological and social needs of organisms comprising ecological systems (e.g., plants, animals, fungi, etcetera), analogous to 'human wellbeing needs' for humans.

The concept and terminology of 'ecological wellbeing needs' are novel. 'Ecological (or ecosystem) wellbeing needs' is used in the literature to mean 'ecosystem quality' or 'ecosystem health', without explicit acknowledgement of the underlying (e.g., physiological) prerequisites [151], [152], [153]; or in the case of 'ecological wellbeing needs performance', 'the efficiency of transforming ecological consumption into human wellbeing' [154].

2.2.2 Quantifying ecological wellbeing needs

Whereas human needs are quantitatively understood [62], [123] and have been applied at the system level, e.g., [155], [156], as far as the author is aware, resource use by plants and animals has not been viewed or applied in the same way [26]. There exists extensive ecological knowledge of organisms and ecological systems, but a poor quantitative understanding of their use of natural resources [157], [158]. This is caused by a lack of quantitative data at the species level, but the problem is exacerbated by poor transference of knowledge from ecology into other disciplines. To address data gaps relating to species' ecological needs, generalised predictions are made through statistical regression [159]. These allometric ('trait scaling') equations allow one attribute (e.g., metabolic rate, growth rate, body size) to be predicted from another, typically at the species level and more generally at higher phylogenetic levels (e.g., genus). The field of this study is 'allometry':

2.2.2.1 Allometry

Allometry is the study of how morphology (e.g., body mass) scales to physiology (e.g., metabolism), behaviour, and other morphological aspects in living things [160]. The practice was originated by Snell

[161] in 1892, with Thompson [162] and Huxley [163] later contributing to the founding literature (in 1917 and 1932, respectively). In the 1960s, McNab published a seminal work demonstrating the relationship between body mass and home range – a measure of land use – in mammals, and the influence of metabolism and diet on this scaling [164]. This was formative in the study of physiological allometry, and the concept of ecological wellbeing needs developed in this thesis. McNab's findings were substantiated and advanced, with the need to disaggregate by trophic level, dietary guild, and body mass ranges when describing allometries demonstrated [165], [166], [167], [168], [169]. With abundant species data now available, and the practice of allometry established, allometries have now been described for mammals, birds, reptiles, amphibians, and invertebrates (to a lesser extent). Studies have also expanded beyond animals, with allometries proposed for plants – describing relationships plant morphology (phytomorphology) and plant physiology, e.g., water uptake [170], [171] – and fungi [172].

Allometric equations are the predominant means of predicting species physiology, particularly when data on specific species are limited or unavailable. Their versatility and applicability across a wide range of disciplines is an important reason for this. Allometric equations are described for countless metrics, describing myriad organisms at different levels of disaggregation (e.g., [173], [174], [175], [176], [177]). However, allometric equations assume uniform scaling (e.g., across a species or genus), which is not always true. Species can demonstrate considerable physiological differences in different temporospatial contexts [178]. Allometric equations are not able to capture the full complexity of physiology across different species and different contexts. Allometric equations typically operate within a restricted data range, contingent on their underlying data. Where their application requires extrapolation of this underlying data, outputs become less accurate and may become erroneous. These operating bounds are not always explicitly clear. While allometric equations represent a field of active research key to overcoming data gaps, their limitations should be acknowledged, and their results implemented with due caution.

2.2.3 State of the art in quantifying the ecological needs of terrestrial animals

Three pertinent issues are observed regarding quantitative data on physiological ecological needs:

1. Data gaps exist. Most data describe individual mammals and birds, often limited to terrestrial, charismatic, and domestic (i.e., livestock) species [179], [180], [181]. Regarding terrestrial animals, there is an important gap for invertebrates species. This is despite these performing important functions as pollinators and decomposers in ecological systems [182], [183].

2. The data often do not possess geo-spatial metrics. This makes it difficult to disaggregate by location characteristic (e.g., habitat type, climate), forcing over-generalised regressions and low-granularity applications.

3. Water consumption is, overall, poorly understood. The lack of data describing the quantities and qualities of water needed to support different organisms is obstructive to sustainable resource management.

2.2.3.1 State of the art in quantifying rates of water consumption in terrestrial animals

Rates of water consumption is the least well documented ecological need. There is often a greater interest in the mechanism of water collection and intake than the quantities being consumed (e.g., [184]). The water intake of domestic species is best documented. Data exists for non-domestic species; however, these are limited to mammals and birds [185], [186]. The work of Paul & Roces (2003) on the fluid intake of ants [187] serves as a rare example of water intake data for invertebrates. Traditionally, research has focused on physiological studies, measuring parameters like water intake, evaporative water loss, and hydration needs [188]. These offer data on species-specific water demands and information on how different species adapt to water-scarce environments. The application of 'stable isotope analysis' represents an important area of research in explicitly mapping water use in ecological systems. This technology has been used to trace water sources utilised by different organisms to better understand their contribution to organisms' (including plants) diets, alongside other metabolic requirements such as minerals (e.g., [189], [190], [191]). This technology has also been shown suitable for use in quantifying invertebrate physiologies [192], where such data is commonly lacking.

Quantifying the water requirements at ecological (i.e., system-wide) scales remains challenging. The dynamics of water cycling are complicated, with considerable variability in the quality and accessibility of water in different contexts. It is also difficult to extrapolate individual water needs to populations and communities without comprehensive trophic data to underpin those extrapolations. One key difficulty lies in translating individual physiological data into ecosystem-level understanding. While technological advances like remote sensing have improved our ability to observe species movements, quantifying water use requites high granularity – beyond the capacity of satellite surveillance. This includes '(very) high resolution satellite imagery' [193], [194], [195] which lack adequate granularity to measure water intake in any species . Difficulties quantifying water intake through food, particularly in wild species, further hinders our understanding of water consumption in animals [196]. With a reliance on measuring water consumption through observation, addressing this quantitative data gap is likely to be time and labour intensive.

2.2. State of the art in quantifying land (area) use in terrestrial animals

Land use for individual animals is described in several ways. Traditional methods have relied on field work and observation: tracking species, describing movement patterns, home range sizes, and habitat preferences, or through inference: i.e., from resource stocks or pray abundances [197], [198], [199]. Technological advancements now complement traditional methods, improving their accuracy and yield. These include genetic (eDNA) monitoring techniques [200], and the use of satellite technology to collect high-resolution spatial data [193], [194], [195]. Traditional approaches, especially indigenous knowledge, remains critical to conservation efforts however [201], [202]. Integrating quantitative approaches with traditional field observations and modelling techniques represents an area of ongoing research. Despite recent advances, however, land use by individual animals remains difficult to predict at the ecosystem level, in part because species are highly adaptive when it comes to land use, producing considerable variability even at the species level. This makes generalisations (e.g., through allometry) difficult. It is also difficult to distinguish what portion and facets of land use are essential for wellbeing. Non-physiological needs and individual preferences (i.e., animal personalities) compound the problem: social needs are context specific, influenced by other individuals and the local environment, and personal preferences are inherently intractable [203].

There are also challenges when attempting to quantify land use consistently across species and ecosystems. Different animals exhibit diverse movement patterns – some are sedentary, and some are migratory – and use land areas for different purposes (e.g., hunting, breeding, courting). This complicates data collection and hinders generalisation across different species (i.e., at the population level) [204]. Furthermore, attempts to generalise interspecific (inter-species) interactions, such as predation patterns, introduces additional complexity; the interplay between different species and their environment can be unpredictable over large timescales [205], [206]; they may also occur over different temporal scales, which makes aggregation difficult. Variation in temporal and spatial scales in the ecological context makes selecting appropriate scales for studies critical and contentious [207], [208].

Population density (individuals km^{-2}) is a useful measure of land use for allometry. Concepts like home range (km^2 $individual^{-1}$), for example, describe the land use of individuals in isolation [167], whereby land use is reduced to a single-use resource [209]. In contrast, population density describes the land use of individuals as part of a wider population [210]. This better reflects population dynamics, making population density better suited for use in allometries. To this end, there now exists much data describing population densities of different species (e.g., TetraDensity and PanTHERIA [211], [212]). These data are generalised, through allometry, to describe mass-land use relationships at the trophic level disaggregated by dietary habits [213], [214]. Some geo-spatial disaggregation is also possible.

However, these data demonstrate bias towards terrestrial mammals and birds. Critically, data on invertebrate land use is omitted. This hinders the development of invertebrate allometries. This bias – of species and geographies – represents an important deficiency in our quantitative understanding of land use.

2.2.3.3 State of the art in quantifying energy (metabolic) requirements in terrestrial animals

Energy provision is often the limiting resource within ecosystems. The type, quantity, and quality of energy within an ecosystem strongly influence which species co-exist under different conditions [215]. Its provision therefore drives ecosystem function [215]. Energy enters the ecosystem as sunlight, where it is converted by primary producers (autotrophs such as plants) into chemical energy (such as carbohydrates or cellulose). It then travels through an ecosystem by means of different species interactions, such as pollination, predation, and decomposition. This metabolism is well studied in a multidisciplinary capacity. Allometries have also been developed describing the metabolic needs of animals in both wild and captive settings, e.g., [216], [217]. However, their transference in ecological engineering has not yet been fully realised. The flow of energy through ecosystems (its metabolism) is typically portrayed in food webs, which have mostly been qualitative in the past. Quantitative food webs are now being studied with urban ecosystem creation in mind, demonstrating quantitative and empirical methodologies, the identification of conservation-priority species, and influences of food web dynamics on ecosystem function [27], [218], [219], [220], [221], [222], [223].

Our current understanding of the energy requirements of terrestrial animals is based on measuring metabolic rates and energy expenditure at individual species levels [224], [225]. The application of molecular ecology, particularly environmental DNA (eDNA) analysis, is improving our understanding [226] of ecological systems. By coupling eDNA analysis with traditional ecological understanding and metabolic data, energy flow within ecosystems can be more described. This gives important information on trophic structures and population dynamics (i.e., inter- and intra-species interactions) [227]. This has important implications for ecological conservation [228], [229].A noteworthy development is the use of eDNA in sampling stomach contents. Examples include I. Nimz et al. (2021) (seabirds) [230], L. Díaz-Abad et al. (2022) (turtles) [231], and V. Merten et al. (2023) (deep-sea organisms) [232]. eDNA technology is used here to quantify feeding habits and hence, trophic dynamics at the community level. This facilitates the development of quantitative rather than qualitative trophic models, as contrasted in Fig. 2.1.

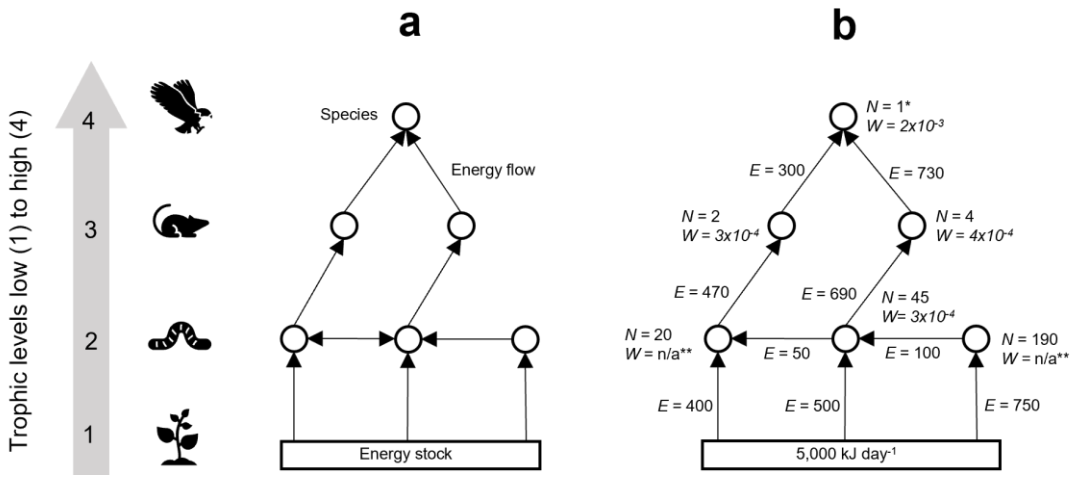


Figure 2.1 An example of (**a**) a qualitative (conventional) food web, and (**b**) a quantitative (ecosystem needs) food web. Both food webs have four trophic levels, with primary producers (e.g., plants), the starting point for energy flow within the food web, occupying the lowest trophic level. *N* (individuals) represents population size, relative to the apex species in this case (identified by an asterisk, where *N* = 1). *E* (kJ day^{-1}) represents an energy flow. The circles represent a species, the arrows represent an energy flow (e.g., predator-prey interaction), and the lowest trophic level (vegetation; primary producers) represent an energy stock. The arrow direction indicates the direction of energy flow. *W* denotes an estimated water consumption (m^3 day^{-1}) for each species; 'n/a**' refers to invertebrate water requirements which are not yet quantified [26]. The quantitative food web represents an ecosystem within specific spatial and temporal conditions. The web structure and values are generated for demonstrative purposes only

Fig. 2.1 contrasts (a) qualitative and (b) quantitative trophic understanding, in the form of food webs. In Fig. 2.1b, the energy flow between different species is quantitively understood. Stocks and flows of energy within the system are described. These relate to the constituent species' metabolic needs. In contrast, Fig. 2.1a demonstrates only a qualitative understanding of trophic interactions and energy flow: the type of food eaten by a species but not the quantity.

Though an enormous task, eDNA may offer a means of describing ecological systems in a way that allows 'quantitative food webs' (Fig. 2.1) to be produced with greater confidence than observational data or assumptions (as described in Chapters 4 and 5) can offer. However, eDNA is not without its limitations. False positives, variations in eDNA shedding (from animal to the aquatic environment) rates and decay rates, and difficulties in detecting rare or cryptic species pose challenges [233], [234]. Moreover, the quantification of energy requirements at broader ecological scales, encompassing entire

communities and ecosystems, remains a substantial challenge. These data gaps hamper our ability to predict how changes to resource provision, e.g., through human intervention, may impact ecological systems [226]. Furthermore, translating novel findings into conservation strategies requires ongoing transference of scientific knowledge and conservation action. Despite these challenges, the integration of quantitative approaches such as eDNA technologies can enhance our understanding of energy dynamics in terrestrial ecosystems and, subsequently, facilitate more effective environmental conservation efforts.

2.3 Supporting urban biodiversity through urban design

2.3.1 Proposed terminology: 'biodiversity-supporting component' and 'biodiversity-supporting infrastructure'

The term 'biodiversity-supporting component(s)' describes ecological (e.g., soil, water) or man-made (e.g., shelter: nest boxes, eaves) elements that can support the physiological and/or social requirements of organisms and hence ecological systems. In the urban context, infrastructures often contain biodiversity-supporting components that allow them to support organisms' needs. Prominent examples are structures (e.g., buildings, bridges) and artificial habitats (e.g., planted areas, façades), including cracks and crevices that serve as microhabitats.

The term 'biodiversity-supporting infrastructure(s)' is used as an abbreviation of, '*structures and habitats within the socioeconomic domain that comprise biodiversity-supporting components and hence can support organisms and hence ecological systems'*. It also replaces the term 'urban nature-based solutions', which is not fully suitable in the context of this thesis.

The International Union for Conservation of Nature (IUCN) defines nature-based solutions as, '…actions to protect, sustainably manage, and restore natural and modified [ecological systems] that address societal challenges effectively and adaptively, simultaneously benefiting people and nature' [235]. This includes actions within both socioeconomic systems and ecological systems. The term 'urban nature-based solutions' is used to distinguish its applications within the urban context. The problem is that terms relating to 'nature-based solutions' have anthropocentric connotations and convey too great a socioeconomic focus. Ecological components are reduced to tools in strictly problem-driven applications – the operative word being *solutions*. In contrast, this thesis focuses on ecological needs. Ecological components are viewed in a more holistic way with their provisional capacity at the fore.

The focus of this thesis is on use of ecological components in urban areas to support biodiversity rather than their operationalisation against socioeconomic pressures (biodiversity loss being the exception). The absence of 'solution', compared to 'nature-based solutions', emphasises that the integration of ecological components does not need to be problem-driven. As the meaning of 'biodiversity-supporting infrastructure' (i.e., infrastructure that supports biodiversity) is intuitive, it offers distinction without compromising the accessibility of this thesis. Finally, the proposed terminology is novel, which further prevents its meaning being obscured. No results for 'biodiversity-supporting infrastructure', as described above, were found in peer-reviewed literature. Similarly, the term 'biodiversity-supporting components' is intuitive and appears to be novel.

Finally, the term 'nature-based solution(s)' is not considered appropriate in this thesis as biodiversity-supporting infrastructures do not necessarily constitute a 'solution'; they may even degrade ecosystems or biodiversity, or negatively affect socioeconomic systems. For example, where their biodiversity-supporting components introduce highly-competitive, non-native species into established ecosystems; or instigate undesirable changes to the provision of ecosystem services. For these reasons, the use of the proposed terminology is appropriate and justified.

2.3.2 Urban biodiversity and modes of urban occupation in different organisms

Urban biodiversity describes the variety of living organisms and ecological systems within the urban context. The study of urban biodiversity represents an active area of research in urban ecology [236]. The study of biodiversity within urban contexts underscores that biodiversity is not confined to natural habitats. The biodiversity-supporting potential of urban areas is well documented, yet the conservation value of urban areas is often overlooked. An important reason being that urbanisation is associated with biodiversity loss and ecological damage. Disconnect between socioeconomic and ecological systems compounds the problem: urban areas do not resemble conventional natural spaces – they are dominated by concrete and human infrastructure. Furthermore, concepts like island biogeography theory state that species richness increases with area size [237]. This suggests small and fragmented habitats, which make-up urban landscapes, have little individual conservation value. However, they represent significant cumulative conservation value. The high heterogeneity of urban landscapes also confers high alpha diversity (i.e., local species richness), which can often exceed that of their homogenised (e.g., through agriculture) rural surroundings.

Many urban species were deliberately introduced by humans (e.g., for agriculture or horticulture) [238], but plants, animals, and other organisms have long since establish themselves in urban areas unaided -

in some cases, despite discouragement (e.g., rats, Japanese knotweed). Urban areas can support rich biodiversity because they produce abundant food, shelter, and stimulation, which opportunistic urban species exploit. Some provision comes from bio-productive urban habitats [239]. These generate different natural resources through ecological processes underpinned by autotrophic species. These natural resources are supplemented by human waste (e.g., sewage, food, refuse) [238], [240] and through acts of intentional provision like bird-feeding and 'pollinator-friendly' planting. Non-bio-productive habitats are also exploited, with urban species often found growing and nesting on infrastructures intended to satisfy human needs. Through these provisions, different species are able to meet their physiological and social wellbeing needs and survive within the urban landscape. Given these needs vary species to species, the capacity to support different species in urban areas is context specific and depend on factors like climate, location, and local species-pools.

The current state of urban biodiversity is not comprehensively understood. There is a lack of long-term data and standardised monitoring protocols for urban biodiversity, impeding comprehensive assessments. Conventional surveying methods do not fully account for the diversity of urban species or describe the interactions between them. Spatially explicit indicators and characterisation factors are subsequently lacking or described with insufficient granularity to describe urban landscapes comprehensively [241]. Research bias, resource constraints and accessibility challenges have also led to some urban areas being under-represented in datasets [242]. Upcoming technologies including remote sensing, (eDNA), and citizen science initiatives, offer more efficient and holistic data collection, increased accessibility to urban environments, and improved monitoring at finer temporal and spatial scales [243]. These advancements represent increasing opportunity to quantitatively understand urban biodiversity. Finally, the interdisciplinary nature of urban ecology poses challenges in terms of data integration and holistic understanding. An important consequence of this is poor transference of ecological knowledge into urban design. Effective resolution requires ongoing interdisciplinary collaboration between ecologists, urban planners, and policymakers [243].

Urban biodiversity is categorised into several modes of urban occupation with species demonstrating varied preferences and proclivities for urban living. Species' traits strongly influencing their capacities for urban living, with phylogenetically similar species experiencing relationships with urban areas [244], [245]. The extent to which different traits are beneficial in urban areas is context specific, but large litter size and generalise dietary habits are common. In this regard, urban species have also shown morphological and physiological adaptations: mice have adapted to digest more fatty food, and birds have begun singing in a higher pitch to be heard through traffic [246], [247], [248]. In cases, producing specialised urban populations [249]. However, the extent to which species benefit from urban living

does not always correspond to their likelihood of using urban areas. Spotswood et al. (2021) [250] categorises three primary modes of urban area use in species: active avoidance, active selection, or occasional use, based on species' tolerances to urban conditions, including anthropogenic disturbances, and the suitability of urban resources and habitats relative to their non-urban counterparts (Fig. 2.2).

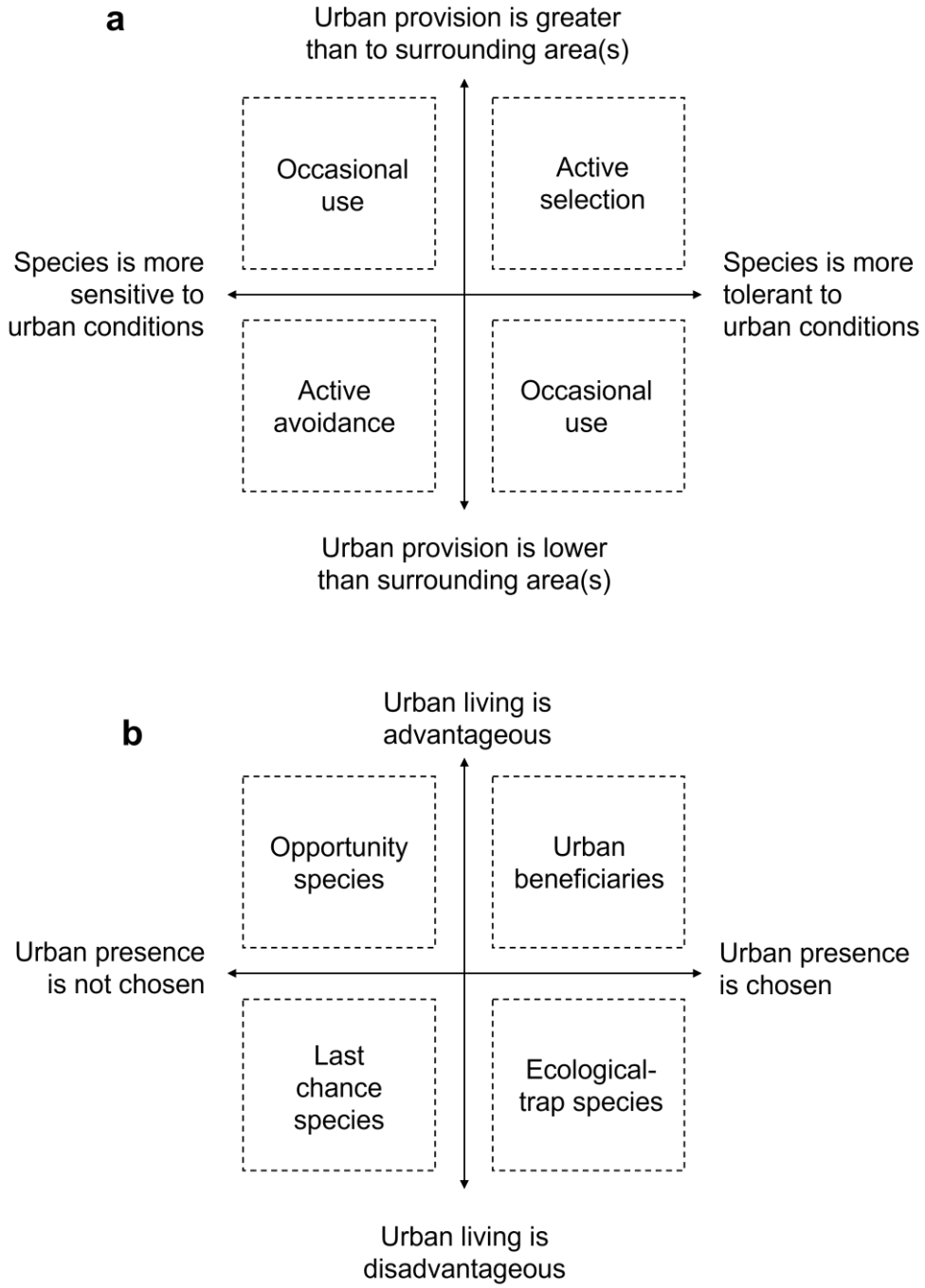


Figure 2.2 Generalised interactions between species and urban areas showing (**a**) the inclination of species to use or occupy urban area based on (i) their tolerance/sensitivity to urban conditions and (ii) the provision of resources in urban areas relative to surrounding areas, and (**b**) the different dynamics

between urban species and urban areas based on (i) the extent urban presence is (actively) chosen and (ii) the extent to which urban living is advantageous/disadvantageous. Figure adapted from Spotswood et al. (2021) [250]

Species demonstrating active avoidance of urban areas are typically highly disturbance intolerant, possess narrow ecological niches, or specialized resource requirements that urban settings seldom satisfy. Consequently, these species are rarely observed in highly urbanised environments, exemplified by species like the Lynx [251]. Conversely, species displaying active selection of urban areas are frequently observed in such habitats, as they exhibit a high degree of adaptation to urban living. These species are often small, opportunistic, and agile, reducing conflicts and predation risks. In extreme cases (e.g., brown rat (*Rattus norvegicus*) and rock dove (*Columba livia*)), species become dependent on urban conditions. Most species, however, exhibit behaviour that falls between these two extremes. These 'occasional use' species access urban resources as needed.

The dynamics of species interactions with urban environments reveal significant implications for conservation efforts. Here, Spotswood et al. (2021) describes four relationships between urban species and urban areas: 'urban beneficiaries,' 'opportunity species,' 'ecological trap species,' and 'last chance species.' Urban beneficiaries are species that demonstrate increased reproductive success and higher survival rates in urban areas, actively exploiting these advantages, often exclusively. Opportunity species, on the other hand, tend to be more abundant in non-urban settings despite potential benefits offered by urban environments, presenting opportunities for population enhancement through targeted interventions. Last chance species are typically endemic to urban areas following urbanization, often characterized by limited dispersal ranges [252], [253]. These endemic species represent a final conservation opportunity and thus confer considerable conservation value. A pertinent example of this is Cape Town, South Africa; with some 1,200 endemic species, it is among the most endemically-rich cities globally [254], [255].

‘Ecological trap’ describes the dynamic where, species occupy urban areas despite overall disadvantage. For example, an urban population of Eurasian kestrels in Vienna that favoured nesting sites within the city's urban core, affording increased reproductive opportunities but reduced food provisioning, resulting in comparatively lower reproductive success compared to their rural counterparts [256]. Where human activities (e.g., pesticide use, traffic) and human infrastructures (e.g., power lines) cause injury or death, they too contribute to ecological trap dynamics. The dynamic of ‘ecological trap’ species demonstrates a challenging aspect of predicting urban biodiversity-urbanisation interactions. Though qualitatively described, what represents advantage and disadvantage

for different urban species is not comprehensively quantified [250]. This represents an important barrier to supporting urban biodiversity through contemporary urban design.

2.3.3 Urban biodiversity-support in contemporary urban design

Urban areas are increasingly being used to support biodiversity, foremost motivated by its socioeconomic and ecological benefits. he presence of ecological components and biodiversity in urban areas has many benefits including improved human physical and mental wellbeing and improving the resilience and liveability of urban areas: purifying the air (e.g., through phytoremediation), and water (e.g., by means of wetlands), and regulating the local climate [257], [258], [259]. The ability to support urban biodiversity also represents a capacity to offset some of the biodiversity impacts caused by urbanisation. The social benefits of urban biodiversity include providing opportunities for recreation, education, cultural events, and a sense of community identity. Its education value is seen as particularly important, too, as ecological literacy has been diminished in urban populations through generations of urban living and shifting baselines of ecological diversity [260], [261]. Improving ecological literacy can bolster public enthusiasm for more effective urban conservation action [262], [263].

For these reasons, the support of urban biodiversity is at the fore of contemporary urban design, with many design concepts advocating the use of ecological components. Amongst the most common example is the use of ‘urban greening’: the creation of artificial, bioproductive habitats comprising plants like green roofs, green walls, and gardens [76]. Urban greening represents a prominent technology in the context of nature-based solutions, owing to their ability to mitigates the urban heat island effect, reduce energy consumption, and minimise enhancing urban resilience to climate change impacts. Other actions to support urban biodiversity include (i) increasing the bio-productivity and connectivity of urban habitats, natural and artificial; (ii) reducing environmental pressures within urban areas, e.g., light and noise pollution; (iii) (re)introducing ecological components (this includes functionally important organisms) to urban areas; and (iv) engaging with stakeholders and the public. This thesis focuses on (iii); at the fundamental level, ecological components can be used to support urban biodiversity through the provision of natural resources, such that their physiological and social (wellbeing) needs are met.

Efforts to support urban biodiversity are constrained by economics and stakeholder demands. These may conflict with or undermine the aims of urban conservation [236]. The temporospatial conditions at sites – their climate, land cover, topography, and the disturbances (i.e., from urban living) they are subject to [264], [265], [266] – also constrain urban conservation. The connectivity of sites to other

habitats and other ecological systems is also important: unless species are translocated, local (to each site) species pools represent the upper limits of species diversity, regardless of human interventions [27], [237], [267]. Persistent anthropocentrism in urban design compounds these practical issues, compounded by a poor qualitative understanding of how different species interact with urban areas and urban infrastructure of different designs [268].

Human needs are prioritised in contemporary urban design, or rather ecological needs are neglected [269]. This issue stems from the fact human needs drive urbanisation and hence guide urban design. In the UK, contemporary urban designs must meet strict regulations (e.g., [270]) that uphold safe and decent living conditions [17]. Ecological needs do not inform urban design in the same way. As such, infrastructures are not built to satisfy ecological needs (masonry-integrated [homes] for bats, birds, and bees the growing exception [271]). One reason for this is that ecological needs had not been quantified prior to this thesis. A poor quantitative understanding of how species interact with urban infrastructures compound the issue; what constitutes biodiversity-supporting and biodiversity-deterring infrastructures (or components) is unclear. This deficiency undermines contemporary urban design and, in many cases, result in approaches that overlook ecological needs and ecological impacts entirely. 'Biophilic design' for example.

Biophilic design is a prominent contemporary urban design concept that advocates the integration of ecological components into urban areas [272], [273]. Biophilic design aims to improve human wellbeing and improve the liveability of urban areas by exposing humans to nature. However, it prioritises human needs and overlooks any impacts beyond those to humans. This can be directly (physical contact with ecological components or features: light, air, water, vegetation, animals, weather, natural landscapes, and fire) or indirectly (contact with representations of ecological components or features, e.g., images and simulations of nature, and the forms and décor that evokes nature) [273]. Yet with human needs and aesthetic value prioritised over the requirements of ecological systems in biophilic design, ecological components are reduced to instruments for human comfort. Another issue is that the impacts of biophilic design components on urban biodiversity are overlooked and insufficiently monitored in the long-term. These issues undermine efforts to adequately support urban biodiversity through biophilic design. Where non-native species or monocultural designs are used, urban ecosystems may even be degraded [274].

Though anthropocentrism is a common issue in contemporary urban design, there are some contemporary design concepts that advocate 'wildlife-inclusive design'. The concept of 'regenerative

urban design' is a good example. Its main aim is to create sustainable and resilient urban areas by enhancing and restoring ecological systems while simultaneously supporting human wellbeing [275]. To this end, the concept seeks human-nature integration for mutual benefit of both systems. Unlike biophilic design, there is a focus on enhancing the biodiversity-supporting capacity of urban areas and their infrastructures. Again, urban greening is the most common practice. Wildlife-inclusive design features in several design practices [276], but was described comprehensively, as a distinctive concept, by B. Apfelbeck et al. (2020) [275]. The following are its main aspects: minimising the degradation of existing habitats, supporting urban species through provision of resources, and achieving greater understanding of the ecological needs of urban fauna. Importantly here, ecological needs are explicitly considered. However, considerable data gaps – namely the limited quantification of ecological needs and species physiologies (discussed in Chapters 3 and 4) – hinder the efficacy of biodiversity-supporting design in all its forms (i.e., in both urban and non-urban settings).

Another important limitation to designing urban landscapes for human and ecological use lies in the conceptual 'end-goal' of contemporary urban design: human-nature co-existence, 'humans and nature existing in proximity to one another' [277]. The aim of 'human-nature co-existence' is a common feature of urban design concepts seeking to support urban biodiversity (ecological needs) concurrent to human needs. However, it represents an important limitation. Urban areas comprise socioeconomic and ecological systems, which interact with one another. The term 'co-existence' gives the impression that the two systems are segregated when they are in fact consolidated: socioeconomic systems operate within the boundaries of ecological systems. This perpetuates the misconception that socioeconomic and ecological systems are separate. A notable exception to this is the concept of 'ecological engineering' which distinctly advocates for human-nature *integration*. An overview of the concept is provided below.

2.3.4 An overview of the concept of ecological engineering

Since its introduction in 1962 [278], the concept of ecological engineering has come to describe the creation, restoration, and management of sustainable ecosystems that integrate human and natural systems in ways that benefit both [278], [279]. Ecological engineering seeks to create, restore, or maintain appropriate conditions for ecological '*self-organisation*' (i.e., self-establishing and self-sustaining systems). Ecological self-organisation refers to the idea that ecosystems are self-organising at all scales [280]. Self-organisation describes how ecosystem dynamics and functionality are driven by internal processes alongside external (i.e., human) influence [281]. This means the functional outputs of ecological engineering designs are inherently speculative and may demonstrate variation over time.

By exploiting the self-organising capacity of ecosystems, functional and self-sustaining ecosystems are established with minimised need for intervention (energy input) throughout the ecosystem's lifetime.

Underpinning ecological engineering is the provision of suitable (type and quality) and adequate resources (e.g., food, water, infrastructure) to establish a stable population dynamic (i.e., trophic structures) without the need for further human intervention [56], [281]. Where designs are problem-driven, specific population dynamics are targeted to produce specific ecosystem functions. Given that different populations produce different ecosystem functions, provision of resources is key. Ecological engineering has five main applications [282]:

1. Engineering (an) ecosystem(s) to address a pollution problem (e.g., phytoremediation [283])
2. Engineering an ecosystem to address a resource problem (e.g., through forest restoration [284])
3. Restoring an ecosystem following its degradation (e.g., [285], [286], [287])
4. Modifying an ecosystem in ways that best maintain or improve ecosystem quality (e.g., biomanipulation of invasive species populations [288], [289])
5. Using an ecosystem for human benefit without destroying provisioning capacity (e.g., permaculture and 'sustainable use' frameworks [290], [291])

Large scale (e.g., regional systems: > 100 km^2) [292] applications of ecological engineering have focussed on the design of wetland or marine environments. Wetland design focusses on pollution remediation, whereas marine design focusses on habitat restoration. There are many examples of wetland design, most prominently in wastewater processing including sustainable urban drainage systems (SuDS) [293]. A notable example is the pioneering Everglades Nutrient Removal Project in Florida, USA, where ca. 4,000 acres of wetland was constructed to remove phosphorus (and other nutrients) from agricultural run-off that would otherwise enter and damage the Floridian everglades [294]. Large-scale marine ecosystem restoration and creation has been demonstrated in both fluvial (e.g., [295], [296]) and oceanic (e.g., [297], [298], [299]) contexts. Here, reef restoration has come to the fore, as a tool for conservation [297], [300] as well as the expedition of net carbon neutrality targets [301], [302].

In urban areas, ecological engineering at the large scale (i.e., city-wide) becomes difficult. Since urban areas are relatively densely populated, undeveloped land is relatively scarce and expensive [303], restricting ecological engineering applications to the small scale (e.g., mesocosm: 0.1-100 m) [292]. Here, applications tend to be problem-driven and have focussed on green infrastructures, like gardens,

and green roofs and walls [304]. They are often used to offset the ecological impacts of urban development, or to address pollution and resource problems relating to human health, urban resilience, or resource circularity [305], [306]. In this sense, urban ecological engineering applications are better known as a nature-based solutions [71], [72]. Common examples include green infrastructures (green roofs and walls, and gardens), street trees, integrated bird nesting sites, and wildlife ponds [307], [308], [309], [310]. As with all ecological engineering designs, these solutions operate by providing infrastructure and resources (energy, water) to support plants and animals [311], [312], [313] and thus facilitate the self-design of multifunctional ecosystems within urban settings [314].

2.3.5 A need for quantitative data to supplement qualitative ecological understanding

While the concept of ecological engineering represents a meaningful approach to supporting urban biodiversity alongside humans through design, it remains unclear what urban infrastructures must provide to support different species. At the heart of this issue is the need to quantify ecological needs (see Chapter 1). The creation of suitable foundations for ecosystem self-organisation is core to the creation and management of ecosystems in ecological engineering. The exact requirements are context-specific and ultimately depend on which species (at the community level) or which functions (at the system level) are to be supported. Uncertainty on the physiological requirements of different species and hence different ecological systems under different spatial and temporal conditions therefore represents a serious hindrance to the successful application of the concept. This problem is compounded by poor transference of knowledge from ecological disciplines into engineering disciplines, and a need to quantify what is qualitatively understood.

With considerable gaps in our understanding of urban biodiversity support, the design of biodiversity-supporting infrastructures often relies on qualitative data rather than quantitative metrics. A very general example being selecting plant species known to better support pollinator species to support nectivorous invertebrates. Where infrastructures are designed with species-specific provision in mind, there is often a reliance on species profiles (e.g., [315]) or equivalent to guide design. These sources are typically qualitative regarding species' ecological needs, describing only the *types* of food or habitat species require. So, while biodiversity-supporting infrastructures may be tailored to the type of food different species require or the type of shelter they seek, their capacity to meet the physiological needs of species is not understood. The reason for this qualitative approach is the poor quantitative understanding of ecological wellbeing needs, compounded by poor transference of ecological understanding from ecological into engineering disciplines. Until these deficiencies are addressed, biodiversity-supporting infrastructure will continue to be designed uniformed, potentially falling short of their underlying aim: supporting urban biodiversity.

In summary, urban biodiversity has become a critical consideration in contemporary urban design. Nature-based solutions like green roofs, green walls, and urban forests playing pivotal roles in supporting urban ecological systems for their own benefit while simultaneously benefitting the socioeconomic system. Wildlife-inclusive design – design informed by ecological needs – is increasingly common, but human wellbeing is the foremost motivating factor across contemporary design concepts. Ongoing research and data collection are needed to improve our understanding of urban biodiversity and its (potential) interactions with urban infrastructures. Difficulties quantifying what urban infrastructures should provide to support different species, and hence different functionalities, coupled with the imperative to quantify ecological needs, remain significant barriers to urban design. This undermines both the design and assessment of biodiversity-supporting infrastructures and prevents a comprehensive understanding of the potential impacts of human intervention – the use of biodiversity-supporting infrastructure in this case – on urban biodiversity. Addressing these issues is critical for the future of urban design and the eventual integration of socioeconomic and ecological systems. An important aspect of this is including biodiversity-supporting infrastructure within life cycle assessment.

2.4 State-of-the-art in biodiversity life cycle impact assessment (LCIA)

Assessing the life cycle impacts of different activities and interventions represents a substantial area of research in life cycle impact assessment (LCIA), a procedure in life cycle analysis (LCA). LCA procedure is standardised according to ISO 14040 and ISO 14044, where it is explained in detail [14]. Life cycle analysis (LCA) is a method for assessing the potential environmental impacts of a process (e.g., mining limestone, manufacturing cement, constructing houses) within a defined temporospatial boundary [316]. LCA is distinguished from similar assessment frameworks (e.g., product line analysis [317], [318]) by its focus on environmental sustainability. Social and economic sustainability are, for the most part, not assessed in LCA (though their incorporation has been demonstrated [319]). As a detailed discussion of LCA method is not pertinent for a thesis, the reader is referred to these ISO standards for further reading.

2.4.1 Species loss as an indicator of ecosystem quality at the endpoint

Of the 21 impact pathways described in IMPACT World+ (see Fig. 2.3), 13 contribute to ecosystem quality change: impacts to ecosystem health, resilience, and functioning, governed by biodiversity [33], [320]. These impact pathways include direct (e.g., land transformation) and indirect (e.g., climate change) mechanisms, in both terrestrial and aquatic (freshwater and marine) systems. Species loss serves as the indicator of ecosystem quality, reported as the ‘potential disappearing fraction’ (PDF) of

species over a defined area during a certain time (PDF.m^2.year) [19], [84], [321]. The strengths and weaknesses of using 'potentially disappeared fraction' are discussed alongside other metrics of biodiversity impact assessment in [322]. Potentially disappearing fraction has two notable limitations. Integral units, such as PDF.m^2.year, allow comparison of different potential impacts in LCA [323]; however, their form makes them equivocal, obscuring the timeframes of species richness and ecosystem quality change [324]. This makes it difficult to elucidate potential environmental impacts in the final stage of LCA (2.5.) or action resolution thereafter. Furthermore, potential disappeared fraction focusses on species richness and as such lacks data on the abundances or functional capacities of different species [325], [326]. Nevertheless, potentially disappeared fraction remains the most common indicator of potential ecosystem quality in LCA [327].

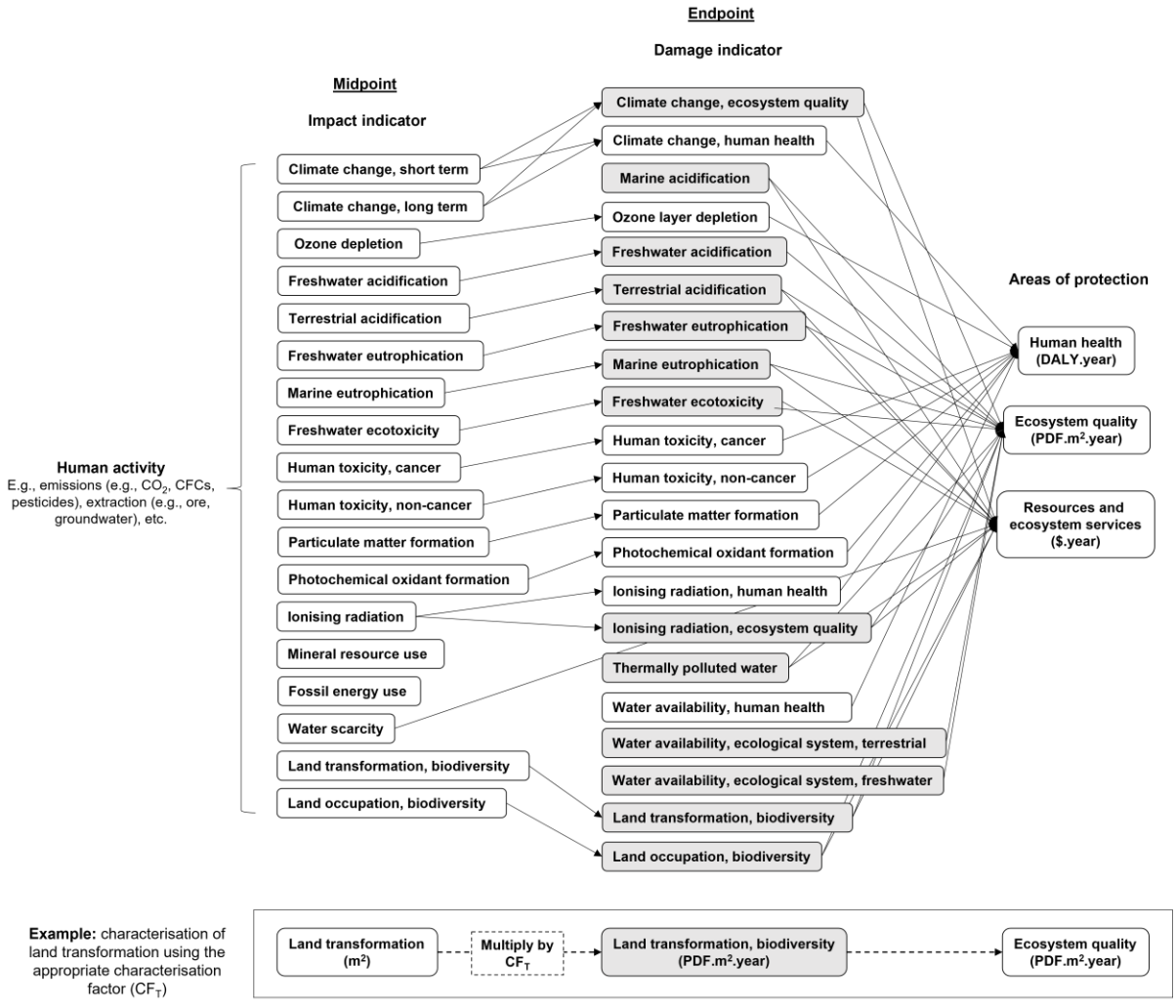


Figure 2.3 Impact pathways in LCA based on the IMPACTWorld+ assessment method, adapted from Bulle et al. (2019). Endpoint damage indicators pertaining to biodiversity (species) loss are distinguished, coloured grey. In the areas of protection, 'DALY' describes 'Disability-Adjusted Life Years', and 'PDF' describes 'potentially disappeared fraction'. The application of characterisation

factors to causally link impact indicators at the midpoint to damage indicators (potential impacts) at the endpoint is also illustrated. An example of the characterisation procedure is presented, showing the impact pathway for land transformation. Here, CF_T is the characterisation factor for land transformation.

2.4.2 The need to better include biodiversity and the functional effects of biodiversity change

There is a need to include more facets of biodiversity and a greater variety of environmental pressures associated with biodiversity change within LCA. Winter et al. (2017) and D. M. Souza et al. (2014) offer pertinent review of this topic [refs]. Most indicators used for biodiversity monitoring focus on alpha (α) species diversity – species richness predominantly – as reflected in the use of 'PDF' or 'species year^{-1}' in LCA. Genetic diversity, the variety of genes within a species or population [ref], ecosystem diversity, , and functional diversity, the variety and range of ecological functions performed by organisms within ecosystems [ref], are typically lacking. Measures of species abundance, too. Several indicators of biodiversity change are included in LCIA (see Fig. 2.3), but many environmental pressures known to contribute to species change are lacking. For example: noise and odour pollution, nutrient depletion, poaching, extreme weather events, changes to ecosystem dynamics, and land fragmentation. This limits LCIA method. It also reasserts the transference of ecological understanding into LCA is poor. What is urgently needed for comprehensive biodiversity LCIA is that these facets are measured concurrently across all potential drivers of biodiversity change. The inclusion of ecological wellbeing needs represents an important component of that advancement related to net primary productivity (NPP), human appropriation of net primary productivity (HANPP), and biotic production potential (BPP).

2.4.3 Land use and biodiversity in LCIA

Regarding land use in particular – the focus of this thesis – Larrey-Lassalle et al. (2022) [82] emphasise the need to distinguish between local (in-situ) and global (ex-situ) life cycle impacts of land use on biodiversity. This distinction is critical for understanding potential impacts in the context of urbanisation, particularly biodiversity-supporting infrastructure. Herein lies an important issue: while in-situ and ex-situ impacts are distinguished, assessment of local biodiversity impacts is lacking.

At present, ex-situ impacts are attributed to off-site land transformation and land occupation within (typically pre-defined) background processes (e.g., transport, resource acquisition, utility generation). Whereas in-situ impacts are attributed to on-site land transformation and land occupation during the use

phase of a product and must be inventoried manually. However, there is an important omission regarding land use in prominent LCIA methods like ReCiPe and IMPACT World+: the causal relationship between resource provision and availability at the midpoint, which is affected by land cover change (e.g., habitat creation or destruction), and species richness at the endpoint [83]. Therefore, despite biodiversity-support often driving their application, the potential impacts of different interventions that affect land cover and bio-productivity, like nature-based solutions, on local ecosystems are obscured.

Alejandre et al (2022) [328] make a noteworthy advancement in characterising the potential in-situ biodiversity impacts of different land cover types. They relate different land cover types to pollinator abundance, thereby establishing a new midpoint indicator for use in LCIA. This research leveraged expertise and data on pollinator abundances to bridge knowledge gaps in the causal relationship between provision and species richness, thus achieving transference of knowledge between ecology and engineering. Though an important step-wise advancement, the scope of the work is restricted to pollinators with wider impacts at the system level, relating to trophic structure or ecosystem functionality, for example, are overlooked. This methodology was subsequently applied by the authors to characterise potential impacts of different land use changes on pollinator abundance [329]. Both publications cited difficulties from data limitations and compatibility issues as limitations to this and similar work.

2.4.3.1 A conceptual barrier: the reductive inventory of land use change in LCA

The way land transformation and land occupation are described in LCA does not reflect the relationship between ecosystem quality (from which local species diversity is inferred) and changes to land use changes over time; it is reductive [80]. In LCA, land transformation constitutes two parts: '*transformation from…*' and '*transformation to…*' a land use type. When inventories, land transformations lack a temporal dimension. They are input as an area (e.g., m^2 transformed). As a result, they represent instantaneous conversion from one type of land cover and its associated 'potential ecosystem quality' (a metric inferred from indicators of environmental pressure at the endpoint, Fig. 2.2 [51]) to another. Consequently, land transformations simply change the point (the value of potential ecosystem quality) from which relaxation – natural restoration of the site to its reference state, i.e., the potential land cover in the absence of human activity [330] – commences.

In contrast, land occupation is input as a single, temporospatial parameter in LCA, '*occupation as…*': the area occupied multiplied by the duration of its occupation (m^2.year). In some cases [331], land

transformations are described as though they are land occupations, but the two interventions are generally differentiated. When describing occupation in LCA that ecosystem quality is static throughout occupancy. This allows the simplification of dynamic, non-linear impacts associated with ecological systems [80], and deficiencies in assessing species loss during occupancy [27]. These simplifications produce discrepancies between land use change and is rendering in LCA. Figure 2.4. demonstrates how these limitations manifest. It compares a theoretical life cycle of land use change (2.4.a) to its rendering in LCA (2.4.b).

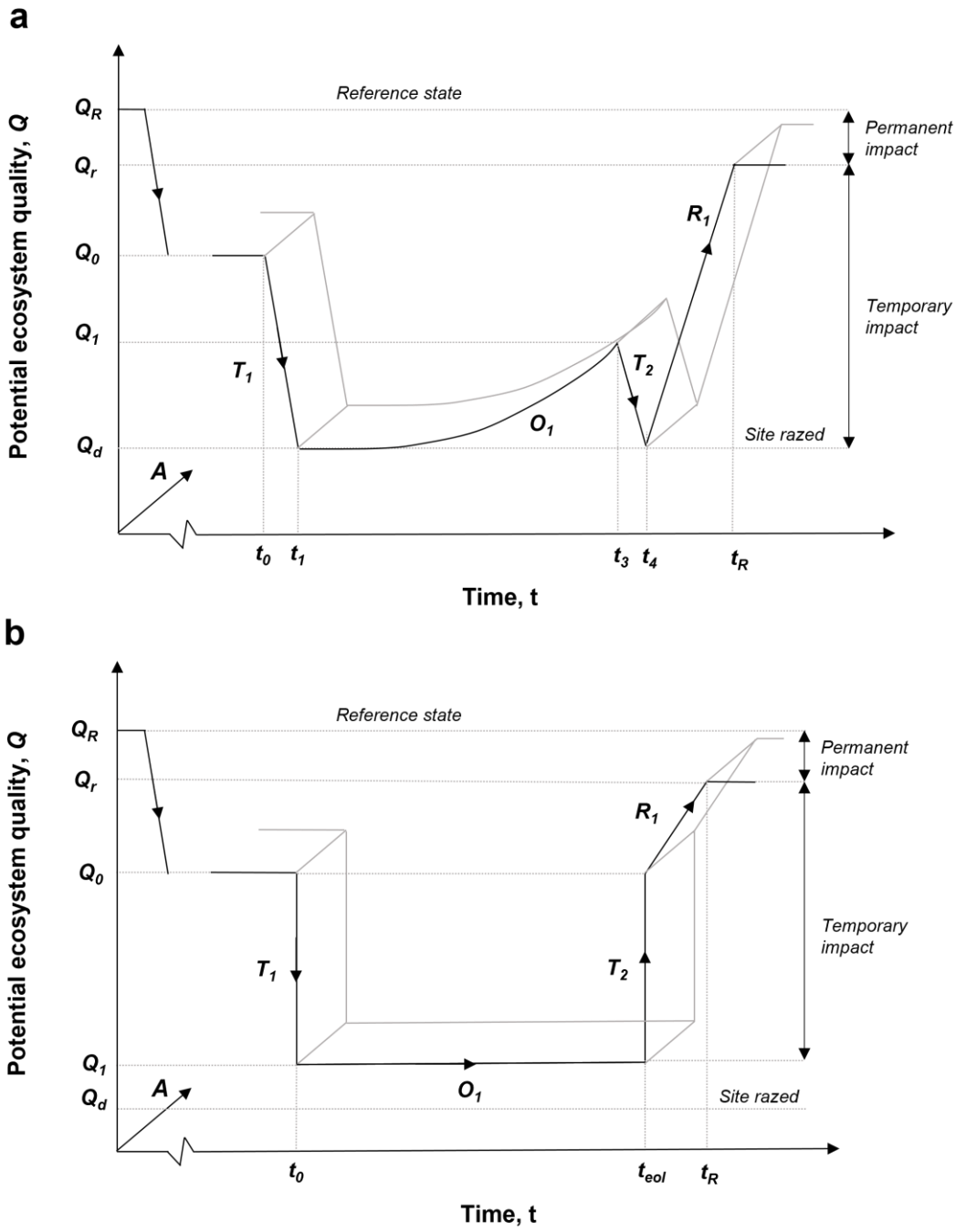


Figure 2.4 Land use during a development, (**a**) faithful to reality, and (**b**) as conventionally rendered in LCA. The figures describe change to potential ecosystem quality, Q, over time, t. The axis in the third dimension represents the area, A, of land cover change. The volume of the shape represents the life cycle impact on potential ecosystem quality (ΔQ). The activities in (**a**) are (i) land transformation T_1, from urban fallow (Q_0) to a razed site (Q_d); (ii) land occupation O_1, as residential, with ecosystem quality increasing over time to Q_1; (iii) land transformation T_2, from Q_1 to a razed state (Q_d) at t_4; and (iv) relaxation, R_1, which plateaus at t_R at Q_r ($< Q_R$). The activities in (**b**) are (i) land transformation T_1, from urban fallow (Q_0) to residential (Q_1); (ii) land occupation, O_1, as residential with static Q_1; (iii) land transformation T_2, from residential (Q_1) to urban fallow (Q_0); and (iv) relaxation, R_1, which

commences at the development's end of life (t_{eol}) and plateaus at t_R at Q_r (< Q_R). In both (**a**) and (**b**), the reference state is Q_R. Where Q is restored, i.e., through transformation T_2, impacts are temporary. Where full restoration is not achieved, i.e., R_1 fails to reach Q_R, impacts are permanent.

Consider the development of a site, of area A, from urban fallow (i.e., brownfield, Q_0) to residential (e.g., a housing estate). The reference state (Q_R) represents the site's natural, undisturbed state. In contrast to the contemporary landscape, the reference state for much of north-western Europe is broadleaf forest [330]. The site is first transformed through construction; the site is prepared, and buildings (houses and other amenities) and their foundations are built. In Fig. 2.4a, this activity is T_1, which occurs between t_0 and t_1. Ecologically, the site is razed to a state of low potential ecosystem quality (Q_d). Occupation (O_1) takes place between t_1 and t_2. Potential ecosystem quality increases to Q_1. Unless Qd is maintained, this will occur provided there is adequate provision to support increased species richness (e.g., through biodiversity-supporting infrastructure). A second transformation, T_2, occurs between t_2 and t_3. This represents end of life activities, including the demolition of houses and infrastructure, and the return of the site to an ecologically razed state (Q_d). From here, relaxation commences, but plateaus at t_R at a potential ecosystem quality (Q_r) less than its reference state.

In Fig. 2.4, the impacts of land use are described as permanent or temporary. Permanent impacts of land use change constitute irreversible change [80], [332]. These typically includes soil degradation, groundwater loss, and persistent ecotoxicity [333], [334]. Temporary impacts, on the other hand, are reversable. These include the installation and removal of temporary structures, such as solar panels [335]. The (ir)reversibility of environmental impacts (i.e., species losses) depends on both the type and magnitude of the intervention, and the resilience of the system being disturbed [336]. In Fig. 2.4, ecosystem quality fails to return to its original state (Q_R) through relaxation. The difference ($Q_R - Q_r$) constitutes a permanent impact. The rest are restored and thus temporary. Where the potential outcome is uncertain, timescales are used to differentiate permanent and temporary impacts of land use in LCA [331]. If impacts are restored within a 100-year timeframe, they are considered short-term and temporary; If recovery exceeds 100 years, the impacts are considered long-term and permanent.

Fig. 2.4b illustrates how the life cycle land use described in Fig. 2.4a is rendered in LCA. Most notable is the linearity of changes described. Contrast is most clear when the occupation stages in Fig. 2.4a and Fig. 2.4b are compared. This is because in LCA, characterisation linearises causal relationships. In this case, the causal relationship is between land occupation or transformation at the midpoint to potential ecosystem quality (and hence potential species diversity) at the endpoint. Three further points (i, ii, iii) demonstrate that land use is rendered in a reductive way in LCI:

i. Land transformations are described as instantaneous and their potential impacts able to cancel out. In LCIA, land transformations can produce a net-zero biodiversity impact for land transformation. This happens because 'transformation to' and 'transformation from' are equal and inverted (x , $-x$) in LCIA methodology. In Fig. 2.4b, $T_1 = -T_2$. Where a site is returned to its original land use category, a net-zero biodiversity impact for land transformation is therefore produced (i.e., $x + (-x) = 0$). However, if relaxation during occupation is described, as it is in Fig. 2.4a, T_2 may no longer be the inverse of T_1. This will affect the net impact of land transformation in LCIA. The inclusion of non-linear processes in LCA has been investigated [337], suggesting capacity to characterise the biodiversity impacts of land use in a less reductive way. However, this study did not explicitly consider biodiversity. The intractability of including non-linear causal relationships involving biodiversity or ecological systems, which are both complicated and dynamic, therefore remains unclear.

ii. Potential ecosystem quality during occupation is static. A site will undergo relaxation during occupation unless it is intensively maintained throughout its use. A gradual increase in ecosystem quality would typically be observed. The use of biodiversity-supporting components would further stimulate this increase. Fig. 2.4a illustrates this scenario, which Fig. 2.4b omits, (i) as gradual change (O_{1a}), characteristic of relaxation [338], and (ii) as linearised change (O_{1b}), as typically characterised [339]. This has an important effect on the second transformation stage, T_2, and hence the outcome of the subsequent LCIA.

iii. Inter-transformation land cover (i.e., the state Q_d) is omitted. In Fig. 2.4, this represents an 'ecologically razed' condition, such as that following demolition (high disturbance with lower potential ecosystem quality than the stable residential 'residential' land use type, Q_1). In LCI, each land cover type is allocated a value of potential ecosystem quality and transformation denotes change from one potential ecosystem quality to another only. To this end, Fig. 2.4b omits the ecologically razed state (Q_d) and describes the occupied state as its established state, Q_1. This impacts the occupation stage (O_1), the second transformation stage (T_2), and the restoration (R_1) stage.

These issues identified underscore the need for a practical, ecological mindset when applying LCIA. Here in particular, the potential change in ecosystem quality throughout a site's lifetime is for the most part intuitive. Knowledge of relaxation is widely understood, at least in a general sense (i.e., the process of urban ecological succession [340]). This knowledge is not ecological expertise, yet it is lacking in

LCIA applications. Ultimately, the inability to capture land use change 'faithful to reality (Fig. 2.4b)' produces flawed inventories that undermine LCA results. Resolving this deficiency requires more than a greater capacity for characterising biodiversity impacts (per the aim of this thesis), but also concerted effort to both transfer and then apply ecological knowledge. In this case, to apply LCA in a more faithful and comprehensive way.

2.4.4 Functional diversity and the functional value of species diversity in LCA

Ecosystem quality is directly related to the functionality and hence the functional value of ecosystems. Measuring ecosystem quality and functional value involves assessing various ecological indicators that reflect the health, resilience, and dynamics of ecological systems. In LCIA, measurements of ecosystem quality are typically restricted to alpha species richness. Other indicators are described and have been included in LCA, but units in LCIA are mostly restricted to PDF.m^2.year, species.year^{-1} [341]. Including functional diversity and the functional value of species diversity into LCA represents an active area of research. Though beyond the scope of this thesis, characterising the potential impacts of land use on ecosystem function and human wellbeing is of particular interest.

Species diversity, the composition of ecosystem – the variety of species and their distribution within ecosystems, is the most common quantitative indicators of ecosystem quality. The composition of species within an ecosystem and their interactions with one another, and their surroundings determine its stability, resilience, and productivity [48]. Therefore, species diversity provides insight into ecosystem complexity and resilience. Diverse ecosystems are capable of varied functionality and possess fewer functionally unique organisms (or greater 'functional redundancy'), which reduces the likelihood of functional roles going unfulfilled. It is typically through species diversity that functionality and functional value are captured within LCA.

While most ecosystem quality indicators included in LCA provide a static measure of diversity or composition, ecosystem function is the result of dynamic intra- and inter-species interaction. It is dynamics and, consequently, more difficult to describe and incorporate into LCA. Functional traits, at the species scale, and ecosystem function, at the system scale, are important indicators of functional value. The provisioning, regulating, supporting, and cultural services (e.g., carbon sequestration, water purification, pollination, and recreational opportunities) provided by ecosystems can be evaluated and related to human wellbeing. By relating the composition of ecosystems to their functionality, it is possible to link ecosystem quality to human wellbeing across different contexts. Furthermore, by examining the functional traits of species, such as their metabolism of resources and their roles in

different ecological processes, the functional value of different ecosystems and hence different measures of ecosystem quality is quantifiable.

A notable work here is that of Rugani et al. (2019) which focuses on the inclusion of the 'ecosystem service cascade method' [342] – first proposed by Haines-Young and Potschin, (2010) [343] – into LCA. Human activity is causally linked to potential impacts on ecosystem services and human wellbeing due to different forms of ecosystem change (e.g., land use change, eutrophication). Despite its innovative approach, however, biodiversity is included in a somewhat reductive way. Population dynamics (i.e., trophic structure) are not explicitly modelled or changes in these dynamics related to changes in ecosystem function; doing so is complicated by incomplete understanding the underlying causal relationship [320], [344], [345]. Consideration of temporospatial variation is also limited, and ecosystem wellbeing needs are not explicitly described. Consistent with other publications, limited data is cited as an important barrier to the work [341], [342], [346], [347]. The need for ongoing interdisciplinary collaboration – including efficient transference of knowledge – is also emphasised. Rugani et al.'s work prompted calls for more comprehensive and context-specific approaches to assess biodiversity impacts in LCA, to which subsequent publications have contributed [348], [349], [350], [351], [352].

2.4.4.1 Land cover and plant diversity monitoring for biodiversity impact assessment

Remote sensing represents an important advancement in the monitoring and quantification of species diversity across myriad applications, including LCA [353]. Measures of the underlying habitat and flora provides important insight into the composition, quality, and functionality of different ecological (and anthropogenic) systems. Examples include land cover (habitat) type, topography, heterogeneity, and connectivity, as well as plant diversity and bio-productivity. When measuring and monitoring species diversity, indicator species are often used. These may be functionally important (e.g., 'keystone') species or tractable gena, whose presence or absence indicate the presence or absence of other species [354]. Of particular interest here are land intensity and habitat fragmentation, which govern the dynamics and characteristics (functionality, resilience, longevity) which themselves determine the inherent and functional values of ecological systems [355]. Remote sensing tools are at the fore of land cover and plant diversity monitoring. Monitoring terrestrial plant biodiversity involves assessing the abundance, distribution, and health of plant species within specific ecosystems over time. In IMPACTWorld+, these are described at the ecoregion level [19], [356].

Remote sensing techniques, such as hyperspectral satellite imagery and aerial photography, record data over large spatial scales [357]. These techniques can detect changes in vegetation cover, species composition, and habitat fragmentation over time without the need for manual fieldwork (e.g., field survey: in-person observation within a defined area). By monitoring plant biomass and bio-productivity provides insights into ecosystem quality and functioning over time, in relation to different natural and anthropogenic pressures [358]. Indicator species serve as indicators of habitat quality, environmental changes, or ecosystem disturbances. By focussing on changes in the abundance or distribution of these species, ecosystem quality can be tracked. Ecosystem dynamics can also be inferred. However, remote sensing is incapable of recording species data at high granularity. Ground-level observation is obstructed by vegetation at the canopy level. That notwithstanding, data is often restricted to > $1m^2$ $pixel^{-1}$ resolution [refs], preventing fully comprehensive (to the species level) biodiversity monitoring. 30 cm^2 $pixel^{-1}$ is possible through WoldView series satellites [359], but lower resolution monitoring (e.g., Sentinel series: ~30 m^2 $pixel^{-1}$ [360]) is more practical, more cost-effective, and therefore more common.

Finally, it is important to collect temporally and spatially diverse data. Time-series offer insights unavailable in isolated (static) images. Here, long-term monitoring networks such as the National Ecological Observatory Network (NEON) are valuable [361]. These facilitate large-scale and long-term monitoring across different sites. However, long-term, and large-scale collaboration is not without its challenges. Establishing a global remote biodiversity monitoring system has been proposed, but its realisation has proven challenging. While remote sensing is here at the fore, technological and social barriers must still be overcome.

2.4.5 Metabolic theory of ecology for dynamic and long-term life cycle biodiversity impact assessment

Metabolism governs the flow of energy, nutrients, and resources through ecosystems, ultimately shaping their dynamics and functioning [362], [363]. The ‘metabolic theory of ecology’ (MTE) aims to explain ecological patterns and processes based on the fundamental principles of metabolism, building upon metabolic scaling theory: the observation that rates of biological processes, such as growth, reproduction, and mortality, scale with the metabolic rate of organisms [364], [365], [366], [367]. These scaling laws (or allometries) form the basis of this thesis in a similar way given they enable physiologies to be estimated from morphologies, such as body mass, as detailed in Chapter 3 [26]. By integrating metabolic scaling with mechanistic models of ecological dynamics, the metabolic theory of ecology considers how organisms interact with their environments and the components that comprise them, how trophic systems are structured, and how ecosystems function under different conditions and

configurations. Through understanding the dynamics of ecological systems, the potential impacts of environmental and socioeconomic pressures on species diversity and ecosystem function can be better understood.

MTE and LCIA contribute to our understanding of ecological and environmental systems but operate at different levels of analysis; they have distinct focuses and make use of different data. These reflect the different origins of MTE and LICA: ecological and engineering, respectively. Temporal inconsistencies make integrating MTE into LCIA difficult [368]. MTE is dynamic, describing ecological processes over time, whereas LCIA is static, representing an ecological assessment within rigid temporospatial parameters. LCIA typically operates at temporal scales corresponding to human lifetimes and focuses on human activities or processes, often across small spatial scales, and the potential impacts of different activities on (i) human health, (ii) ecosystem quality, and (iii) resources. A variety of environmental indicators are used. In contrast, MTE seeks to understand the causal relationships between (organismal) metabolism and ecological processes. Metabolism is therefore the focus but considered in greater detail than LCIA. MTE can operate at much longer timeframes – those related to evolutionary processes for example – and spatial scales encompassing ecological processes.

MTE relies on empirical data, mathematical models, and mechanistic frameworks derived from physiology, ecology, and biophysics. These represent considerable data which is in many cases – invertebrates for example – lacking [369], [370]. This simplification, though widespread in ecological modelling, has drawn criticism. The need for comprehensive temporospatial ecological data represents a considerable barrier to developing robust empirical models through MTE, and the intractability of ecological data makes it difficult to model all ecological variables at play. To overcome data limitations and to make the modelling of complex ecological systems more manageable, MTE relies on simplifying assumptions including the extrapolation of metabolic relationships across different scales [369]. The oversimplification of complex ecological systems limits the accuracy of model predictions. There is therefore a need to exercise caution when applying MTE results, for example: to quantify long-term impacts on biodiversity and ecosystem function in the context of LCIA.

MTE offers a valuable mechanistic understanding that is currently lacking – and called for – in LCIA [80], [341]. Like the work in this thesis, including MTE in LCIA or using MTE alongside LCIA can helps quantify the potential impacts of anthropogenic activities on ecosystem quality. Unlike the work in this thesis, MTE can also be used to quantify the potential impacts of anthropogenic activities on ecosystem function. This includes information on the stability of trophic systems and thus the

intergenerational security of ecosystem functions in the long-term. MTE could be used within or alongside LCIA to assess more confidently the life cycle impacts of different operational or legislative scenarios across both socioeconomic and ecological systems. Such comprehensive, ecological insight in LCIA would be of considerable value to supporting biodiversity alongside human development. MTE therefore also represents rich opportunity to advance the work proposed in this thesis, with the two concepts – ecological needs and MTE – converging to produce dynamic, empirical models of ecological metabolism that describe resource use associated with all physiological needs of organisms under different, multifactorial (temporal, spatial, environmental) conditions.

2.5 Conclusions

Ecological components are increasingly used in urban design to support urban biodiversity alongside meeting human needs, but a poor quantitative understanding of species needs undermines conservation urban conservation. The topic of urban biodiversity and urban design for the purpose of urban conservation represents an area of active research, motivated by both socioeconomic and ecological benefits. A critical application of ecological components in the urban context is as biodiversity-supporting infrastructure. With human needs the foremost motivating factor in urban design, biodiversity-supporting infrastructure often serves to support human wellbeing directly (e.g., through human-nature interactions) and indirectly (e.g., by improving the urban resilience and liveability). In contemporary urban design, ecological components are commonly employed as nature-based solutions; an important example being green infrastructure comprising vegetation, e.g., green roofs and walls. Here, ecological components are used to support ecological systems for their provision of ecosystem services that socioeconomic systems have come to rely on.

The design and assessment of biodiversity-supporting infrastructures is undermined by the fact the (physiological and social) wellbeing needs of ecological systems is not quantitatively understood. There is a reliance on qualitative understanding of species needs in the design of biodiversity-supporting infrastructure. Quantitative data is often limited to individual species, with physiology at the system level poorly quantified. There is a focus on mammals, birds, and charismatic species; reptile and invertebrate data is lacking. Many organisms and contexts (locations: climates, habitats, trophic compositions) are also underrepresented, with the global north and biodiversity hot-spots most often described. This knowledge gap demonstrates a considerable hindrance to designing for urban conservation. Until the needs of different species and different ecological systems is known, it is unclear how best to design urban areas to support biodiversity either for its inherent value or to stimulate specific

functionalities. There is therefore a critical need to quantify the physiological and other wellbeing needs of species comprising ecological systems, analogous to human wellbeing.

In addition to undermining the design biodiversity-supporting infrastructure, a poor quantitative understanding of ecological needs also hinders their life cycle assessment. This is an area of ongoing research. An important aspect for urban design is the need to characterise local biodiversity impacts due to land cover change more comprehensively. T the potential impacts of land cover change on local biodiversity are currently obscured in LCA. E. Alejandre et al. (2022) represents a singular example of land cover being causally linked to species richness in LCA whereby bio-productivity and hence provision is related to different scenarios of pollinator diversity. However, the metabolic requirements of species are not explicitly considered in the publication. Its scope is also limited as local biodiversity impacts are not characterised at the system-level. Moreover, there is also a need to change the way sources of local biodiversity impacts are inventoried in LCA (Fig. 2.4). Current practice fails to accurately capture all potential local biodiversity impacts relating to land/surface cover. Local biodiversity impacts are attributed to land transformation and land occupation and inventoried as areas of land cover change at the midpoint. However, other sources of local biodiversity impacts, i.e., biodiversity-supporting infrastructures and components, represent more than 'land cover' (e.g., vertical surfaces, nest boxes), constituting a new environmental pressure in the life cycle inventory stage. This is discussed in Chapter 4 as well as Chapter 7, 'Recommendations for Further Work'.

Quantifying ecological needs is an enormous and complicated undertaking, but one that would facilitate major interdisciplinary advancement. This includes improving our understanding of the potential biodiversity impacts of human intervention, with benefits far beyond urban design, and advancements in resource management (e.g., to facilitate more comprehensive material flow analyses [12]), policy-making (e.g., to set aside land and resources for ecological use, in similar vain to the 'half earth' hypothesis [371]), and environmental conservation (e.g., more informed habitat suitability assessments preceding species translocation, e.g., [372]). If ecological needs were understood as well as human (socioeconomic) needs, it would be possible to understand what sustainable human development (alongside adequate environmental conservation) entails and what global actions are needed to achieve it.

3 Quantifying the natural resource requirements of terrestrial animals (or 'ecological needs') analogous to human wellbeing needs

This chapter is based on the manuscript, Mason, A. R.; Gathorne-Hardy, A.; White, C.; Plancherel, Y.; Woods, J.; Myers, R. J. (2022). Resource requirements for ecosystem conservation: A combined industrial and natural ecology approach to quantifying natural resource use in nature. *Ecology and Evolution,* 12 (e9132).

3.1 Introduction

The field of industrial ecology was developed to understand, analyse, and assess the environmental impacts of socioeconomic (human) actions [1]. Within industrial ecology, natural resource use is empirically modelled to systematically and quantitatively analyse the environmental impacts associated with human development [7], and what effects human development will have on the quality and availability of natural resource in the future [8], [9]. However, models of natural resource use are limited to socioeconomic analyses; they lack a holistic perspective and do not yet capture the natural resource demands of organisms (plants, animals, fungi, etc.), which make use of the same natural resources simultaneously, within ecological systems.

At the most fundamental level, human development requires the use of natural resource to satisfy certain 'human wellbeing needs'. These are a set of essential (e.g., shelter, food, and water) and non-essential (e.g., access to technology) social, economic, and physiological requirements for good physical and mental health [16], [17]. A quantitative understanding of natural resource use in socioeconomic systems allows it to be empirically modelled [18] and the associated environmental impacts assessed [19]. Within the last two decades, a quantitative set of human wellbeing needs has also been developed [20], [21]. This has improved our ability to model and assess socioeconomic natural resource use associated with human development. It is now possible to infer the minimum level of natural resource use needed to support socioeconomic systems [22], and assess the environmental impacts associated with achieving contemporary targets for human development – an important example being the Sustainable Development Goals (SDGs) [23].

All living organisms (plants, animals, fungi, etc.) demonstrate physiological needs. Like humans, they require water, energy, and nutrients, and in some cases – namely plant and animal species – they also

demonstrate social needs [24], [25]. These requirements constitute the 'wellbeing needs' of individual plant and animals that make up ecosystems, analogous to human wellbeing needs. However, it is not yet quantitatively understood what natural resource is needed to support individual plants and animals, and by extension biodiverse populations of species in ecosystems; nor do we know what natural resource must be allocated to ecosystems to protect essential ecosystem services on which contemporary socioeconomic systems depend.

This knowledge gap is significant in the context of natural resource management as where socioeconomic- and ecological systems make use of the same, finite stocks of natural resource to satisfy their wellbeing needs – namely land area, water, and energy (from food, etc.) [30], [31], [32] – the two systems must compete. Intuitively, the rate of natural resource production cannot be exceeded. Hence, where the combined rate of natural resource use across socioeconomic- and ecological systems exceeds the rate of natural resource production, a trade-off is necessary. This manifests as a compromise between satisfying human and ecological wellbeing needs. Biased intentions to satisfy human wellbeing needs in the present may therefore, inadvertently, limit our ability to satisfy those same human wellbeing needs in the future, if ecological wellbeing needs are sufficiently deprived (see 1.4) [35].

This chapter describes a novel concept and accompanying method to quantify the natural resource needed to support terrestrial species within ecosystems, analogous to how natural resource use by humans is quantified in a socioeconomic context. The method applies allometric equations relating the adult body mass of terrestrial species to their requirements for land area, water, and energy, to describe the natural resource required to support different levels of biodiversity within an ecosystem. A case study site in Scotland, UK, is used to apply and demonstrate the method at this initial stage. The method conceptualises, for the first time, a comprehensive 'dual-system' approach: modelling natural resource use in socioeconomic- and ecological systems simultaneously. It can facilitate the management of natural resource at the global scale, and in both the conservation and creation (e.g., rewilding) of biodiversity within managed ecosystems. It also represents an advancement in determining what socioeconomic compromises are needed to achieve contemporary conservation targets alongside ongoing human development.

3.1.1 Conceptual framework for quantifying ecological needs

Three variables are needed to assess the sustainability of natural resource use in socioeconomic systems: natural resource supply (stocks, rate of production), socioeconomic demand, and ecosystem demand. As ecosystem demand is not known, the tools we currently use to model natural resource use (e.g., material flow analyses [11]) and assess the sustainability of its use (e.g., the ecological footprint [373])

are not yet comprehensive. By extension, current targets regarding sustainable human development may be flawed. For example, we can quantify what natural resource is needed to satisfy the socioeconomic SDGs [22]. However, we do not know what natural resource is needed to satisfy the ecological SDGs – those that relate to ecological conservation (namely goals 13, 14, and 15). This means that we are currently unable to determine whether the social, economic, and the ecological SDGs can be achieved concurrently, and whether enough natural resource remains once the socioeconomic SDGs are met to satisfy the ecological SDGs. Furthermore, we do not know whether the ecological SDGs, even if they were to be achieved in their entirety, adequately satisfy ecological wellbeing needs underlying critical ecosystem functionality.

A comprehensive, systems approach to natural resource management is needed to determine what environmental conservation can be achieved alongside contemporary human development goals. Fig. 3.1 illustrates how natural resource production, and human and ecological wellbeing change in response to different distributions of (constant) natural resource stocks – here distributed between a socioeconomic system and an ecosystem – corresponding to four different scenarios of socioeconomic development (S1, Low human development, high ecosystem conservation; S2, moderate human development, moderate ecosystem conservation, S3, high human development, low ecosystem conservation; S4, collapse of the socioeconomic- and ecological system). Fig. 3.1a shows increasing socioeconomic development (corresponding to increasing natural resource use) from S1 to S4, alongside decreasing biodiversity (as less natural resource is available for ecosystem use). In turn, decreasing biodiversity drives a reduction in natural resource production and hence its availability. As natural resource availability decreases, individuals in both the socioeconomic- and ecological system are less able to satisfy their wellbeing needs (and vice versa). This deprivation results in the barren landscape illustrated in S4.

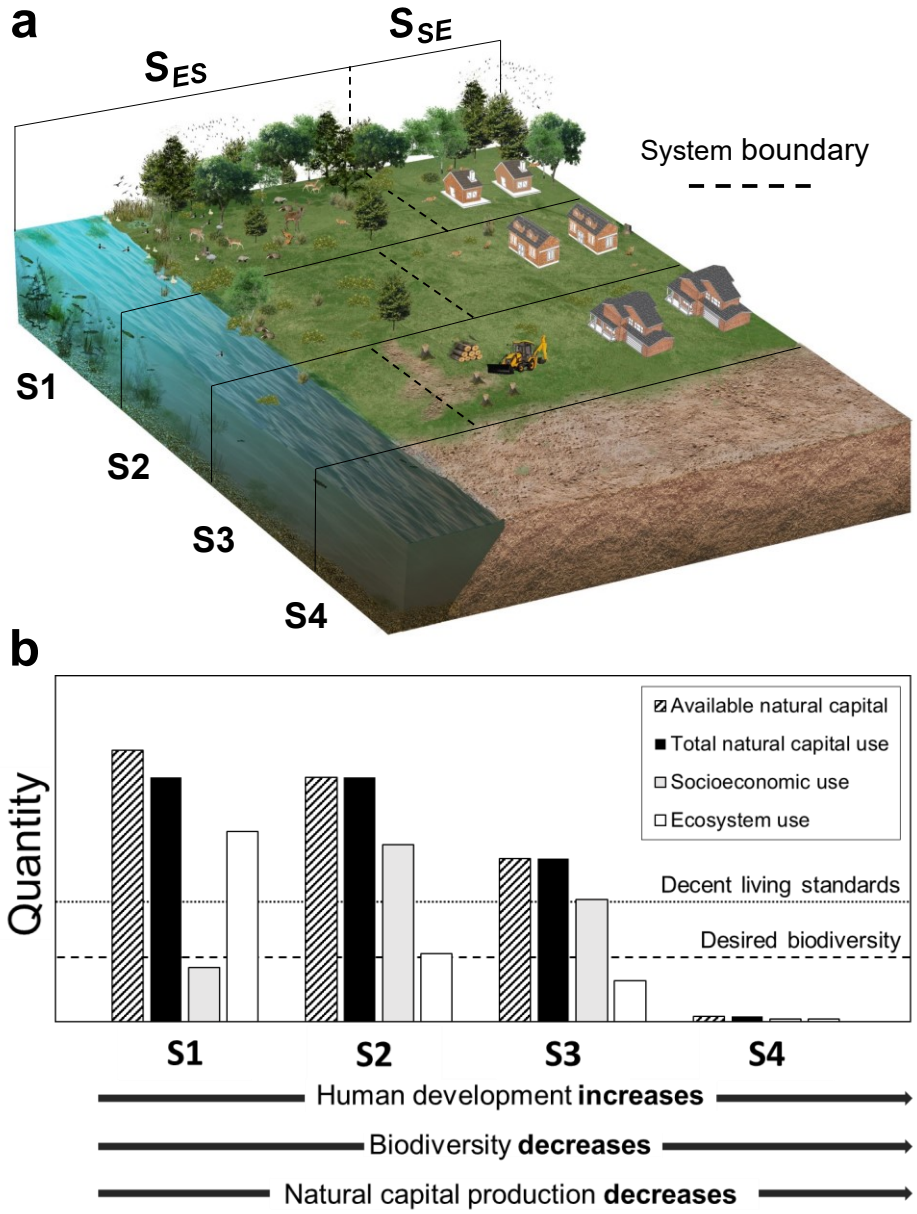


Figure 3.1 (**a**) Schematic showing the changing ability to satisfy socioeconomic (human) and ecological wellbeing needs concurrently, in response to four different scenarios of socioeconomic development, corresponding to different distributions of natural resource stocks. S_{ES} is the ecosystem share; and S_{SE} is the socioeconomic share. Scenario 1 (S1), high biodiversity and low human development, $S_{ES} > S_{SE}$; Scenario 2 (S2), moderate biodiversity and moderate human development, $S_{ES} < S_{SE}$; Scenario 3 (S3), low biodiversity and high human development, $S_{ES} << S_{SE}$; and Scenario 4 (S4), dual-system collapse. (**b**) We present qualitatively the results of quantifying 'Ecosystem use' using our method and combining this with data describing 'Socioeconomic use' across the four scenarios. In (**b**), the dotted line labelled 'Decent living standards' indicates the point at which the socioeconomic population achieves decent living standards. The dashed line labelled 'Desired biodiversity' indicates the natural resource demand associated with a desired minimum ecosystem population. For illustrative purposes, 'Desired biodiversity' is here the minimum level of biodiversity needed to produce enough natural resource to sustain the socioeconomic population.

Fig. 3.1b demonstrates qualitatively the application of our conceptual framework, showing the amount of natural resource allocated between the socioeconomic- and ecological systems. Here, the natural resource allocated to the socioeconomic- and ecological systems contribute to human development (e.g., to achieve a decent standard of living [374]) and ecosystem conservation respectively. This example also uses a constant socioeconomic population across S1-S3. In Fig. 3.1b, the four scenarios (S1-S4) describe the following:

- **S1**: Low human development alongside high levels of biodiversity. A large amount of biodiversity is supported owing to abundant natural resource production, but insufficient human wellbeing needs are met. Decent living standards are therefore not achieved across the socioeconomic population.

- **S2**: Human development increases, degrading the ecosystem. The provision of natural resource decreases. Much less biodiversity is supported, but this remains above the desired level. Sufficient human wellbeing needs are now met; decent living standards are achieved across the socioeconomic population. This scenario of natural resource management represents a sustainable outcome, where human development and conservation targets are met simultaneously.

- **S3**: Human development and ecosystem degradation increases further. The provision of natural resource is further decreased. Sufficient human wellbeing needs continue to be met; decent living standards are achieved across the socioeconomic population. However, the desired level of biodiversity can no longer be supported; there is insufficient natural resource to satisfy socioeconomic- and ecological system wellbeing needs concurrently.

- **S4**: A worst-case scenario: ecological needs continued to be deprived. The provision of natural resource is now insufficient to support either system. This scenario is unsustainable; socioeconomic- and ecological system needs cannot be met, and the systems collapse.

S3 is a critical stage; it is a 'tipping point', where satisfying human wellbeing needs diminishes our intergenerational needs (as illustrated in Fig. 1.4, Chapter 1). Importantly, our ability to identify this tipping point in natural resource management is impaired by our current inability to quantify what natural resource is needed to support certain levels of biodiversity in ecosystems. As such, we cannot accurately distinguish between sustainable and unsustainable scenarios of natural resource management until both socioeconomic- and ecological system demands are quantitatively described.

As the only species capable of safeguarding the biophysical environment, it is the responsibility of humans to consider the intergenerational needs of all species, not just our own. Comprehensive models of natural resource use, describing both socioeconomic- and ecological systems demand, are needed to assess and manage the trade-offs triggered by human development and limited natural resource availability [1]. Especially where the impacts of socioeconomic use threaten to reduce the production and hence the availability of natural resource. However, there is at present a fundamental knowledge gap that impairs our ability develop such models. Specifically, data on what natural resource is needed to preserve and/or improve the life-supporting functionality of ecosystems while achieving ongoing human development in socioeconomic systems.

3.1.2 Morphology-physiology relationships in the literature

Ecologists have long studied the relationship between morphology (e.g., body mass, body length, height) and physiology (e.g., metabolic rate, population density, water consumption) in animals and plants. These data are described extensively in the ecological literature and plethora correlations between morphology and physiology have been demonstrated. Equations describing these correlations are termed 'allometric equations' [159]. In his seminal work [164], McNab demonstrated the relationship between body mass and home range in mammals, suggesting that diet and metabolic rate are also important factors in an individual's land area use. This was later confirmed by Damuth [375], and expanded upon: variations in physiology are related to trophic level and body mass ranges [165], [166], [167], [168], [169] too. McNab's limited scope has been expanded beyond mammals, to include birds, reptiles, and invertebrates (to a lesser extent), with allometric equations for plants (e.g., water uptake, energetics) also described [170], [171].

Alternative means of predicting the land area use, metabolic requirements, and water consumption of species exist. These include species-area curves [376], [377], which are used to predict biodiversity (number of species) losses due to land area use changes, and modelling software that predicts species' response to environmental pressures [378]; and metabolic rate predictions using heartrate measurements [379], and oxygen consumption [380]. Alternative means of predicting water consumption appear limited to agricultural contexts. Furthermore, these take the form of single-species mathematical approaches [381], [382], ultimately equivalent to allometric expressions. While alternatives exist, allometric equations remain a prominent means of predicting species physiology given their simple development/application, and the abundance of prerequisite data.

Information on natural resource use or production can be inferred where allometric equations are applied in the ecological literature. However, their application appears limited to ecological and

agricultural literature. Examples include predicting biomass production (e.g., roots, timber, vegetation) across different land-cover types [383]; predicting the physiologies of undocumented species [384]; and for design [385] and husbandry in an agricultural context. With adjustment, and used concurrently, allometric equations can be incorporated into empirical models dealing with natural resource management and urban development. Despite its inherent benefits, this transference has not been realised. To overcome this, we build upon existing work on allometric equations and develop a generalisable method to quantify the use of natural resource in ecosystems, analogous to established methods to quantify human wellbeing needs [22], quantifying the natural resource cost associated with environmental conservation concurrent to that necessitated by human development.

3.2 Method

3.2.1 Land use

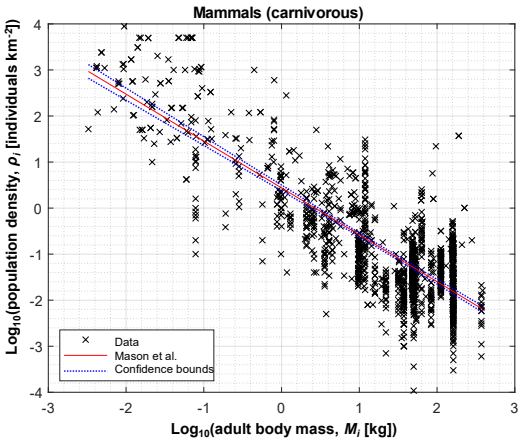
We develop a new set of allometric equations that relate the adult body mass (M_i, kg) of individual mammals, birds, reptiles, and invertebrates, to their land area use (L_i, km^2 individual^{-1}) at the species level. The development of these equations improves upon existing work in the literature; we aggregate data corresponding to existing allometric equations for land area use – e.g., those published by Damuth (1987), Silva & Downing (1995), Stephen et al. (2019), Robinson & Redford (1986), and supplement these data with average adult body mass and trophic level data. We also incorporate the substantial datasets TetraDensity and PanTHERIA [211], [212], which describe the population density (ρ_i, individuals km^{-2}) of mammal and bird species. These datasets were not captured in the preceding allometric equations for land area use since they were compiled after their publication of those preceding equations. The full, supplemented data used are presented in Tables A7-A14, in Appendix 4.

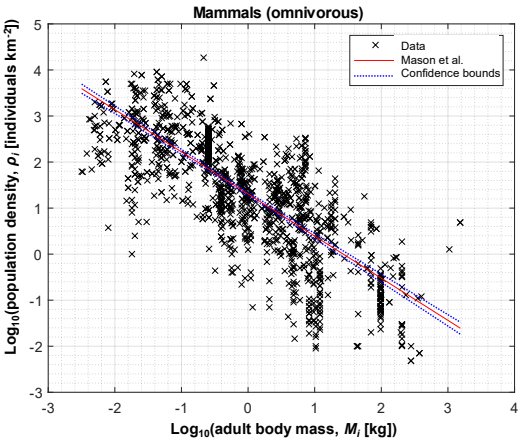
The reciprocal of population density (i.e., $(\rho_i)^{-1}$, km^2 individual^{-1}) is equivalent to land area use [167], where population density describes the land area use of an individual as part of a wider single-species population [386]. It is important to capture land area use in this way because land area is not a single use resource in ecosystems, nor is it used by individuals in solitude [209]. The concept of population density is therefore preferable to other measures of land area use, such as home range, which employ an individual-species approach to quantifying land area use [167].

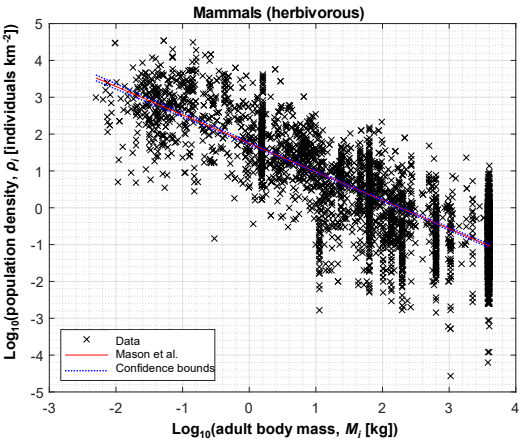
The allometric equations for land area proposed in this paper were developed through linear regression analysis. The 'fitlm' linear regression model [387] in the MATLAB computing environment was used to produce linear equations describing the relationship, '$\log_{10}$(adult body mass, kg) versus

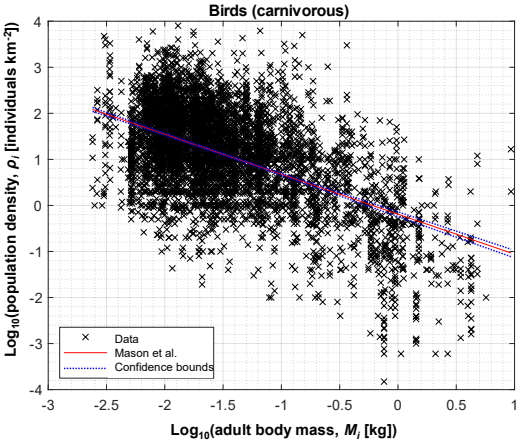
$\log_{10}$(population density, individuals km^{-2})', for terrestrial mammals, birds, reptiles, and invertebrates. A substantial dataset consisting of over 17,000 data points was used to perform these analyses, which are present alongside corresponding adult body mass data and trophic level data in Tables A7-A14, Appendix 4.

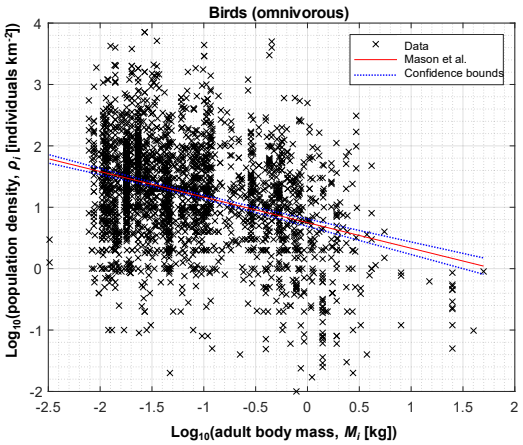
Previous studies demonstrate that mammal and bird species exhibit different adult body mass-land area use relationships across the different trophic levels [213], [214]. Therefore, we disaggregated population density data by trophic level (i.e., herbivore, omnivore, and carnivore) for mammals and birds, and performed regression analyses for each category case. Therefore, the population density data for mammal and bird species was disaggregated by trophic level. In contrast, the regression analyses for reptile and invertebrate species were performed across all trophic levels. This is due to limited data at the species level in both cases. The results of the linear regression analyses are presented in Fig. 3.2. The coefficients calculated for each linear regression are presented in Table A1 and Table A2, Appendix 1.

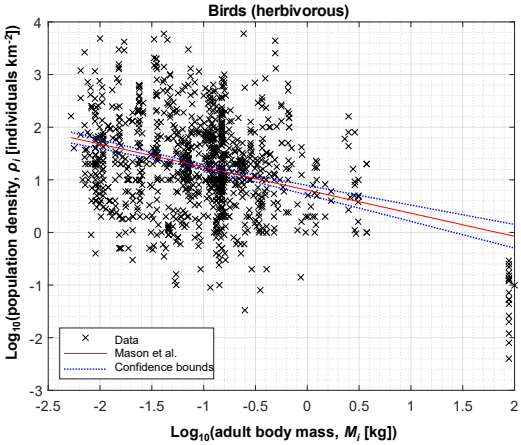

Mammals (carnivorous)
Log10(population density, ρi [individuals km-2])
Log10(adult body mass, Mi [kg])
Data
Mason et al.
Confidence bounds

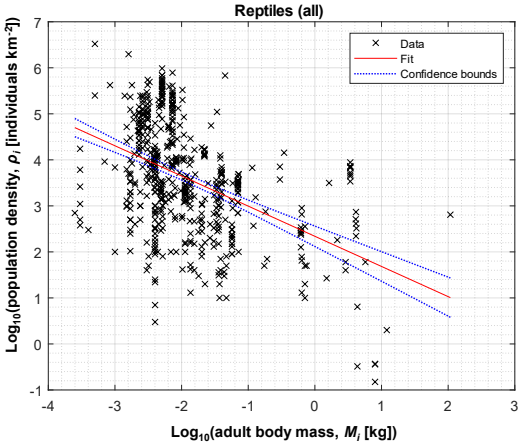

Mammals (omnivorous)
Log10(population density, ρi [individuals km-2])
Log10(adult body mass, Mi [kg])
Data
Mason et al.
Confidence bounds


Mammals (herbivorous)
Log10(population density, ρi [individuals km-2])
Log10(adult body mass, Mi [kg])
Data
Mason et al.
Confidence bounds


Birds (carnivorous)
Log10(population density, ρi [individuals km-2])
Log10(adult body mass, Mi [kg])
Data
Mason et al.
Confidence bounds


Birds (omnivorous)
Log10(population density, ρi [individuals km-2])
Log10(adult body mass, Mi [kg])
Data
Mason et al.
Confidence bounds


Birds (herbivorous)
Log10(population density, ρi [individuals km-2])
Log10(adult body mass, Mi [kg])
Data
Mason et al.
Confidence bounds


Reptiles (all)
Log10(population density, ρi [individuals km-2])
Log10(adult body mass, Mi [kg])
Data
Fit
Confidence bounds

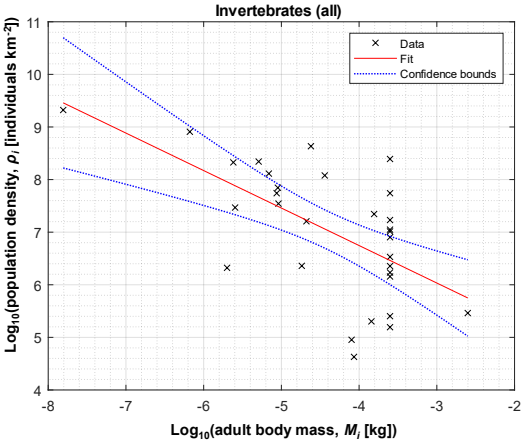

Invertebrates (all)
Log10(population density, ρi [individuals km-2])
Log10(adult body mass, Mi [kg])
Data
Fit
Confidence bounds

Figure 3.2 Plots showing the fits of our proposed allometric equations for land area use (red line, foreground; see Table 4.1), and the 95% confidence intervals for each line of best fit, for mammals (carnivorous, omnivorous, herbivorous), birds (carnivorous, omnivorous, herbivorous), reptiles, and invertebrates. The full data used is presented in Tables A7-A14, Appendix 4.

In Fig. 3.2, the equations describing each linear line of best fit (red lines) take the following form (Eq. (3.1)):

$$Log_{10}(\rho_i) = (b_i \pm e_{b,i}) \times Log_{10}(M_i) + Log_{10}(a_i \pm e_{a,i}) \quad (3.1)$$

Where ρ_i (individuals km^{-2}) is the population density of an individual (subscript i), L_i (km^2 individual^{-1}) is its reciprocal, the land area use of an individual; M_i (kg) is the adult body mass of the corresponding species; a_i and b_i are the intercept and the gradient (respectively) of the line of best fit – biological class-specific parameters derived through regression analyses; and $e_{a,i}$ *and* $e_{b,i}$ are the standard error terms for a_i and b_i respectively. a_i, b_i, $e_{a,i}$ *and* $e_{b,i}$ are presented in Tables A1-A2, Appendix 1.

The allometric equations for land area use (Eq. (3.2)) are converted into the standard allometric equation form (Eq. (3.5)) as follows:

$$Log_{10}(\rho_i) = Log_{10}(M_i^{b_i}) + Log_{10}(a_i) \quad (3.2)$$

$$Log_{10}(\rho_i) = Log_{10}(a_i \times M_i^{b_i}) \quad (3.3)$$

$$\rho_i = a_i \times M_i^{b_i} \quad (3.4)$$

$$L_i = \rho_i^{-1} = (a_i \times M_i^{b_i})^{-1} \quad (3.5)$$

Where L_i is the land area use of an individual (km^2 individual^{-1}), the inverse of that individual's population density (ρ_i).

3.2.2 Rates of water consumption

Terrestrial species either satisfy their water needs through active means, whereby water is ingested (i.e., drunk, or acquired through the diet); or through passive means, whereby water is absorbed directly from

the atmosphere or produced metabolically (i.e., as a by-product of respiration) [388], [389]. Here, 'water use' is used to describe active water intake excluding that from the diet. This is because we are interested in the interaction between animal species and bodies of water that humans also utilise; metabolic water cannot be appropriated. We assume that dietary water requirements are satisfied when the energy requirements of an individual are met.

Ecological studies (e.g., [185]) have linked the water use of individual mammals and birds to their adult body masses. Allometric equations have subsequently been developed for wild and captive mammals (Eq. (3.6)) and birds (Eq. (3.7)):

$$W_{i,mammals} = 9.9 \times 10^{-5} (M_i)^{0.9} \tag{3.6}$$

$$W_{i,birds} = 5.9 \times 10^{-5} (M_i)^{0.67} \tag{3.7}$$

Where $W_{i,mammals}$ and $W_{i,birds}$ (m^3 $individual^{-1}$ day^{-1}) are terms quantifying the water use for an individual (subscript i) mammal and bird respectively, and M_i (kg) is the adult body mass of the corresponding species. These equations have conventionally been used in an animal husbandry context to predict the water use needs of livestock [390].

While the mechanism of drinking has been studied in some reptile species (e.g., [184]), the drinking habits of invertebrates and reptile species are not as well understood as those of mammals and birds [186]. Here we assume that invertebrates species satisfy their need for water entirely through their diet, through metabolic means, and by absorbing water from the atmosphere [388], [391], [392]. Similarly, we assume that reptiles satisfy their need for water entirely through their diets and through metabolic means [393]. Therefore, allometric equations describing water use by individual invertebrates and reptiles are not presented in Table 3.1.

3.2.3 Energy requirements

All living organisms require energy, in one form or another, to function. Terrestrial animal species are heterotrophic, they acquire energy from the food they eat [394]. Once eaten, this food is metabolised and used, for example, to regulate body temperature [395]. The concept of 'basal metabolic rate' (BMR), the rate at which energy is expended while an individual is at rest [395], [396], is often used when describing the metabolism of animals [395]. However, animals do not exist at rest. All animals

must search for natural resource and/or interact with other individuals to satisfy their wellbeing needs. A concept called the field metabolic rate (FMR) accounts for this additional exertion [397]. Therefore, we consider FMR to be a more appropriate concept for capturing energy use by terrestrial species. By describing the essential intake of food in terms of energy, this wellbeing need can be compared across different taxonomic ranks and trophic levels (e.g., herbivore, carnivore, omnivore).

Allometric equations relating FMR to adult body mass for individual mammals, birds and reptiles have been proposed with the general form shown in Eq. (3.8) [224]:

$$E_i = a_i \times (M_i)^{b_i} \tag{3.8}$$

Where E_i (kJ individual^{-1} day^{-1}) is the energy use of an individual (subscript *i*) mammal, bird, or reptile, M_i (kg) is the adult body mass of the corresponding species, and a_i and b_i are biological class-specific parameters (see Table A2, Appendix 1). Accounting for the inherent differences in metabolism across different trophic levels, we present allometric equations describing energy use for individual mammals (herbivores, Eq. (3.9); omnivores, Eq. (3.10); carnivores Eq. (3.11)), birds (herbivores, Eq. (3.12); omnivores, Eq. (3.13); carnivores, Eq. (3.14)), and reptiles (Eq. (3.15)):

$$E_{i,mammals,herbivore} = 688.4 \times (M_i)^{0.646} \tag{3.9}$$

$$E_{i,mammals,omnivore} = 652.1 \times (M_i)^{0.678} \tag{3.10}$$

$$E_{i,mammals,carnivore} = 791.2 \times (M_i)^{0.85} \tag{3.11}$$

$$E_{i,birds,herbivore} = 1{,}159.3 \times (M_i)^{0.681} \tag{3.12}$$

$$E_{i,birds,omnivore} = 716.6 \times (M_i)^{0.628} \tag{3.13}$$

$$E_{i,birds,carnivore} = 1264 \times (M_i)^{0.705} \tag{3.13}$$

$$E_{i,reptiles} = 91.0 \times (M_i)^{0.889} \tag{3.15}$$

An allometric equation describing the energy use of individual invertebrates has previously been proposed by Ballesteros et al. (2018) [217] (Eq. (3.16)), which follows the same form as described in Eq. (3.8).

$$E_{i,invertebrates} = 526 \times (M_i)^{0.832} \tag{3.16}$$

However, Eq. (3.16) describes energy use at the BMR, hence it does not describe the energy required to satisfy an individual's wellbeing needs. Transformation of this equation to describe FMR is important to ensure its consistency with the equivalent allometric equations for mammals (Eqs (3.9) – (3.11)), birds (Eqs (3.12) – (3.14)), and reptiles (Eq.(3.15)). Eq. (3.16) is transformed based on the findings that $FMR = 32*BMR$, where $M_{i,invertebrates} < 1x10^{-5}$ kg; and $FMR = 8*BMR$, where $M_{i,invertebrates} \geq 1x10^{-5}$ kg. This produces Eqs (3.17) – (3.18), which are now appropriate for quantifying the energy use associated with wellbeing in individual invertebrates.

$$E_{i,invertebrates} = 16,832(M_i)^{0.832} \quad (3.17)$$

Where $M_i < 1x10^{-5}$ kg

$$E_{i,invertebrates} = 4,208(M_i)^{0.832} \quad (3.18)$$

Where $M_i \geq 1x10^{-5}$ kg

The assumptions that $FMR = 8*BMR$ and $FMR = 32*BMR$ are applied here as it has been shown that BMR scales with FMR for invertebrates in a similar way to mammals and birds. For small mammals and birds, BMR is shown to scale to FMR according to $FMR = 2*BMR$ [398]. For invertebrates, FMR is shown to be 32-times greater than BMR in small invertebrates ($< 1x10^{-6}$ kg), and eight-times greater in large invertebrates ($\geq 1x10^{-6}$ kg) [399].

3.3 Results

3.3.1 Proposed equations for quantifying ecological wellbeing needs for terrestrial mammals, birds, reptiles, and invertebrates

Table 3.1 presents a set of allometric equations for quantifying the natural resource use of terrestrial mammals, birds, reptiles, and invertebrates, in ecological systems analogous to human wellbeing needs in socioeconomic systems [123]. These equations relate the average adult body mass of individual terrestrial mammals, birds, reptiles, and invertebrates to their land area use (L_i, km^2 individual^{-1}), water use (W_i, m^3 individual^{-1} day^{-1}), and energy use, i.e., from food or sunlight, (E_i, kJ individual^{-1} day^{-1}).

Table 3.1 Proposed ecological needs for terrestrial animals, describing the land area use, L_i (km^2 individual^{-1}), energy requirements, E_i (kJ individual^{-1} day^{-1}), and rates of water consumption, W_i (m^3 individual^{-1} day^{-1}), of mammals, birds, reptiles, and invertebrates. The subscript 'i' denotes an individual, e.g., one red fox (*Vulpes vulpes*) – an omnivorous mammal. M_i (kg) is the adult body mass of an individual of species i.

Biological class	Land area L_i (km^2 individual^{-1} day^{-1})		Energy requirement E_i (kJ individual^{-1} day^{-1})		Water consumption W_i (m^3 individual^{-1} day^{-1})
Mammals	Carnivores	$(2.74(M_i)^{-1.018})^{-1}$	Carnivores	$791.2(M_i)^{0.85}$	$9.9\text{x}10^{-5}\text{x}(M_i)^{0.90}$
	Herbivores	$(55.91(M_i)^{-0.773})^{-1}$	Herbivores	$688.4(M_i)^{0.646}$	
	Omnivores	$(20.28(M_i)^{-0.915})^{-1}$	Omnivores	$652.1(M_i)^{0.678}$	
Birds	Carnivores	$(0.65(M_i)^{-0.868})^{-1}$	Carnivores	$1{,}264(M_i)^{0.705}$	$5.9\text{x}10^{-5}\text{x}(M_i)^{0.67}$
	Herbivores	$(6.36(M_i)^{-0.438})^{-1}$	Herbivores	$1{,}159.3(M_i)^{0.681}$	
	Omnivores	$(5.61(M_i)^{-0.416})^{-1}$	Omnivores	$716.6(M_i)^{0.628}$	
Reptiles	$(218.3(M_i)^{-0.656})^{-1}$		$91.0(M_i)^{0.89}$		No data
Invertebrates	$(7{,}852(M_i)^{-0.713})^{-1}$		$16{,}832(M_i)^{0.832\ *}$, $4{,}208(M_i)^{0.832\ **}$		No data

* *Where* $M_i < 1x10^{-5}$ *kilograms,* ** *Where* $M_i \geq 1x10^{-5}$ *kilograms*

The allometric equations have the following general form:

$$x_i = a_i \times M_i^{b_i} \tag{4.19}$$

Where x_i is the natural resource use of an individual (subscript, i), e.g., km^2 individual^{-1} for land area use ($x_i = L_i$); M_i (kg) is the adult body mass of its corresponding species, and a_i and b_i are biological class-specific parameters derived through regression analysis (see Table A2, Appendix 1). Our method uses the 'ecological needs' equations in Table 3.1 in conjunction with the results of ecological surveys (e.g., ecosystem surveys measuring the variables: species name, number of individuals, location details, e.g., habitat type and climate; and where possible, body mass) to determine the land area, water, and energy needed to support desired individuals/networks (e.g., food webs) that comprise these surveyed ecosystems.

3.3.2 Interpreting land use in multi-species populations

Land area use describes activity that satisfies a range of essential (i.e., life-supporting) physiological needs like thirst, hunger, and rest (e.g., space for nesting, and travel to nesting sites). It also describes other, less essential activities. For example, an individual animal's need to socialise or stimulate

themselves. In contrast, plants are sessile; their land area use is limited to essential needs, like the inception of sunlight and the uptake of water, nutrients, and gases associated with photosynthesis and respiration.

Different species co-exist and make use of the same land area within ecosystems. Within a diverse population, social and physiological needs can be met through competitive and non-competitive means. In the latter case, individuals of different species may satisfy their needs without compromising the ability of others to satisfy their own. For the former case, population density captures intraspecific (between the same species) but not interspecific (between different species) competition for land area. Inter-species land area competition includes competition across different spatial planes (e.g., below ground, at ground level, in the canopy level, etc.), and across different trophic levels. While there have been some studies into the shared use of land area between sympatric species (e.g., [400], [401], [402], [403]), comprehensive data on interspecific competition is not readily available in the literature. However, it is not reasonable to take the sum of individual land area use values to describe the land area needed to support a multi-species population.

For these reasons, a conservative approach to interpreting land area use is taken when applying our conceptual framework at this stage. It is proposed that land area use describes a constraint rather than a need in the same way 'energy use' and 'water use' do. Simply, the natural resource used by an individual should be located within the land area use range described by the land area use equation. Additionally, that natural resource must also be accessible to the individual, as illustrated in Fig. 3.3, and of suitable quality. If natural resource is not available to an individual, it is assumed the resource cannot be used to satisfy that individual's wellbeing needs. For illustrative purposes, the land area use of an individual is taken to describe a circular area in Fig. 3.3.

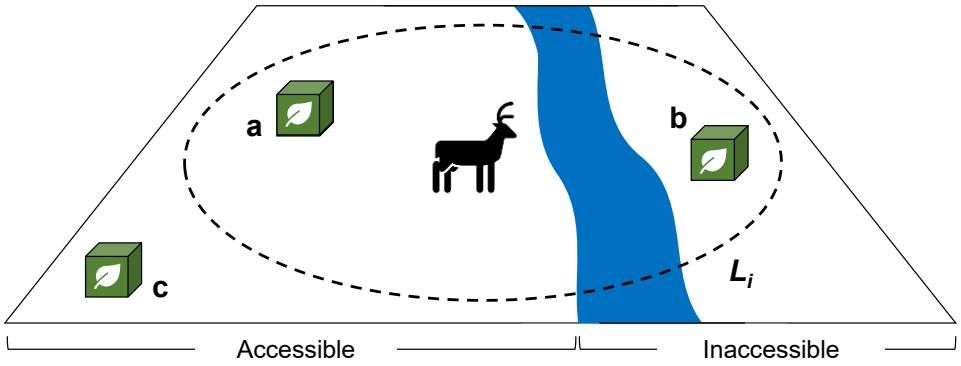


Figure 3.3 Illustration of 'land area use' as a constraint in the estimation of ecological wellbeing needs. L_i (dashed boundary) represents the individual's land area envelope; the land area the

individual – in this case a deer – is anticipated to use. The river acts as a physical barrier. Resources across the river cannot be accessed by the individual. Natural resource stocks – vegetation in this case – are represented by green boxes with leaf icons. Resource (**a**) is expected to be used; it lies within the individual's land use envelope and is accessible. Resources (**b**) and (**c**) are not expected to be used; (**b**) is inaccessible (physical separation – can also include elevated resources), and (**c**) lies outside the individual's land use envelope. In the case of a non-flying species, the land use envelope does not extend beyond their ability to reach (climb, swim, fly, or dig) for resources.

An ecosystem's ability to support biodiversity depends on what energy is available there. Variations in what type and quality of energy is available across different climates and habitat types gives rise to a global species richness gradient [404]. The metabolic values calculated using the proposed equations for energy use (Table 4.1) make it possible to determine the land area needed to support herbivory within a multi-species population – the land area which supports primary producers, herbivores, and omnivores within an ecosystem – based on net primary productivity (NPP) data. This land area, which is termed the 'bio-productive land area, L_{BP} is calculated using Eq. (3.20). Given the homogeneity of ecosystems, L_{BP} varies with climate, habitat-type, and season. We apply and discuss Eq. (3.20) in Section 3.3.3.2.

$$L_{BP} = \frac{E_{p,veg}}{c_{veg} \times \eta_i \times NPP \times \alpha_i} \tag{3.20}$$

Where L_{BP} (km^2) is the bio-productive land area, c_{veg} (kJ kg^{-1}) is the average calorific value for vegetation (primary producers) within the bio-productive land area, η_i (kJ kJ^{-1}) is the assimilation efficiency, i.e., calories extracted divided by calories available; $E_{p,veg}$ (kJ population^{-1} day^{-1}) is the total energy use of individual herbivores and omnivores satisfied through the consumption of vegetation; NPP (kJ km^{-2} day^{-1}) is the net primary productivity; and α_i (kg kg^{-1}) is a coefficient that accounts for the suitability of biomass contributing to *NPP*. For example, if 10% of the *NPP* is used for herbivory (i.e., it is suitable and accessible), α_i = 0.1.

3.3.3 Conceptual intersection with biogeochemical accounting

Applied together, the equations in Table 3.1 represent a novel advancement in modelling resource metabolism through the lens of organisms' physiological needs. Here, the analysis of ecological needs in support of human and ecosystems wellbeing is pursued. There are similar modelling techniques that address the same research question, focussing on various biogeochemical cycles, such as energy [405]. An important example here is 'emergy accounting'; emergy (or 'embodied energy') is defined as, 'the available energy used in transformations directly and indirectly to support human activities or to make a product. It includes both the direct energy inputs (like electricity or fuel) and the indirect energy inputs

required to produce goods and services' [406], [407]. This thesis builds-upon the latter: understanding what energy is required to generate ecosystem services (through supporting organisms or whole trophic systems) or produce animal-based (e.g., agricultural goods) and animal-dependent (naturally pollinated) products.

Though similarities can be drawn, a critical difference between the work in this thesis and emergy accounting is that the emergy concept does not fully account for the prerequisites of ecosystem services (i.e., ecosystem wellbeing needs); land and water, and other resources which concurrently underpin ecological wellbeing and hence ecosystem services fall out of scope. Greater ecological insight and physiological consideration is achieved in this thesis, which offers a more physiologically-informed approach to energy accounting while broadening its scope to account for other, system-critical resources. This similitude offers a further benefit: emergy accounting and comparable concepts can validate trophic models developed through the 'ecosystem needs' approach described in this thesis.

3.3.4 Applying the allometric equations in Table 4.1: a case study at the Natural Capital Laboratory, Scotland

3.3.4.1 Description of the NCL site

The concurrent application of the equations in Table 3.1 is demonstrated using a limited species survey from a site in the Scottish Highlands (Fig. 3.4) that is being studied through a long-term ecological survey [408]. This site, formerly a conifer plantation, is the location for the National Capital Laboratory (NCL), a joint project between AECOM, 'The Lifescape Project', and The University of Cumbria [409]. The NCL was established in 2019 to re-wild the site and monitor changes in biodiversity, habitat types, wildlife, and the quality of natural resource (e.g., soil and water quality) over a five-year period. The aim of the project, in addition to the restoration and conservation of the site, is to understand the causal link between changes in biodiverse ecosystems, climate change, and biodiversity loss. The site itself has a total area of 0.426 km^2 and includes the following broad habitat types: (1) woodland, (2) mountains, moorlands, and heath, (3) freshwater, wetlands, and floodplains, and (4) semi-natural grassland.

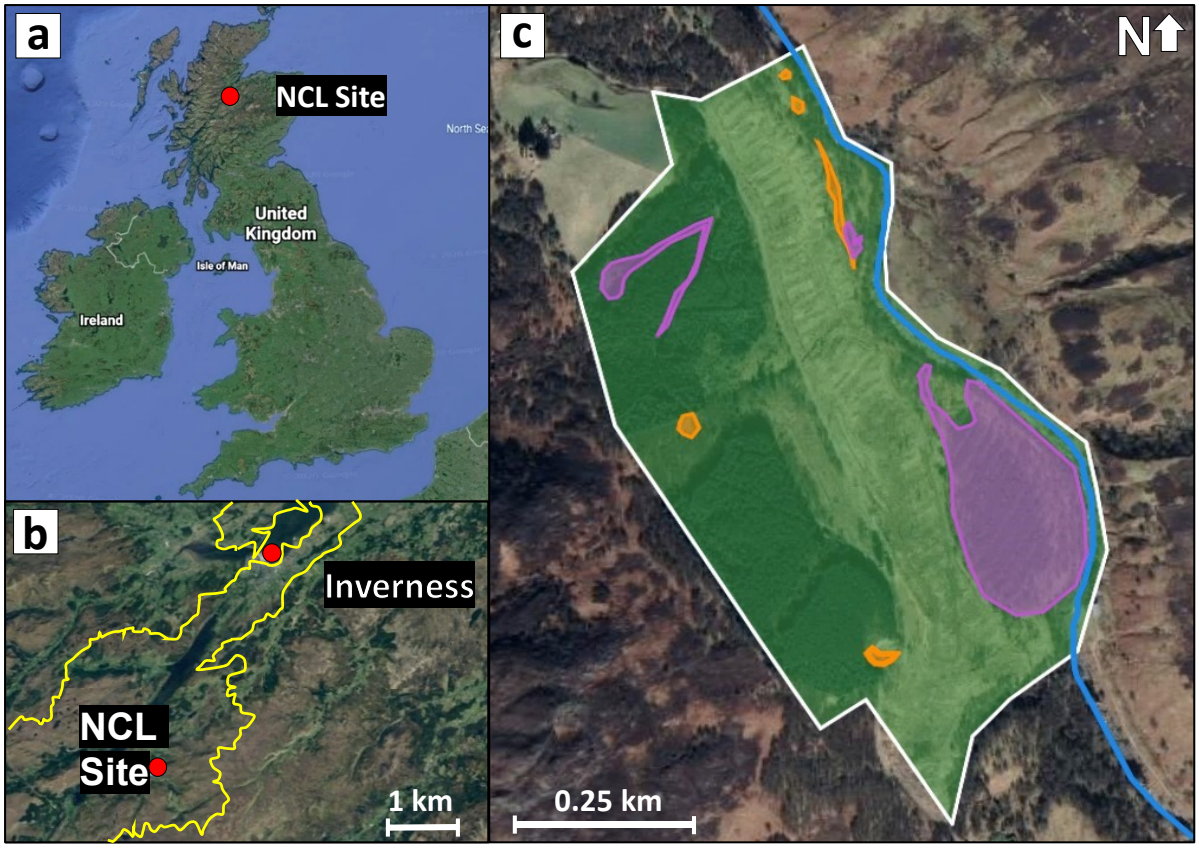


Figure 3.4 Location of the surveyed site within the UK (**a**) in proximity to Inverness, Scotland (**b**), and the habitat types at the NCL site (**c**). In (**c**), the white outline is the NCL site boundary, the green area is woodland, the orange area is semi-natural grassland, the purple area is mountains, moorlands, and heath; and the blue area is freshwater, wetlands, and floodplains – the river Fechlin. The yellow line in (**b**) represents the catchment area of the river Ness. Map data was modified from Google Earth version 9.3.117.0: https://www.google.com/earth/.

The 'freshwater, wetlands, and floodplains' habitat describes an on-site river, which is not considered a terrestrial habitat and so is beyond the scope of this paper. This leaves an area of 0.403 km^2, comprising three broad habitat types: woodland (84.7%); mountains, moors, and heaths (13.7%); and semi-natural grassland (1.5%).

Through site surveys over a two-month period, 45 species (192 individuals) were observed on the NCL site [72]: six (6) species of mammals (N = 9), 31 species of birds (N = 129), seven (7) species of invertebrates (N = 52), and one (1) species of reptile (N = 2) (Table A15, Appendix 4). Once recorded, these species were each assigned an average adult body mass [410]. These adult body mass data are not included in the site surveys but are needed to apply the proposed allometric equations for land use, energy requirements, and water consumption (i.e., in Table 4.1). The number of invertebrates observed in the NCL site is much lower than one would expect. The invertebrate population does not exceed the number of vertebrates, which suggests that the NCL data do not accurately capture the abundance of

species on the site. However, these NCL data are considered useful here to demonstrate the application of our allometric equations and as a basis for more comprehensive data collection in later stages of the site survey. Therefore, total 'land area use' calculated using the NCL data represents an underestimate of that used by biodiversity on-site.

At this stage our framework is applied only conceptually. It aims to describe what quantity of the natural resource produced at the NCL is currently used by biodiversity on-site. In practice, this data would be used to understand the natural resource cost of supporting biodiversity at the NCL site. Specifically, to investigate scenarios of conservation/rewilding versus human development on site: to determine whether current levels of biodiversity (assuming the observed cumulative population (all recorded species) = 1 unit of biodiversity) can be supported at the NCL site given its current natural resource stocks and coexisting socioeconomic use; and whether current or desired levels (*n* units of biodiversity) of terrestrial biodiversity can be supported at the NCL site alongside increased human development (e.g., increased water abstraction, the development of infrastructure on site, etc). The effect of which is decreased natural resource production, and reduced access to natural resource stocks amongst animals (see Fig. 1.4, Chapter 1). The application of our framework could therefore guide conservation and development practices at the NCL site, optimising the site's ability to meet ecosystem and socioeconomic needs simultaneously.

3.3.4.2 Results and discussion

The estimated natural resource requirements of the 45 species observed at the NCL site are presented in Table 3.2 (land use, energy requirements) and Table 3.3 (rates of water consumption). These results do not describe the natural resource requirements associated with larger (i.e., minimum viable) populations of each species observed.

Table 3.2 Land area use, $L_{p,i}$ (km^2 populaton^{-1}), and energy requirements, $E_{p,i}$ (kJ population^{-1} day^{-1}), of populations of species i at the NCL site

Class	Trophic level	N	Land area use [a] L_p (km^2 population^{-1})	Energy use E_p (kJ population^{-1} day^{-1})
Mammals	All	9	2.06	22,106
Mammals	Carnivore	1	0.48	989
Mammals	Omnivore	3	0.48	2,713
Mammals	Herbivore	5	0.70	17,501
Birds	All	129	8.41	14,014
Birds	Carnivore	56	5.33	8,214
Birds	Omnivore	66	2.93	5,249
Birds	Herbivore	7	0.19	551
Reptiles	All	2	4.6x10-4	4
Invertebrates	All	52	2.0x10-5	1,035
Total	**All**	**192**	**10.12**	**37,158**

Table 3.3 Rates of water consumption of populations of species i, $W_{p,i}$ (m^3 population^{-1} day^{-1}), at the NCL site

Class	Trophic level	N	Water use W_p (m^3 population^{-1} day^{-1})
Mammals	All	9	0.01
Birds	All	129	0.00
Reptiles	All	2	No data
Invertebrates	All	52	No data
Total	**All**	**192**	**0.01**

The energy requirement for mammals is greatest despite the observed population of mammals being less than a tenth of the bird population. We attribute this result to the presence of sika deer at the NCL site, which have an average adult body mass of 42 kg. This is the greatest adult body mass of all the observed species at the NCL site, one that exceeds that which is possible for any flying bird species [411]. We suppose that this result may indicate a general trend that the energy requirement of mammals will be high, despite low mammalian populations relative to other species. Overall, these results indicate that birds and mammals will likely contribute most to energy requirement associated with supporting biodiversity. Hence, the estimated energy use of mammals and birds may be sufficient to reliably describe that of a wider population when reptile and/or invertebrate data are unavailable – a scenario which reflects the current situation in the ecological literature.

The extent to which different species share the same land area is not known, so land area use is interpreted as a constraint (as in Fig. 3.3). What can be identified from the NCL data (see Table A15, Appendix 4) is that the greatest land area use at the NCL site is attributed to three species: *Meles meles* (0.48 km^2), *Martes martes* (0.48 km^2), and *Cervus nippon* (0.32 km^2). Given that the NCL site is an

area of 0.43 km^2, it represents sufficient land area to satisfy the land use requirements of most species (excluding *Meles meles* and *Martes martes*). The NCL is not physically isolated; species may make use of natural resource beyond its boundaries to supplement their needs, driven by resource (un)availability or individual preference [412], [413]. More data is needed to confidently determine the extent to which the NCL site supports the land use requirements and other physiological and social needs of each species. Critically, this will be temporally variable, reflecting the seasonality of bio-productivity (e.g., when the plants on-site flower/fruit) and species' diets (e.g., variations associated with malting, gestation, and migration) (e.g., [414], [415], [416]).

Using the estimated energy requirements of species at the NCL site, we estimate the land area needed to satisfy the metabolic needs of primary consumers – i.e., through herbivory; this land area is termed 'bio-productive land area' (L_{BP}) and calculate it using Eq. (3.20). Here the NCL is assumed to be a temperate woodland (broadleaf, deciduous), with average 'net primary productivity' (NPP, the rate at which organic compounds, in this class primary producers, are produced above ground, i.e., leaves, stems, fruit, etc.) of 918 kg km^{-2} day^{-1}. A coefficient (α_i) is applied to adjust the *NPP* because the quality of habitats at the NCL are considered poor [417], and because some biomass will be of inadequate quality or incorrect type to support the observed species. In the absence of comprehensive data on the type of plant species present at the NCL site, or the dietary behaviour of the observed species, it is assumed that 10% of *NPP* at the NCL site will be consumed by terrestrial species [418]. This effectively reduces the availability of energy of the bio-productive land area to 91.8 kg km^{-2} day^{-1}. Finally, an average calorific value for plant matter (c_{veg} , 18,812 kJ kg^{-1}) [419] is used to describe the energy acquired from the bio-productive land area. This means that there is 1.73×10^7 kJ km^{-2} day^{-1} of suitable plant matter produced at the NCL site.

We assume that all (100%) of herbivore energy use is satisfied through the consumption of primary producers and that seventy-five percent (75%) of omnivore energy use is satisfied through the consumption of primary consumers. We acknowledge that a 75:25 ratio of meat to plant matter is unlikely across the observed omnivorous species. However, given the lack of data on the exact diets of the observed species, 75% is used here for the purposes of demonstrating this stage of the method. Using these assumptions, the total energy use that satisfied by the bio-productive land area (i.e., $E_{p,veg}$) is 27,051 kJ $population^{-1}$ day^{-1} (see Table A16, Appendix 4).

This value must next be adjusted to account for the (in)efficiency by which consumed material is converted to usable energy by an individual. This is termed 'assimilation efficiency' (η_i) here. The assimilation efficiency of herbivorous species ranges from 15 kJ kJ^{-1} to 80 kJ kJ^{-1} [418]. Again, owing to the absence of comprehensive data, an average assimilation efficiency of 47.5 kJ/kJ is used for the purpose of this calculation. This means that the herbivorous and omnivorous species in this example must in fact consume 5.69×10^4 kJ $population^{-1}$ day^{-1} to satisfy their metabolic needs.

Inputting the variables (c_{veg}, 18,812 kJ kg^{-1}, η_i = 0.475, NPP = 918 kg km^{-2} day^{-1}, α_i = 0.1 kg kg^{-1}, $E_{p,veg}$ = 27,051 kJ $population^{-1}$ day^{-1}) into Eq. (4.20), a bio-productive land area of 0.033 km^2 $population^{-1}$ is needed to support herbivory at the NCL site. This is equivalent to 8% of the NCL site (0.403 km^2), suggesting herbivory may be adequately supported. The remaining metabolic needs would be met through carnivory, assuming adequate opportunity for predation exists beyond the boundaries of the NCL site.

In total, we calculate that 0.01 m^3 day^{-1} water is required by the species observed on the NCL site. Given that the individual is not restricted to the NCL site, their water requirement can be satisfied using freshwater outside of the NCL site. It was estimated that 0.04 m^3 day^{-1} freshwater was drawn from an on-site well (for socioeconomic use) [408]. This brings the total on-site water demand to 0.05 m^3 day^{-1}. Although abstracted at different points (animals do not draw water from the well on-site), the river Fechlin serves as a constant input of freshwater (one of many possible sources) to the site. Assuming the river Fechlin has a width of 15 m at the NCL site, 32,400 m^3 [freshwater] m^{-1} [depth] day^{-1} [assuming an average velocity of 2,160 m day^{-1} [420]) traverses the NCL site. This suggests that freshwater is not the limiting resource at the NCL site. However, the upper limit of on-site abstraction is difficult to predict. The river Fechlin is part of a larger catchment system, that of the river Ness [421], with a catchment area of 1849.1 km^2 [422], [423] and hence many points of abstraction. Ultimately, combined water use (i.e., by socioeconomic- and ecological systems) from rivers should generally be analysed at the catchment system level to determine whether sufficient water is available to meet their combined needs.

Our findings suggest that conservation projects, like the NCL, would benefit from aligning their project boundaries with physical boundaries that delineate biodiverse ecosystems. The aims of the NCL project includes the restoration and conservation of the site. However, because the observed species likely make use of the NCL as part of a wider land area, it is difficult to determine the full extent to which its

restoration has on biodiversity. Thus, it may be difficult to distinguish which changes are due to the restoration of the NCL site, and which are due to changes beyond its boundaries. It would be beneficial to employ an ecological perspective to assess biodiversity change effectively, i.e., ecosystems should preferably be disaggregated along natural boundaries (i.e., physical boundaries, habitat type, and climate), rather than socio-economic ones (i.e., political, and economic boundaries). For example, at the NCL site, the ecosystem perspective can be lost when socio-economic boundaries are imposed. The results of its rewilding efforts might therefore be misinterpreted. The effects of its restoration would be better understood if the state of its surroundings were monitored simultaneously. Therefore, comprehensive ecological surveys of geographically isolated land areas, and applications of the allometric equations to those sites, presents a promising next step to quantify land area requirements for biodiverse populations.

The NCL site comprises several habitat types (Fig. 4.4): woodland (84.7%); mountains, moorlands, and heath (13.7%); and semi-natural grassland (1.5%). These habitats support biodiversity in different ways, meaning land area use at the class-level will vary by habitat type. However, the distribution of species across the different habitat types at the NCL site is not reported, preventing analysis at this disaggregated level here. More information may be extracted by comparing biodiversity to disaggregated units of the site and its surroundings, like habitat type. The effects of increased land area use by socio-economic systems could then be analysed accounting for the importance of different habitat types for biodiversity amongst different species. This would help to identify which habitat types, and which stocks of natural resource, could be appropriated with the lowest impact on biodiversity. The ability to assess the possible effects of natural resource use on biodiversity would inform decision-making on the allocation of land area (the type and the quantity) for conservation and for socio-economic purposes. Such an improvement would facilitate the achievement of human development goals in ways that may prevent biodiversity loss.

At the fundamental level, all species require adequate space for their ‘wellbeing’. The application of conceptual framework raises questions about how we interpret the needs of terrestrial species in a wider conservation context. For example, the Zoological Society of London (ZSL), the London Zoo, has a collection of over 600 species – over 19,000 individuals – at their Regent’s Park site. However, this site has a land area of 0.15 km^2, which is less than the land area estimated to be used by individual carnivorous mammals with body mass greater than 0.5 kg. This might suggest that biodiverse ecosystems can adapt to exist within greatly reduced land areas if trade-offs between land area and nutritional requirements are managed. For example, through socio-economic intervention such as the provision of food in these areas. This discrepancy might also suggest that we need to improve our

understanding of what natural resources are needed to support individual species and, by extension, multi-species populations in biodiverse ecosystems.

3.4 Conclusions

Our proposed method offers a means by which the natural resource use in ecosystems may be quantified, analogous to human wellbeing needs in socioeconomic systems. This advancement can facilitate the development of much needed comprehensive models of natural resource use, describing socioeconomic- and ecological systems holistically. Critically, this will allow trade-offs driven by the limited availability of natural resource – those between human and ecological wellbeing, and subsequently those between human development and environmental conservation – to be identified, assessed, and managed. This will improve our ability to model and manage natural resource use at the global scale and facilitate the development of sustainable targets for achieving environmental conservation alongside human development.

Our work facilitates a more holistic approach to ecological conservation alongside human development. It addresses a fundamental knowledge gap across the ecological fields, to help satisfy one of the fundamental aims of industrial ecology: to study socioeconomic- and ecological systems holistically. We are not calling for conceptual shift from anthropocentrism to biocentrism. Instead, a pragmatic approach is needed; where new ecological data shows human development causes excessive degradation of ecosystems, socioeconomic bias must give way to compromise by recognising the trade-offs in play.

We present an initial application of our method at this stage, based at the NCL site in the Scottish Highlands, which describes the application of our method in the context of rewilding (a form of ecosystem management). However, our proposed method can be applied in many other contexts. In the same way the ecological footprint [20] describes the cost of global socioeconomic activities (number of planets), our method, in describing the natural resource cost of conserving biodiversity, provides an intuitive basis for natural resource management. Furthermore, by describing the natural resource cost associated with conservation analogous to that of satisfying human needs, there is potential to introduce our method to describe the cost of our conservation ambitions concurrent to socioeconomic activities. In a local context, our method can also complement assessments of urban metabolisms (e.g., [155]). These assessments use material flow analyses (MFAs), which describe the use of natural resource within defined boundaries – here a city. In a specific local context, our method can be incorporated into

the life cycle assessment (LCA) framework [424] to quantify the impact of urban development, for example, on local biodiversity – often called for in the literature [341].

Consider a plot of land within a city, for which several development plans are being considered. Each plan will require different quantities of materials, and each plan will alter the undeveloped plot in different ways (land area use change, the inclusion of blue-green infrastructure, etc.). Given location-specific data on urban animal populations (those expected to make use of the plot of land in question), we can describe the natural resource cost needed to support a population (1 unit) expected to use the plot of land. Hence, we can infer the number of populations (i.e., 'n' units) each development scenario can support. By comparing these results, in conjunction with other LCA results, a more informed decision can be made on which development best suits the socioeconomic and conservational ambitions of the developers, the community, and so forth.

There are two priorities for the advancement of the proposed ecological needs method:

- Capturing the spatial and temporal heterogeneity of ecosystems [220] within our framework. The use of natural resource in ecosystems varies at the local and global scale, in line with variables like habitat type, ecosystem structure, and the seasonality of plant species. Therefore, our allometric equations should be advanced to include additional terrestrial biological class, plant species, and marine species, while also disaggregating between different habitat types and climates.

- Quantifying the natural resource cost of conserving species critical to ecosystem functionality (pragmatic rather than comprehensive conservation, i.e., no loss of species [425]). Thus, environmental conservation targets can be developed that protect critical ecosystem functionally with less socioeconomic compromise than a 'no loss of species' approach. This application of is contingent sufficient data describing the role different species and intraspecific interactions play in critical ecosystem functionality.

Ultimately however, our ability to understand the limits to ecologically sustainable human development is contingent on the collection and availability of ecological survey data across different habitat types and different biological classes, and at the global scale. In the absence of such data, the efficacy of contemporary human-development and environmental-conservation targets are uncertain – a problematic prospect given present-day levels of ecological decline.

4 Including local terrestrial biodiversity impacts in life cycle assessment (LCA)

This chapter is based on the manuscript, Mason, A. R.; Gathorne-Hardy, A.; Myers, R. J.; Smith, B. M.; Puchol-Salort, P. (under review). Local Terrestrial Biodiversity Impacts in Life Cycle Assessment: A Case Study of Sedum Roofs in London, UK. *Journal of Industrial Ecology*.

4.1 Introduction

Urbanization poses a significant threat to biodiversity and the provision of natural resources as over half of the global population resides in urban areas, a number expected to reach 86% by 2050 in developed regions. Increasing urban populations and affluence, particularly in the global north, is driving the demand for urban development and thus considerable land use change. This diminishes terrestrial biodiversity, with critical natural resources and ecosystem services. Ecological components are increasingly being used in urban design for both socioeconomic and ecological benefit, foremost motivated by human needs. To this end, green infrastructures like green roofs, urban wetlands, forests, and ecological corridors are at the fore of urban design. Among their benefits is urban biodiversity support. However, critical knowledge gaps limit our understanding of what urban biodiversity different green infrastructures can support in different contexts. An important deficiency relates to life cycle assessment (LCA), which plays a vital role in guiding biodiversity-supporting urban design.

There remains a need to characterise the potential biodiversity impacts of land use more comprehensively in LCA, particularly in the urban context. These characterisations should capture the local (alpha) biodiversity impacts of urban infrastructures across at the system level. By modelling local ecosystems explicitly, the potential local biodiversity impacts of urban land use can be quantified and characterised in a way that is compatible with existing life cycle impact assessment (LCIA) methods (e.g., ReCiPe 2016 [84], IMPACT World+ [19]). 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. Prior to this thesis [26], a set of ecological wellbeing needs, equivalent to human wellbeing needs [62], [123], had not been quantified. This concept has not yet been incorporated into system analysis methodologies like LCA. Here, we build upon the concept of ecological needs to addresses several deficiencies relating to the assessment of local biodiversity impacts through LCA.

This study presents 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. The energy provision of sedum roofs is related to the metabolic requirements of terrestrial animals to determine what species richness (number of species) and species abundance (number of individuals) can be supported. This methodology is demonstrated using a case study in London, UK, comparing the life cycle biodiversity impact of developing 18 residential buildings with either asphalt-concrete or sedum roofs. This study has three parts:

1. Demonstrate through novel methodology how the natural resource use of terrestrial species can be modelled to produce a more comprehensive inventory of socioeconomic and ecosystem activities.
2. Integrate the results into the LCA method, focussing 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 characterisation factor linking urban land occupation as 'urban, sedum roof, London' to local terrestrial biodiversity change.

This study demonstrates the potential and limitations of quantifying terrestrial biodiversity offsets offered by urban infrastructures (sedum roofs in this case) alongside urbanisation, and the need for realistic expectations of what role it might play in sustainable urban design.

4.2 Method

4.2.1 Case study site: the Meridian Water Development (MWD) 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 an 0.85 km^2 site in Enfield, London ,UK (Fig. 4.1a and 4.1b). 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.

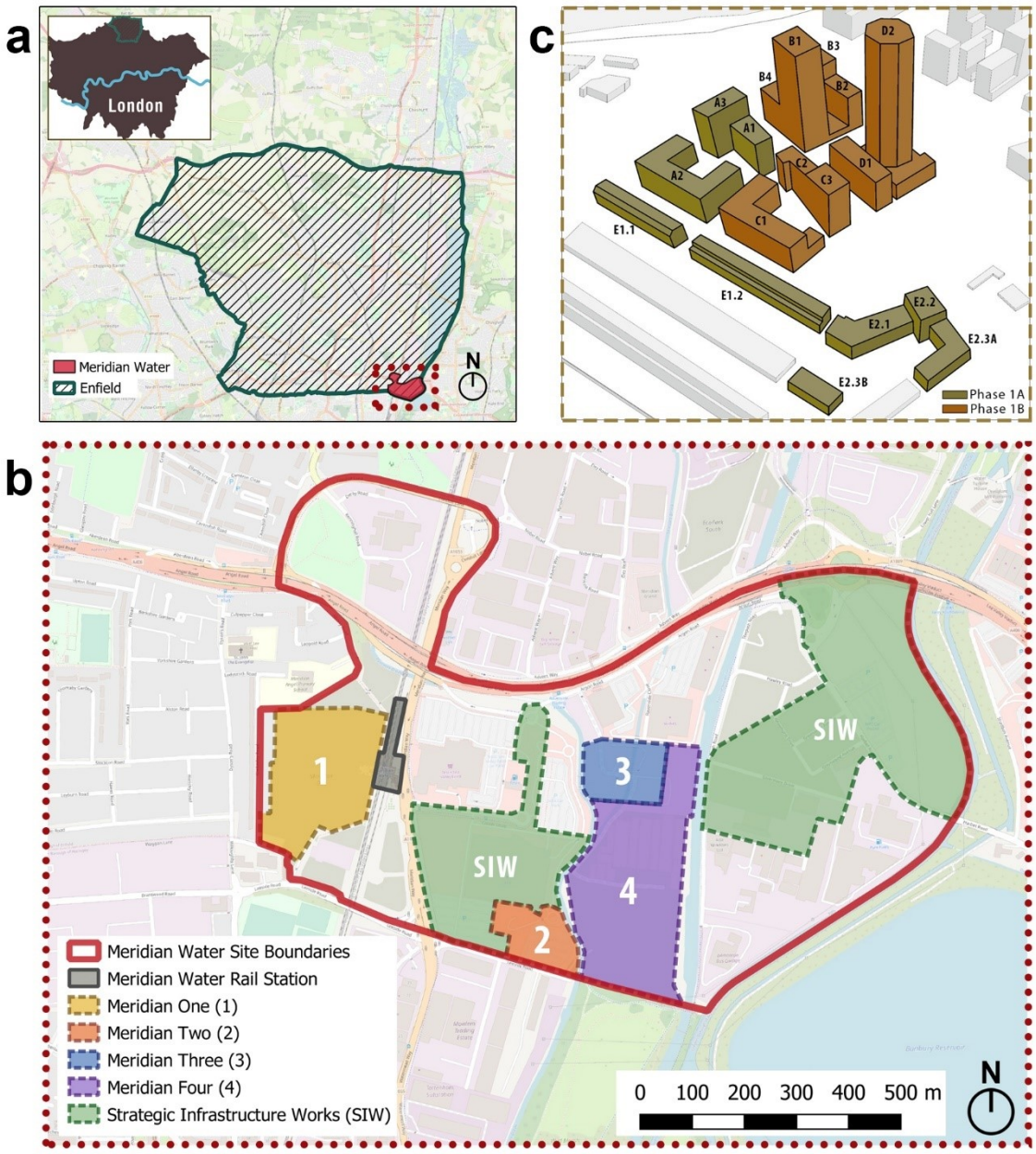
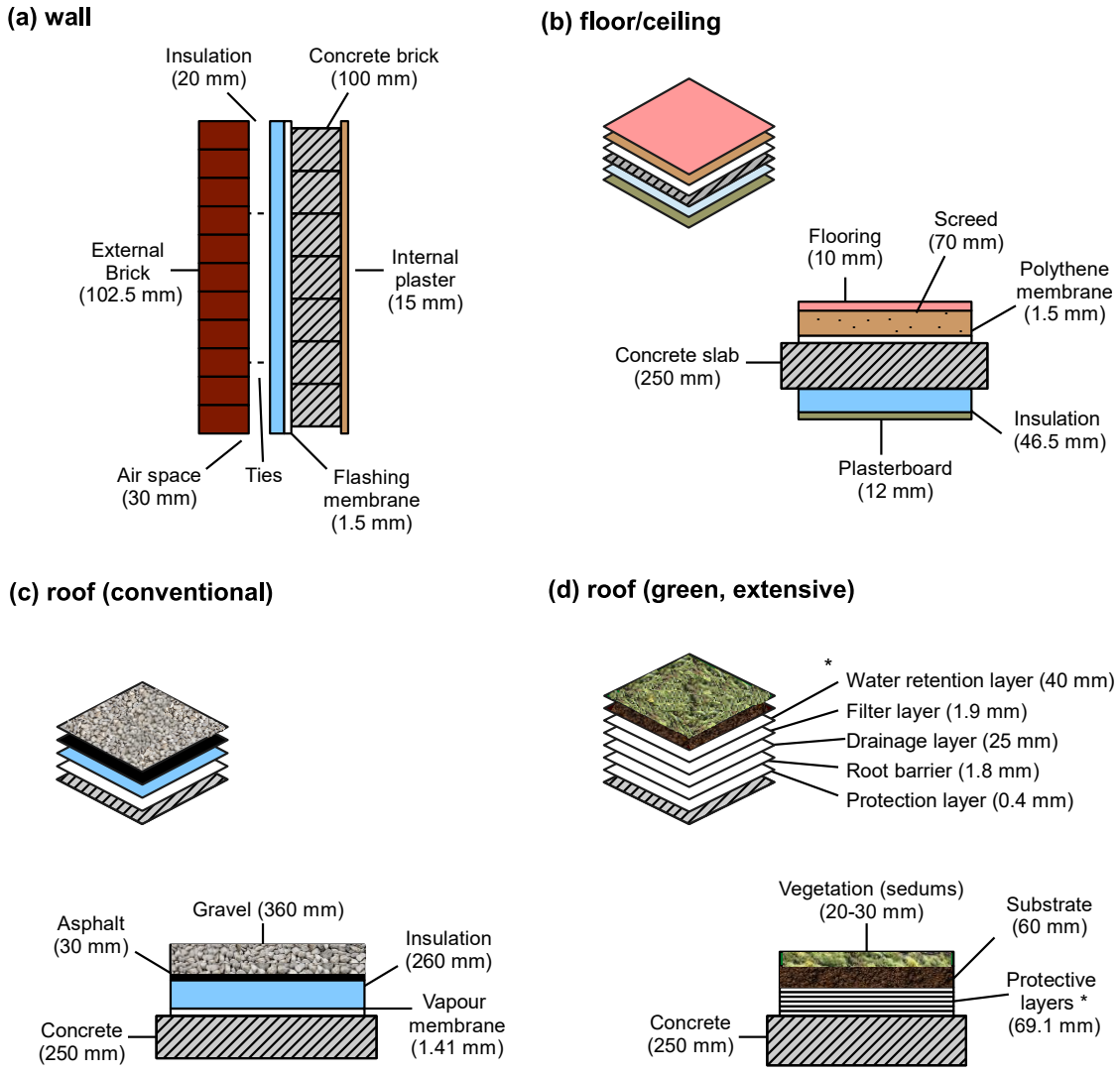


Figure 4.1 Site of the Meridian Water Development (MWD) in Enfield, London, UK, showing (**a**) the location of Enfield (borough) within the Greater Metropolitan London area, (**b**) the development stages of the MWD, and (**c**) the 18 residential buildings being constructed in Meridian One, each assigned an individual 'building ID'. These IDs are used to disaggregate material use for construction in Table 4.1. Parts (**a**) and (**b**) are drawn using QGIS software. Part (**c**) is adapted from a visualisation by 'Hawkins\Brown' and 'HTA design LLP' [426]

The MWD site is currently classified as brownfield land, having previously been developed for industrial use [427]. The first development stage, Meridian One, will comprise 18 residential buildings encompassing 725 apartments (Fig. 4.1c), 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 realised 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. 4.2 shows the modelled construction of (**a**) walls, (**b**) floor/ceilings [9], (**c**) concrete roofs, and (**d**) extensive sedum roofs modelled in this case study [83], [429]. These compositions are used to determine the quantities of each material used during the lifetime of Meridian One.



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Figure 4.2 Schematics for (**a**) walls, (**b**) floor/ceilings, (**c**) conventional roof, and (**d**) sedum roof at Meridian One. Substrate is assumed to be: 10% crushed brick, 10% clay, and 80% compost [83], [430]

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 [431], [432]. 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 [83].

4.2.2 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. Fig. 4.3, a LCA process flow diagram for this case study, describes the processes that make-up our assessment across its three life cycle stages.

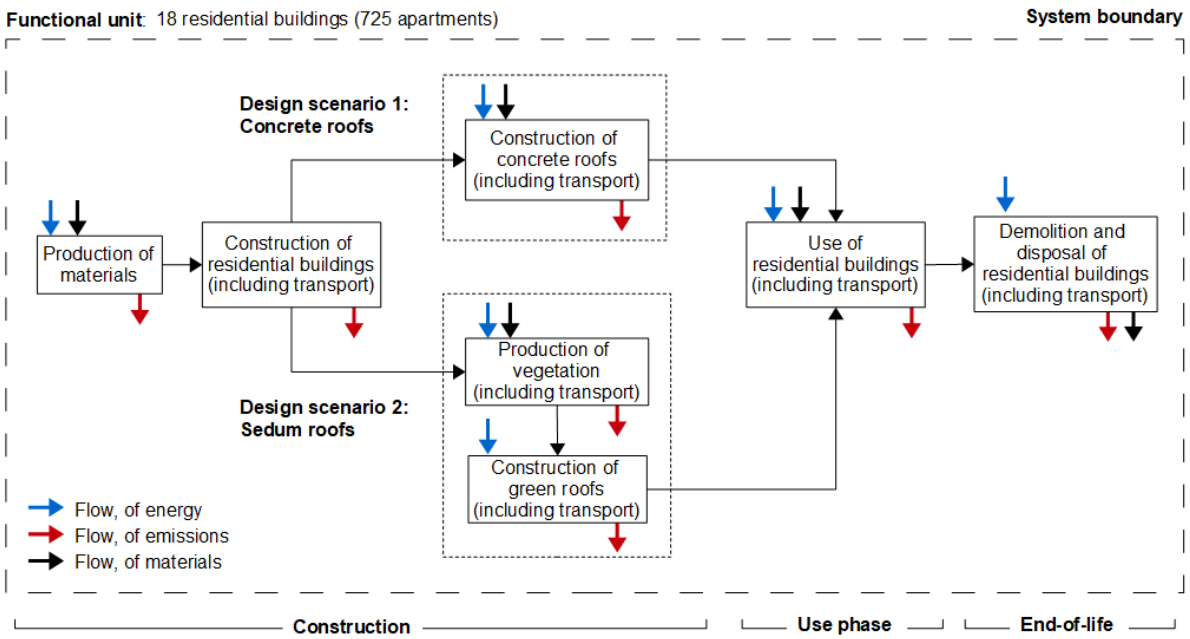


Figure 4.3 Flow diagram describing the processes and flows (thick arrows) involved in the development of Meridian One, from cradle-to-grave, as modelled in OpenLCA v.1.11.0 (with ecoinvent v.3.8). The thick, coloured arrows are the flows (energy in blue, emissions in red, and materials in black). The thin, black arrows signify the direction of flow across three life cycle stages: Construction, use, and end-of-life.

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 modelling was carried out in OpenLCA v.1.11.0, using endpoints in the IMPACT World+ method [19]. Endpoints are chosen since we are interested in biodiversity loss and gain, which is not reported at the midpoint level. 16 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) [19], [84], [321]. 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 [433]).

4.2.3 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' [434], [435]. 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. A3, Appendix 2, 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 (Fig. 4.1c) 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.

4.2.4 Life cycle inventory

Table 4.1 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 Fig. 4.2), including the production of vegetation per Fig. 4.3. All underlying data, calculations, and assumptions are presented in Appendix 1, 'Supplementary information for Chapter 4'; Appendix 5; and in 'Sections 4.2.4.1-4.2.4.6'.

Table 4.1 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 (metric kiloton, 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]
Aluminium	-	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'

4.2.4.1 Production of vegetation

We developed a unit process for the production of vegetation for the sedum roofs by quantifying the material use associated with sedum propagation and growth prior to installation at Meridian One. The vegetation is produced and sold as a 'sedum carpet', containing up to six species of sedum per square metre. With a saturated weight of 14.7 kg m^{-2}, the green roofs have a total mass of 158,798 kg [430]. The sedums are grown from seed on biodegradable coir matting (3 mm thickness), using an organic compost (10 mm thickness) [430]. Propagation and growth take up to 12 weeks [436]. Watering is required during propagation, whereby the coir matting is rehydrated every other day (2 L water m^{-2}) [437], [438]. In total, 84 L water m^{-2} is required. After propagation, the sedums are delivered to site and installed. Rainfall is sufficient to sustain the sedums thereafter [438].

4.2.4.2 Maintenance of concrete and sedum roofs during use

Both roof designs require maintenance during the use phase of the development's 100-year lifetime. The concrete roof's 'vapour membrane' (Fig. 4.2c) has a 20-year lifetime, meaning it must be replaced four times. From a practical standpoint, the asphalt layer, which overlays the vapour membrane, must be re-applied each time the vapour membrane is replaced. The sedum roof's 'protection layer' has a 50-year lifetime, requiring replacement once [83]. Again, from a practical standpoint, all overlaid materials (vegetation, substrate, water retention layer, filter layer, drainage layer, and root barrier) must be replaced when the protection layer is renewed.

Further maintenance of the green roofs is not necessary [83]. Sedums typically do not require irrigation [439] and unlike green roofs grown for aesthetic purposes, gardening (i.e., plant maintenance) and fertilisation regimes are not required for the sedum roof in this case study [440]. The sedum roofs will not be replanted other than at 50 years of age: the design modelled should last at least 50 years, over which time they form an established and integral part of the local ecosystem [430]. Replanting them will alter the rate and quality of energy production on-site over a short timeframe, reducing its ability to support biodiversity until the established level (quantity and quality) of energy provision is restored.

4.2.4.3 Utility use by residents

The expected domestic use of utilities at Meridian One (725 households) over its 100-year lifetime are: 210 GWh electricity, 870 GWh gas, and 10.4 Mt water. We base our assumptions on typical UK domestic consumption values for electricity (medium-level, Profile Class 1) and gas use, as reported by the British 'Office of Gas and Electricity Markets' (Ofgem) [441], [442]; and Environment Agency-

reported UK household consumption values for water use [443], [444]. We extrapolate these data over a 100-year period.

While green roofs can reduce heat-loss from buildings [445], only the uppermost apartments across Meridian One will feel this benefit. Given that the residential units at Meridian One are multi-story (30 floors in some cases), no significant impact on utility use is expected across the residences. Therefore, we assume equivalent domestic gas and electricity use for the two (concrete and sedum) roof scenarios.

4.2.4.4 End-of-life activities

End-of-life activities, including demolition, take place after 100 years. We assume that these activities happen simultaneously, whereupon land occupation ceases, and the site reverts to brownfield (Fig. A3, Appendix 2). After demolition, waste materials are transported from the site for processing, either to recycling centres or landfill sites. Reflecting UK waste management practices, Table 4.2 lists the fates of end-of-life waste at Meridian One [446], [447], [448], [449].

Table 4.2 Fates of end-of-life waste at Meridian One

Material	Recycled (%)	Landfill (%)
Aluminium	70	30
Asphalt	-	100
Brick	87	13
Brick (crushed)	-	100
Cement	100	-
Clay	-	100
Compost	-	100
Concrete brick	87	13
Flashing membrane	-	100
Flooring (wood board)	81	19
Glass	30	70
Gravel	100	-
Plasterboard	-	100
Polyethylene	81	19
Polypropylene	-	100
Polystyrene	-	100
Polythene	-	100
Rock wool	-	100
Sand	100	-
Vegetation	-	100

4.2.4.5 Transportation of materials

Transportation of the materials used in the construction and use phases is modelled by road using a '16-32 tonne vehicle' (UUID: 4ecf1190-7028-4038-89b0-79adeb3e98cc). Transportation of vegetation and

substrate is modelled as by road, using the same 16-32 tonne vehicle type, from Aylesbury 60 km from Enfield [436] . It is assumed that all other materials will be sourced from within Greater London or its surrounding areas, within 50 km of the site [450]. All end-of-life waste (at the 100-year point and disposed of during maintenance) will be transported by road using 16-32 tonne vehicles and processed 21 km away in Ware, Hertfordshire [451]. Table 4.3 describes the tonne-kilometre (t km) transportation requirements across the case study's lifetime.

Table 4.3 Transportation of materials (tkm) during the 100-year lifetime of Meridian One.

Life cycle stage		Transport (tkm)		
			Design scenario	
		Vehicle	Concrete roofs	Sedum roofs
Construction		16-32 tonne	$4.79x10^{10}$	$4.65x10^{10}$
Use		16-32 tonne	$6.68x10^{9}$	$2.04x10^{7}$
End-of-life		16-32 tonne	$2.29x10^{10}$	$1.95x10^{10}$
	Landfill	16-32 tonne	$2.09x10^{9}$	$1.96x10^{9}$
	Recycle	16-32 tonne	$2.09x10^{10}$	$1.76x10^{10}$
Total		16-32 tonne	**$7.76x10^{10}$**	**$6.61x10^{10}$**

4.2.4.6 Distinguishing between land use categories at different elevations

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 A36-A37, Appendix 5). 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.

4.2.5 Biodiversity at Meridian One

4.2.5.1 Constructing a food web to describe energy flows between terrestrial species 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 [452], [453], [454]. The goal here is to develop a reasonable estimate of a local, terrestrial species population, which could make use of the sedum roofs; as well as 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, the population is restricted 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. The population is assumed to be static with no variation in individual species' diets during this period. It is acknowledged 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 (Table A3, Appendix 2), and a trophic web illustrated in line with these assumptions (Fig. 4.4, overleaf). All underlying assumptions are discussed in 'Section 11.2.1, Appendix 2'.

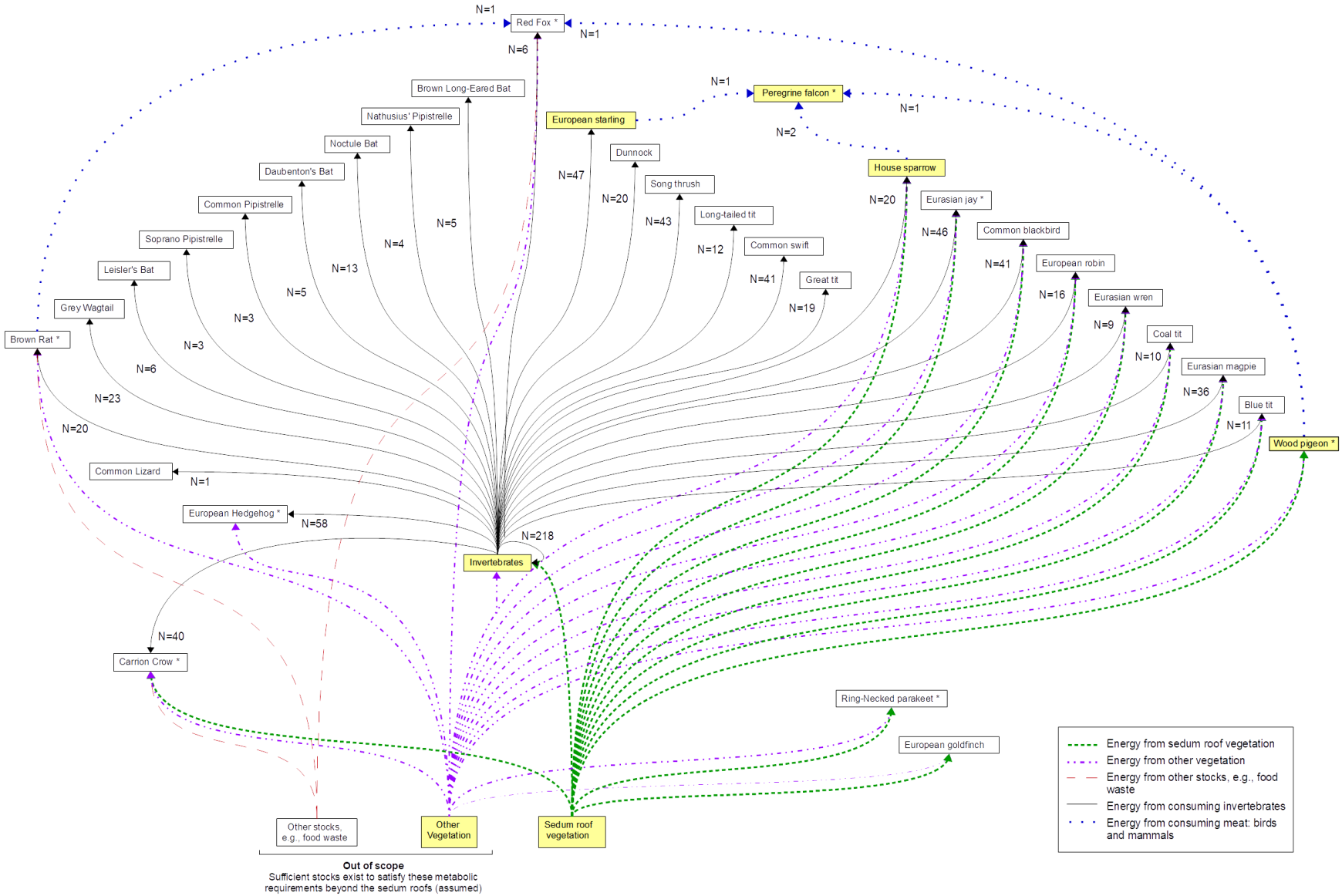

Red Fox *
Peregrine falcon *
Brown Long-Eared Bat
Nathusius' Pipistrelle
Noctule Bat
Daubenton's Bat
Common Pipistrelle
Soprano Pipistrelle
Leisler's Bat
Grey Wagtail
Brown Rat *
Common Lizard
European Hedgehog *
Carrion Crow *
European starling
Dunnock
Song thrush
Long-tailed tit
Common swift
Great tit
House sparrow
Eurasian jay *
Common blackbird
European robin
Eurasian wren
Coal tit
Eurasian magpie
Blue tit
Wood pigeon *
Ring-Necked parakeet *
European goldfinch
Invertebrates
Other stocks, e.g., food waste
Other Vegetation
Sedum roof vegetation
N=1
N=6
N=47
N=20
N=43
N=12
N=41
N=19
N=2
N=46
N=41
N=16
N=9
N=10
N=36
N=11
N=5
N=4
N=13
N=3
N=23
N=218
N=58
N=40
Out of scope
Sufficient stocks exist to satisfy these metabolic requirements beyond the sedum roofs (assumed)
Energy from sedum roof vegetation
Energy from other vegetation
Energy from other stocks, e.g., food waste
Energy from consuming invertebrates
Energy from consuming meat: birds and mammals

Figure 4.4 Interactions between the species likely to exist at Meridian One, (see 'Section 4.4') and five energy sources: energy from sedum roof vegetation (i.e., plant-biomass and nectar; dashed green lines), energy from other vegetation (i.e., not on the sedum roofs; dot-dash purple lines), energy from anthropogenic waste (e.g., food waste, dashed red lines), energy from consuming invertebrates (solid black lines), and energy from consuming other animals (dotted blue lines). For each predator-prey interaction, the quantity of each prey species consumed (*N*, individuals) is described; each predator represents one (1) individual and resource use is scaled where individual land area use exceeds the area of the case-study site (signified by an asterisk, *); here, metabolic requirements are scaled accordingly. Adequate resource provision beyond the sedum roofs is assumed. Species comprising the peregrine falcon 'unit' are highlighted yellow. Energy from waste is described here as it forms part of the species' diets, but its provision lies beyond the scope of this study and is not included hereafter

Fig. 4.4 describes how energy provision by the sedum roofs is expected to support metabolic requirements of a terrestrial ecosystem at Meridian One (Table A3, Appendix 2). It shows that not all species interact with one another or the sedum roof directly. The food web is made up of 31 co-existing 'units', where units are energy-flows from primary producer to final consumer. For example, the unit where energy flows from plants to the peregrine falcon reads: vegetation → invertebrates → intermediate bird species (European starling, house sparrow, and wood pigeon) → peregrine falcon. This unit is highlighted in Fig. 4.4 using yellow name boxes. Fig. 4.4 also describes daily predation rates ([individuals of species j consumed by species i] [individuals of species i]$^{-1}$ day^{-1}). Apex species are assigned a species abundance value of one (1.0) individual. Thus, the species abundances of prey species scale relative to that of the apex species within each unit of the food web.

4.2.5.2 Quantifying the species richness supported by sedum roofs at Meridian One

To calculate the species richness (number of species) 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 [455]. Native and non-native species contribute to alpha biodiversity, so both are considered in this study. This includes invasive non-native species such as the ring-necked parakeet [456].

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 2). Based on the assumptions used (see 'Section 10.2.', Appendix 2), the sedum roofs produce 8.6x10^5 kJ plant biomass day^{-1}, and 12,458 kJ nectar day^{-1} [457], [458], [459].

A bottom-up approach is used to describe the distribution of energy, from the sedum roofs, through the food web. Data on inter-species predation based on observation or from stomach contents, for example, are conventionally used to construct trophic models of this nature [222], [460], [461]. 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. As primary consumers and the most numerous species within the trophic system, invertebrates significantly influence the flow of energy at Meridian One. They mediate the transfer of energy from lower to higher trophic levels here [365]. Furthermore, it is assumed thereafter that each animal has the same opportunity as its competitors

to access energy (i.e., energy stocks are distributed equally between consumers). The energy intake of each species is grounded in validated metabolic scaling laws [217]. The distribution of energy through the trophic system is in reality more dynamic [363]; however, this 'uniform flux' assumption renders the data more manageable for the purposes of this case study. The underlying data are presented in Tables A29 and A30, Appendix 5.

4.2.5.2.1 Worked example: Peregrine falcon (*Falco peregrinus*)

Here we present a worked example for the part of the food web relating to the peregrine falcon, as described in Fig. 4.4. We describe the general method below, the application of which is illustrated in Fig. 4.5, overleaf. The calculation then follows.

Predator-prey relationships are used to quantify relative species abundances within each unit of the species population expected at Meridian One (see Fig. 4.4). This is done using data on the dietary behaviours and metabolic requirements of predator species, and the calorific values of species they prey upon. Dietary behaviours are identified in the ecological literature; the metabolic requirements of species are calculated using allometric equations relating body mass (kg) to required daily energy intake (kJ day^{-1}) [26]; and the calorific values of prey species are sourced from publications where possible or estimated (see example below). The general method used to calculate relative abundances is described below:

1. Identify an apex predator
    - Identify a species in the highest trophic level of the food web. In Fig. 4.4, this could be the Peregrine falcon. This species will be the starting-point, with a population size of one individual (n_i = 1 individual).
2. Quantify the metabolic requirement, E_i (kJ day^{-1} $individual^{-1}$), of one individual of that species [26].
3. Identify the predator's feeding habits:
    - The types of food/species consumed
    - How much each food type/species contributes to the individual's diet
    - The predator's ability to extract energy from food (assimilation efficiency, η_i)
    - The calorific value (kJ kg^{-1}) of each food eaten, for converting energy to mass
4. Quantify the amount of each organism needed to satisfy the predator's metabolic requirement on an energy basis: kJ (prey) day^{-1}

5. Quantify the amount of each prey and vegetation needed to satisfy the predator's metabolic requirement on a mass or abundance basis: kg day^{-1} or individuals day^{-1}
6. Repeat these steps for each prey, ending after primary consumers, herbivores, have been described. Plant species (vegetation) are not described.

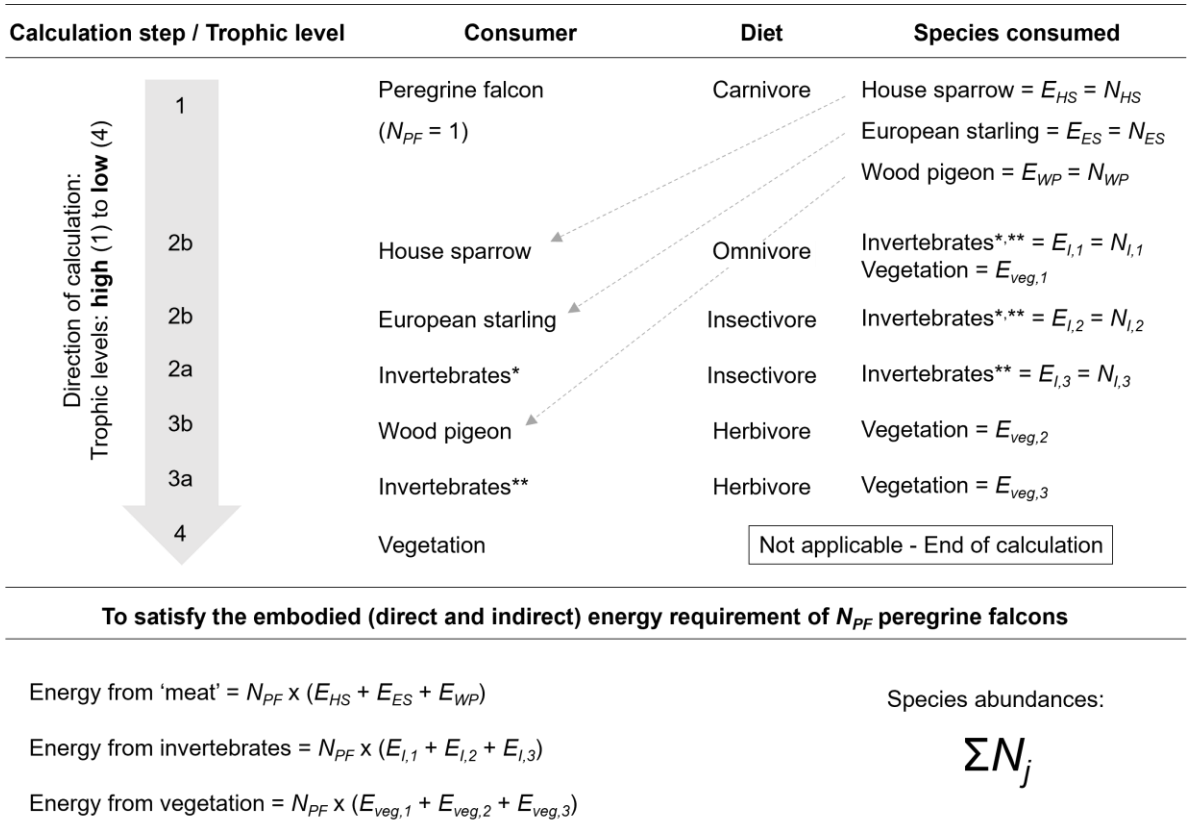


Figure 4.5 Step-wise method to quantify species abundances. Here, the peregrine falcon unit is described (see Fig. 5.4, yellow boxes). The 'consumer' is *the species doing the eating*, and the 'species consumed' is *the species being consumed as prey*. E_j (kJ [consumed as prey] day^{-1}) represents the quantity of each species, j, consumed as prey per day as 'energy'. N_j ([individuals consumed as prey] day^{-1}) represents the quantity of each species, *j,* consumed as prey per day, as 'numbers of individuals' (based on the consumed species' assumed calorific values, c_j). The following subscript notations are used: 'PF' for peregrine falcon, 'HS' for house sparrow, 'ES' for European starling, 'WP' for wood pigeon, 'I' for invertebrates, and 'veg' for vegetation. Numbers are used where prey contributes to more than one predator's diet. Here, 'invertebrates' describes 35 species, aggregated (see Table A3, Appendix 2). Invertebrates as prey can describe both insectivorous invertebrates (Invertebrates*), and herbivorous invertebrates (Invertebrates**), and are distinguished here using asterisks. The large, grey arrow (left, top to bottom) shows the direction of calculation. The dashed, grey arrows highlight that consumed species in one trophic level are then considered as consumers in the subsequent trophic level(s)

We start with an individual (1) peregrine falcon, with an adult body mass of 0.88 kg [462]. We first quantify its energy requirement using Eq. (5.1) [26], which relates the adult body mass (M_i) of a carnivorous bird to a daily metabolic (energy) requirement (E_i). Using Eq. (4.1), the peregrine falcon's metabolic requirement is calculated to be 1,065 kJ day^{-1} (Eq. (4.1)).

$$E_i = 1159.3 \times (M_i)^{0.681} \tag{4.1}$$

Next, we describe the peregrine falcon's diet. The peregrine falcon is carnivorous (meat-eating) [462]. Here we assume its diet comprises house sparrow, song thrush, European starling, and common swift in equal amounts (in calorific value), as the MWD site is intended to support these species. An aggregated calorific value for meat is used: 5,767 kJ kg^{-1} (see Appendix 3). Next, we must consider the peregrine falcon's assimilation efficiency (η_i) – its ability to extract energy from its food [463]. One study reports an assimilation efficiency of 85% [464]. Eq. (4.2) is then used to describes the quantity of each food type needed to satisfy the falcon's metabolic requirements.

$$M_{c,j} = \frac{E_i}{c_j \eta_i} \tag{4.2}$$

Where $M_{c,j}$ (kg day^{-1}) is the mass of a consumed species, *j*, corresponding to the adjusted, 'energy consumed', energy requirement of the species acting as consumer: species i; c_j is the calorific value (kJ kg^{-1}) of the consumed species, j; E_i (kJ day^{-1}) is the energy requirement of an individual of the species acting as the consumer, i; and η_i is the assimilation efficiency of the consumer species. Using Eq. (4.3), this is converted into species abundance N_i (individuals): number of individuals consumed.

$$N_i = \frac{M_{c,j}}{p_j} \tag{4.3}$$

Where p_i ([kg meat] [kg prey]$^{-1}$) is the consumable portion of a prey species' (j) adult body mass. p_i depends on the morphology of the prey species and on the physiology of the predator (method of consumption) – for example, whether the prey is consumed whole or whether parts are selectively/preferentially consumed. Here a p_i value of 0.65 for birds and either 1.0 (where eaten whole) or 0.75 (where eaten by spiders) for invertebrate is assumed [465].

In this example, p_j *for birds* is the meat-to-body mass ratio of small bird species (eaten by the falcon). Farmed poultry have dressed weights around 75% of their live weights [179]. This value will be higher than that of small wild birds as farmed species have been engineered to produce unnaturally large quantities of meat. The dressed weight of small wild birds is assumed to be 65% of their live weight. There is a lack of data on the dressed weight of wild bird species, so we base our assumption on data for larger wild bird species (e.g., pigeons [~65%] [466] and guinea fowl [~70%] [467]), which is scaled-down to reflect the leaner build of small UK-native birds. In later steps, where invertebrates are consumed: a p_j *for invertebrates* of 1.0 is assumed where the invertebrate is eaten whole. This will be less for spiders, which possess piercing mandibles and employ external digestion. It is reported that invertebrates are approximately 25% exoskeleton by mass (chitin content) [468]. p_j *for invertebrates* is therefore taken to be 0.75 where the predator is a spider.

Assuming constant dietary habits, 0.09 kg meat falcon^{-1} day^{-1} is needed to satisfy the metabolic needs of one peregrine falcon (the first trophic level (1) in Fig. 4.5). Assuming p_j *for birds* = 0.65 and assuming the sparrow, pigeon, and starling contribute equally to the falcon's diet, this equates to:

1. 4.29 (house sparrow) (falcon)$^{-1}$ day^{-1} → 5 (house sparrow) (falcon)$^{-1}$ day^{-1}
2. 0.30 (wood pigeon) (falcon)$^{-1}$ day^{-1} → 1 (wood pigeon) (falcon)$^{-1}$ day^{-1}
3. 1.74 (European starling) (falcon)$^{-1}$ day^{-1} → 2 (European starling) (falcon)$^{-1}$ day^{-1}

Here, values are rounded-up to avoid the presence of partial individuals. The rationale here is that the consumption of any portion of an individual requires a whole individual to fall prey. We next consider the dietary requirements of the bird species consumed by the peregrine falcon (Calculation steps/trophic levels 2a, 2b, and 2c; Fig. 4.5), then the requirements of invertebrates (Calculation steps/trophic levels 3a and 3b, Fig. 4.5) and plants (Calculation step/trophic level 4, Fig. 4.5).

4.2.6 Developing an endpoint characterisation factor for urban sedum 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 [237]. Our ecological model describes a similar relationship for sedum roofs at Meridian One, facilitating novel

characterisation 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 8, Supplementary Information.

We develop an endpoint characterisation 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+ method. Our biodiversity methodology and results can also be applied to calculate CFs consistent with other LCIA methods.

In IMPACT World+, the impact of land occupation on ecosystem quality, $I_{occupation,LU_{a,b}}$, (PDF.m^2.year) – where the subscript '$LU_{a,b}$' denotes the case-specific context: land use, LU, of type a in location b – is calculated using Eq.(4.4) [469], [470]:

$$I_{occupation,LU_{a,b}} = CF_{occupation,LU_{a,b}} \times A_{occupation,LU_{a,b}} \times t_{occupation,LU_{a,b}} \quad (4.4)$$

Where CF (PDF.m^2.year (m^2.a)$^{-1}$) is the endpoint characterisation factor for land occupation, $A_{occupation,LU_{a,b}}$ (m^2) is the occupied area, and $t_{occupation,LU_{a,b}}$ (years) is the duration of occupation. By rearranging Eq. (5.4), the endpoint characterisation factor for sedum roofs in this case study can be calculated using a known I_o value (Eq. (4.5)):

$$CF_{occupation,LU_{a,b}} = \frac{I_{occupation,LU_{a,b}}}{A_{occupation,LU_{a,b}} \times t_{occupation,LU_{a,b}}} \quad (4.5)$$

4.3 Results and discussion

4.3.1 Biodiversity supported by sedum roofs at Meridian One

We found that primary production by the green roofs supports 53 terrestrial species comprising 673 individuals (see Table A35, Appendix 5). These 53 species and the flow of energy between them, across four trophic levels, are described in Fig. 4.6. While not included in the IMPACT World+ method, species abundance is a useful additional output of our modelling. It confers information on the population that should inform decision-making in urban design. In this case, the population has low species diversity ('Section 10.2.7', Appendix 2) [471]. 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 'Section 10.2.7', Appendix 2). 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 IMPACT World+ is that reported by de Baan et al. (2013b): 3,279 terrestrial species [19], [472]; a value is derived from the WWF WildFinder database and the work of G. Keir (2005) (Table A65, Appendix 5) [473], [474]. 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 $1.8x10^4$ m^2 (total roof area) over 100 years (total development lifetime) [339]. This description is then condensed; the units are combined into the form PDF.m^2.year, as required: (0.016) X ($1.8x10^4$) X (100). The modelled local terrestrial biodiversity impact is: $–2.91x10^4$ PDF.m^2.year.

Inter-species 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 3, Supplementary Information), with a lower species richness than perhaps might be observed [475]. 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.

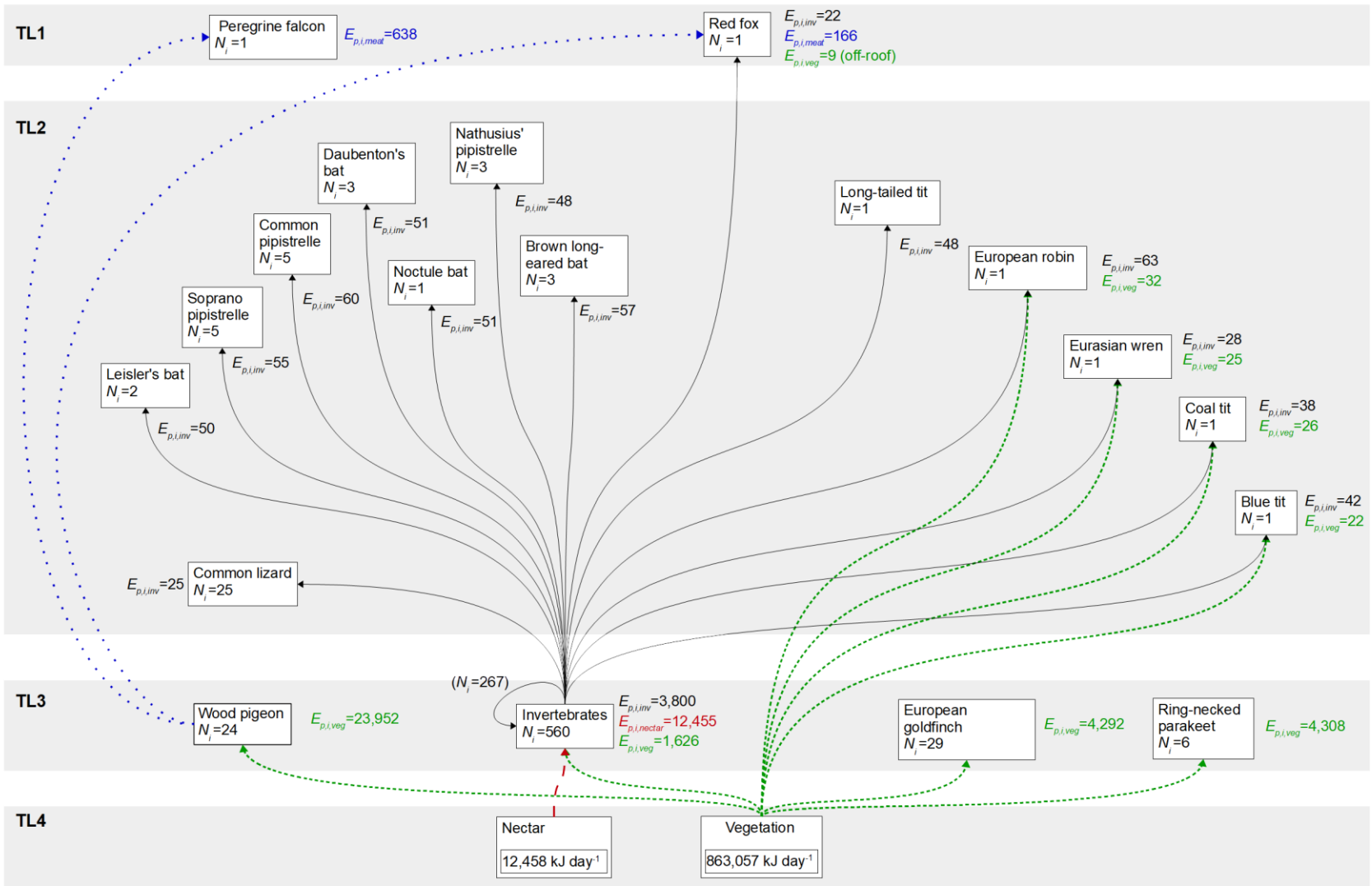

Trophic levels (TL)
TL1
TL2
TL3
TL4
Peregrine falcon
N_i=1
$E_{p,i,meat}$=638
Red fox
N_i=1
$E_{p,i,inv}$=22
$E_{p,i,meat}$=166
$E_{p,i,veg}$=9 (off-roof)
Daubenton's bat
N_i=3
$E_{p,i,inv}$=51
Nathusius' pipistrelle
N_i=3
$E_{p,i,inv}$=48
Common pipistrelle
N_i=5
$E_{p,i,inv}$=60
Noctule bat
N_i=1
$E_{p,i,inv}$=51
Brown long-eared bat
N_i=3
$E_{p,i,inv}$=57
Soprano pipistrelle
N_i=5
$E_{p,i,inv}$=55
Leisler's bat
N_i=2
$E_{p,i,inv}$=50
Long-tailed tit
N_i=1
$E_{p,i,inv}$=48
European robin
N_i=1
$E_{p,i,inv}$=63
$E_{p,i,veg}$=32
Eurasian wren
N_i=1
$E_{p,i,inv}$=28
$E_{p,i,veg}$=25
Coal tit
N_i=1
$E_{p,i,inv}$=38
$E_{p,i,veg}$=26
Blue tit
N_i=1
$E_{p,i,inv}$=42
$E_{p,i,veg}$=22
Common lizard
N_i=25
$E_{p,i,inv}$=25
(N_i=267)
Invertebrates
N_i=560
$E_{p,i,inv}$=3,800
$E_{p,i,nectar}$=12,455
$E_{p,i,veg}$=1,626
Wood pigeon
N_i=24
$E_{p,i,veg}$=23,952
European goldfinch
N_i=29
$E_{p,i,veg}$=4,292
Ring-necked parakeet
N_i=6
$E_{p,i,veg}$=4,308
Nectar
12,458 kJ day^{-1}
Vegetation
863,057 kJ day^{-1}

Figure 4.6 Energy flow diagram describing the number of individuals supported by the sedum roof (N_i , individuals) and the interactions taking place between them. The energy requirements of each population of consumer/predator (of population size 'N_i'), $E_{p,i\,[...]}$ ([individuals of species j consumed] [predator population (species i)]$^{-1}$ day^{-1}) are also described: $E_{p,i,nectar}$ corresponds to the consumption of nectar by each population of consumers (dashed red line); $E_{p,i,veg}$ corresponds to the consumption of plant biomass by each population of consumers (dashed green lines); $E_{p,i,inv}$ corresponds to the consumption of invertebrates by each population of consumers (solid black lines); and $E_{p,i,meat}$ corresponds to the consumption of birds and mammals by each population of consumers (dotted blue lines). The grey areas (background) indicate trophic level grouping, numbered (left) from the highest trophic level (TL1) to the lowest (TL4)

4.3.2 Combined LCA and biodiversity results

The lifetime impacts of Meridian One on ecosystem quality under both design scenarios are presented in Fig. 4.7. 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 ($5.46x10^{10}$ PDF.m^2.year) is 15% less than developing Meridian One with concrete roofs ($6.39x10^{10}$ 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: $2.94x10^6$ 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 towards its reference state [339]. 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 (Fig. 4.7).

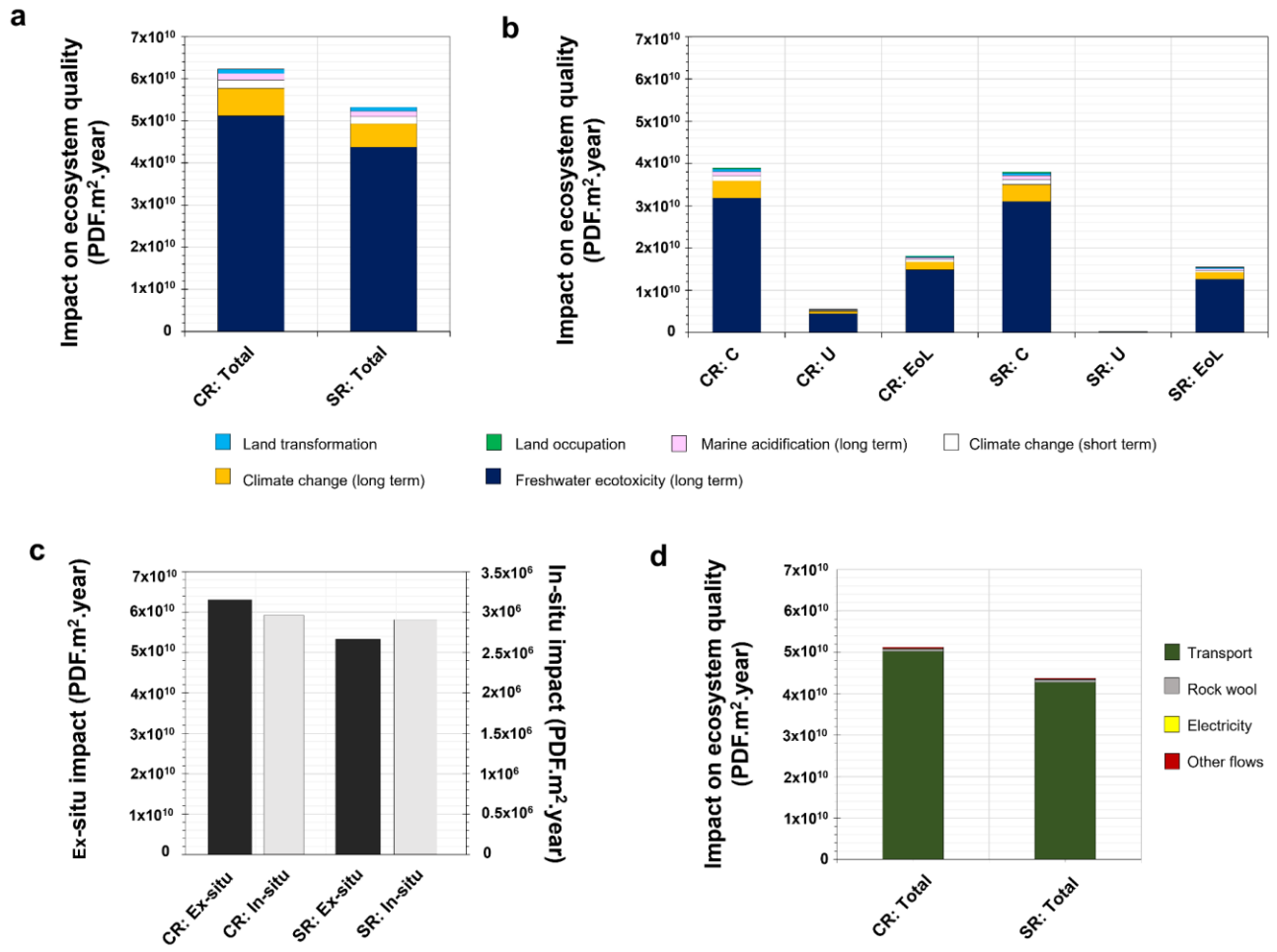


Figure 4.7 Endpoint impacts on ecosystem quality associated with the development of 18 residential buildings at Meridian One with either concrete roofs ('CR') or sedum roofs ('SR') over its 100-year lifetime. In (**a**), the total life cycle impacts of both roof scenarios (concrete and sedum roofs) on 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 ('U'), Use ('U'), and End-of-Life ('EoL'). (**c**) illustrates the contribution of ex-situ (black, primary (left) axis) and in-situ (grey, 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 (grey), 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

In all cases, the greatest contributor to ex-situ impacts at Meridian One is the endpoint impact category 'freshwater ecotoxicity, long term' (Fig. 4.7). Freshwater ecotoxicity describes biodiversity loss caused by the pollution and subsequent degradation of aquatic ecosystems [476], often resulting from stressors like pesticides and metals entering freshwater ecosystems [477], [478]. Over the lifetime of Meridian One, 98% of freshwater ecotoxicity is attributed to transport, predominantly via the emission of metals (including aluminium, copper, and iron) to water (Fig. 4.8). Transport contributes most to freshwater ecotoxicity in all life cycle stages except the use phase in the sedum roofs scenario (Fig. 4.8g). This discrepancy reflects the fact that less transport is required in the use phase of the sedum roof scenario; vehicle use in the sedum roofs use phase ($2.04x10^7$ t.km) is two orders of magnitude less than in the concrete roofs use phase ($6.68x10^9$ t.km). 92% of freshwater ecotoxicity is instead attributed to electricity – utility use by residents during the apartments' occupation.

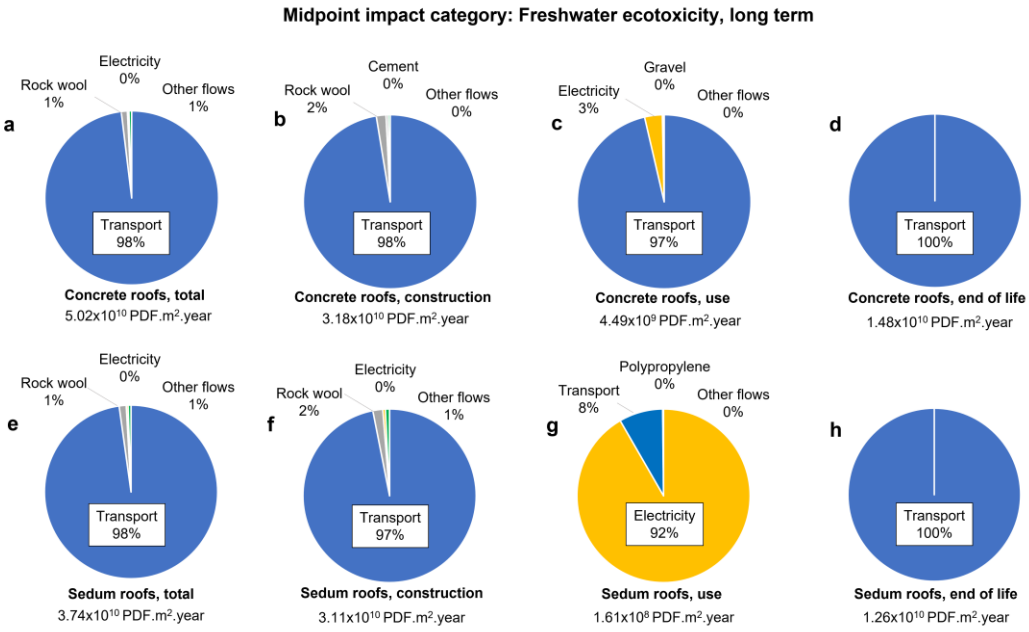


Figure 4.8 Contributions of process flows to 'Freshwater ecotoxicity, long term' in the concrete roofs design scenario (**a-d**) and the sedum roofs design scenario (**e-h**)

The fact that most of the development's life cycle impacts relate to freshwater ecotoxicity raises concerns surrounding ecological equivalence in biodiversity offsetting [479], [480]. 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 modelled. 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-neck parakeet in this case – would not count towards the restoration of reference-state conditions.

4.3.3 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. We model a ±25% change in transportation distances, and the use of either '7.5-16 tonne' or '>32 tonne' lorries. Fig. 4.9 describes the effect of changing these parameters on both the overall life cycle ecological impact and the midpoint impact category 'freshwater ecotoxicity, long term', for both the concrete and sedum roofs scenarios. The results are presented as percentage change compared to the case study results.

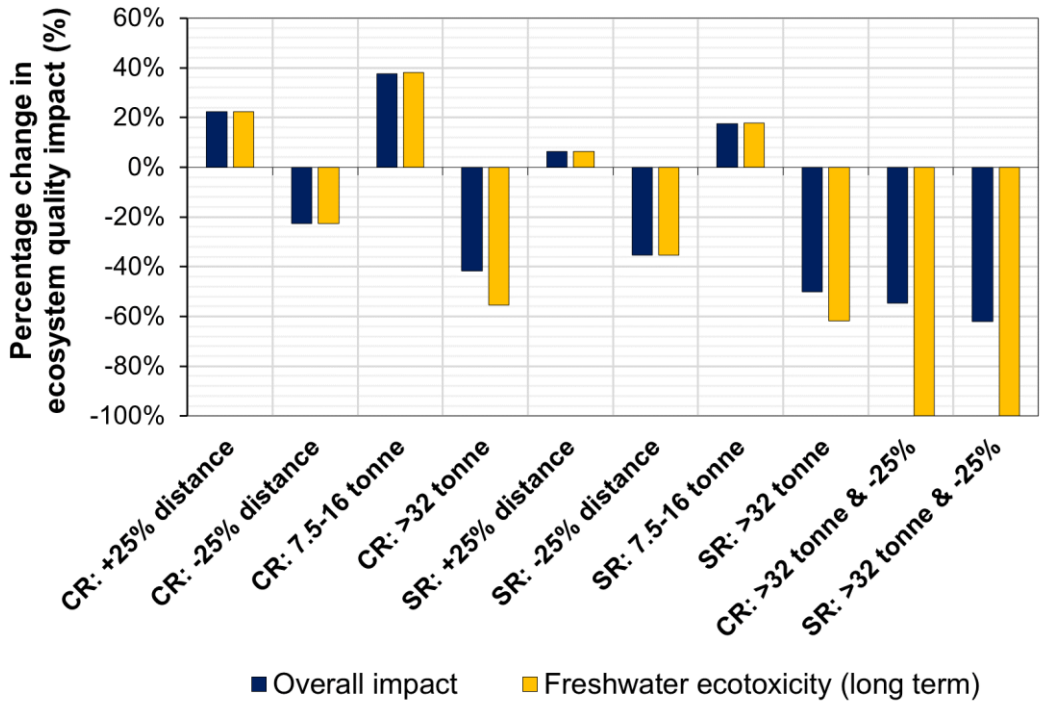


Figure 4.9 Sensitivity analysis of life cycle impacts on ecosystem quality when developing Meridian One with concrete roofs (CR) or sedum roofs (SR), varying the transportation distance (±25%) and vehicle type, 7.5-16 tonne and >32 tonne. A combined, best-case scenario is also presented: >32 tonne vehicle 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

4.3.3.1 Effects of transportation distance 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 favourable 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%.

Fig. 4.9 illustrates that the best-case is: >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.

4.3.3.2 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.

4.3.4. Describing an endpoint characterisation factor

Using IMPACT World+ methodology (Section 4.2.5), we calculate the relative impact of land occupation on ecosystem quality, $I_{occupation,LUa,b}$, (PDF.m^2.year) for the sedum roofs in this case study to be –2.91x10^4 PDF.m^2.year. Therefore, using Eq.(4.5), 'Section 4.2.5', the endpoint characterisation factor of occupation for the case study, which we term 'urban, sedum roof, London (UK)' ($CF_{occupation,\ urban,\ sedum\ roof,\ London\ (UK)}$) is –1.6x10^{-2} PDF.m^2.year (m^2.a)$^{-1}$.

4.4 Perspectives

We applied our methodology, to quantify the local ecological impact of land occupation related 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 [23]. 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 modelling 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 characterisation 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 modelled explicitly in the first instance, characterisation will likely be restricted to small spatial scales (i.e., meso-scale: community level within a forest, city, or lake [481], [482]) and limited gena until ecosystem dynamics are quantitively 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 generalisation 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 [483]. 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 north-western Europe [435]. However, this is contingent on local species pools being comparable to that in this case study. If dissimilar, the local biodiversity impact should be modelled 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.

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 [339]. 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 [339]. 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.

Next, aggregating species data across an entire year would produce characterisation 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 idealised scenario. Urban areas typically comprise many, heterogenous 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. Finally, there is the question of inventory; there is a need to define a new category of environmental pressures so that the characterisation factors described in this chapter may be incorporated into the LCA method. Furthermore, recent works in 'metabolic theory', exemplified by [362], [363], [367], [484], offer alternative means of modelling energy flow through trophic systems. Metabolic theory, particularly methods to model energy distribution through interspecific consumption – which is assumed uniform in this case study – provide greater ecological insight to enhance the results of subsequent case studies.

4.4.1 Including biodiversity-supporting infrastructure and other biodiversity-supporting components in life cycle impact assessment (LCIA)

The need to define a new environmental pressure to facilitate inventory of biodiversity-supporting components within LCIA was identified. The creation of a new category 'biodiversity-supporting components' in LCIA is recommended. It should be ensured that this inventory is comprehensive, to facilitate more accurate life cycle assessments. This requires all biodiversity-supporting components (infrastructures and land/surface cover) are considered – something currently overlooked in LCA applications. Fig. 4.10a illustrates a development with several biodiversity-supporting components. Fig. 4.10b illustrates the inventory of components contributing to local biodiversity impacts through land use. In contrast, Fig. 4.10c illustrates inventory of the same development (Fig. 4.10a) including the

proposed new category, 'biodiversity-supporting components' alongside land use. Fig. 4.10c represents a more comprehensive inventory of local biodiversity impacts.

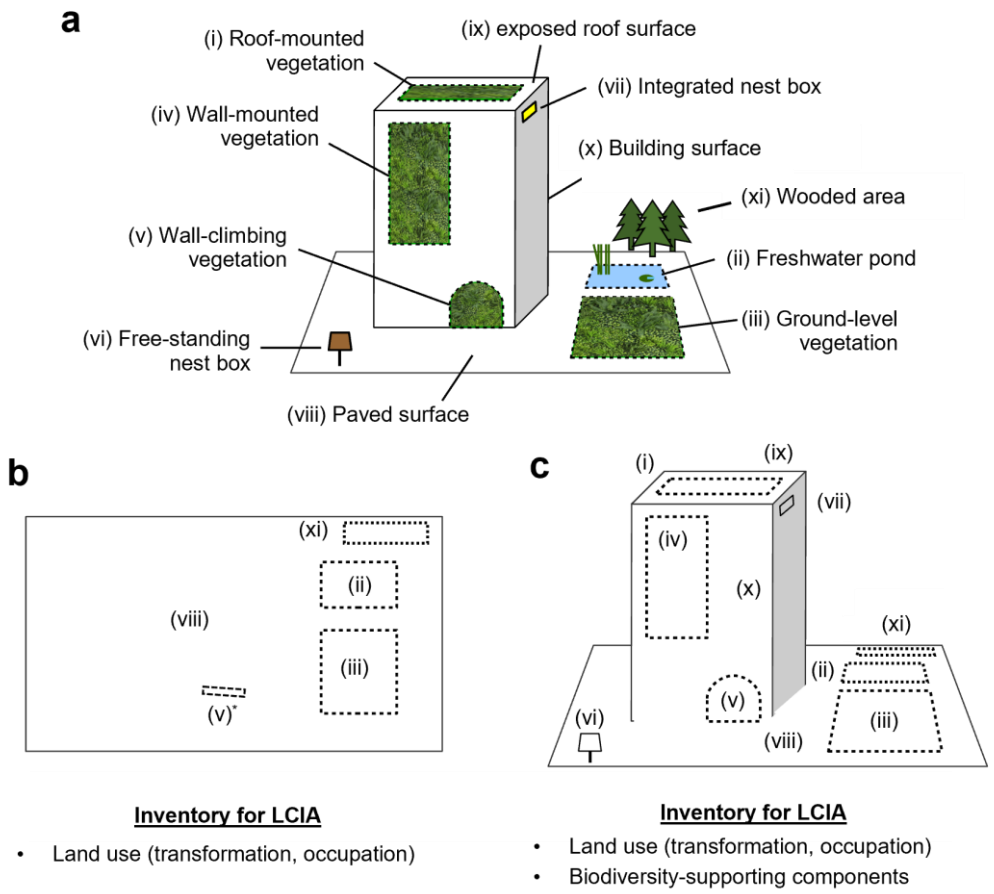


Figure 4.10 Inventorying local biodiversity impacts at a residential development. (**a**) illustrates a site comprising a residential building and 11 distinctive types of land occupation: (i) green roof, (ii) freshwater pond, (iii) ground-level vegetation (e.g., gardens), (iv) green wall, (v) wall-climbing, ground-planted vegetation, (vi) free-standing nest box, (vii) integrated nest box, (viii) paved surface (surrounding the residential building), (ix) exposed roof surface, (x) building surface (e.g., cladding), and (xi) a wooded area. Every component could support biodiversity to some degree. (**b**) and (**c**) illustrate two modes of inventory. (**b**) inventories land use (transformation and occupation), whereas (**c**) also inventories a new category of local biodiversity impact: 'biodiversity-supporting components'. (**b**) fails to capture all biodiversity-supporting components on-site or, in the case of (v)*, in a misleading way (ground level cover is not the bio-productive surface area). Intuitively, (**c**) facilitates a more accurate life-cycle assessment of the development's ecological impacts

For green infrastructure, the area inventoried relates to the components' primary productivity (nectar or bio-mass production). Similar definitions would be put forward for biodiversity-supporting infrastructures not comprising vegetation (e.g., bare surfaces, feeding tables, artificial roosts including 'insect hotels') on a case-by-case basis. Inventorying roosts is anticipated to be the least intuitive. This recommendation facilitates the application of the ecological needs concept in LCIA through characterisation. It also allows a more comprehensive inventory to be compiled for more accurate net

local life cycle ecological impact assessments. This will produce more robust and ecologically-supportive human intervention.

5 Quantifying the land use, energy requirements, and rates of water consumption of wild species in terrestrial ecosystems: A case study of the American kestrel (*Falco sparverius*) on St Martin Island, Caribbean

This chapter uses the ecological needs equations proposed in Chapter 3 alongside the method described in Chapter 4 to quantify the natural resource requirements of an ecological system, at the community level, pertaining to the American kestrel (*Falco sparverius*) on St Martin Island. The aim of this chapter is hence to demonstrate the methodology using a case study for a well quantified food web. The underlying ecological data describing species presence and rates of predation (observed interspecific interactions) are from: L. Goldwasser and J. Roughgarden (1993), 'Construction and Analysis of a Large Caribbean Food Web', *Ecology*, 74(4), 1216–1233 [87].

This chapter represents the culmination of previous innovation and learning in this thesis. The work in Chapter 3 and Chapter 4 makes this research possible, in providing a means by which ecological needs can be quantified, ecological systems and the energy flows therein can be explicitly modelled, and in identifying aspects requiring further consideration when quantifying natural resource use at the (ecological) system level. The interconnection of Chapters 3, 4, and 5 are illustrated in Fig. P.1 as part of the Preface.

5.1 Introduction

In ecology, food webs describe interspecies interactions – acts of predation and consumption (e.g., herbivory) between different organisms – and thus describe qualitatively the flow of energy through ecological systems (Fig. 5.1). First modelled in the late 19th century, the concept of food webs was pioneered by Charles Elton in the 1920s and is now well established [485], [486]. However, debates are ongoing about how best to model interspecies interactions, particularly energy flow, though food webs [487], [488], [489]. The complexity of ecological systems makes producing comprehensive food webs difficult [486], [490], [491]. Consequently, food webs typically describe interspecies interactions within defined temporospatial parameters and are often limited to the community level. In 1972, Robert May attempted to model the co-variability of species within an ecological system to determine its stability [492], leading to the inclusion of quantitative metrics like species composition (diversity of species), species abundances (numbers of individuals), stability, and metabolic and nutrient efficiencies to

produce ‘quantitative food webs’ and facilitate their analysis [189], [219], [460], [493]. This includes the inference of population dynamics – describing what species make-up ecological systems and how those species interact in different contexts – where there is no empirical data, and the prediction of future dynamics as part of ‘theoretical ecology’: the use of theorical rather than empirical data in the production and application of food webs [494].

A pertinent type of quantitative food web is the ‘energy flow web’, which describes the energy flows associated with intraspecific interactions within an ecological system. The ability to estimate species’ metabolic requirements through allometry represents an important means of producing energy flow webs, which themselves now represent a means of predicting the effect of changing ecological system dynamics (e.g., due to human intervention) on different ecosystem functions. Considerable datasets of qualitative (e.g., ECOWeB v.1.1. [495]) and quantitative (e.g., International Institute for Applied Systems Analysis (IIASA) [496]) food webs are published, reflecting considerable ecological knowledge in this field. However, there are opportunities to further supplement food webs to produce more comprehensive models of resource use within ecological systems and in doing so, describe quantitatively the natural resource requirements of different species under different temporospatial and ecological contexts.

Therefore, this study demonstrates how the ecological needs equations described in Table 3.1. Chapter 3 and method in Chapter 4 can be used to build-upon quantitative food web data: to quantify the natural resource (land, water, and energy) requirements of a species, within a defined trophic system, and describe ecological systems more comprehensively. In this case, for the food web associated with the American kestrel (*Falco sparverius*) on St Martin Island (henceforth referred to as ‘kestrel’). Hence this chapter aims to increase our understanding of that ecological system. It has value across ecology and engineering by demonstrating a method to inform natural resource management across socioeconomic and ecological systems. The results can guide human interventions to support the American kestrel and the species that underpin it. For example, for their conservation or the (re)introduction or translocation of the kestrel in the observed ecological system.

In the context of life cycle assessment, the quantitative results – the natural resource requirements of the ecological system – can be used to quantify potential impacts of different forms of land use change and changes to local water sources on the ecological system: by relating the metabolic (energy) requirements of the system to potential bio-productivities of different land use categories (e.g., urban, agricultural, forest) and by relating the water requirements of the system by the ecological system

to changes in freshwater quality and quantity within the system's reach, respectively. . This allows the value of different land use types for conserving the kestrel on St Martin Island to be determined. It also facilitates more sustainable urban development via the design and assessment of biodiversity-supporting infrastructure.

5.2 Method

This study uses a subset of data from Goldwasser and Rougharden [87]. Their paper described a large Caribbean food web centred around five apex bird species: American kestrel (*Falco sparverius*), scaly-breasted thrasher (*Allenia fusca*), pearly-eyed thrasher (*Margarops fuscatus*), gray kingbird (*Tyrannus dominicensis*), and yellow warbler (*Setophaga petechia*). This chapter uses the part of the food web pertaining to the American kestrel (Fig. 5.1). Taxonomic and dietary data were presented for 44 species, including frequencies of interspecies predation ([acts of predation] hectare^{-1} day^{-1}) that described how each species contributed, as prey, to their predators' diets. However, the food web Goldwasser and Roughgarden produced lacked quantitative details about the ecological system including relative (to the apex species) species abundances, and the quantity of energy and other resources used (i.e., required) and produced (metabolised) by the system and its constituent species. Here, the method used to construct the Meridian One food web in Chapter 4 is applied, with some refinements (identified during that study, e.g., including seasonal increases in species' metabolic requirements like that associated with the moulting of feathers in birds feather, and defined fully here), to produce a more comprehensive, quantitative food web.

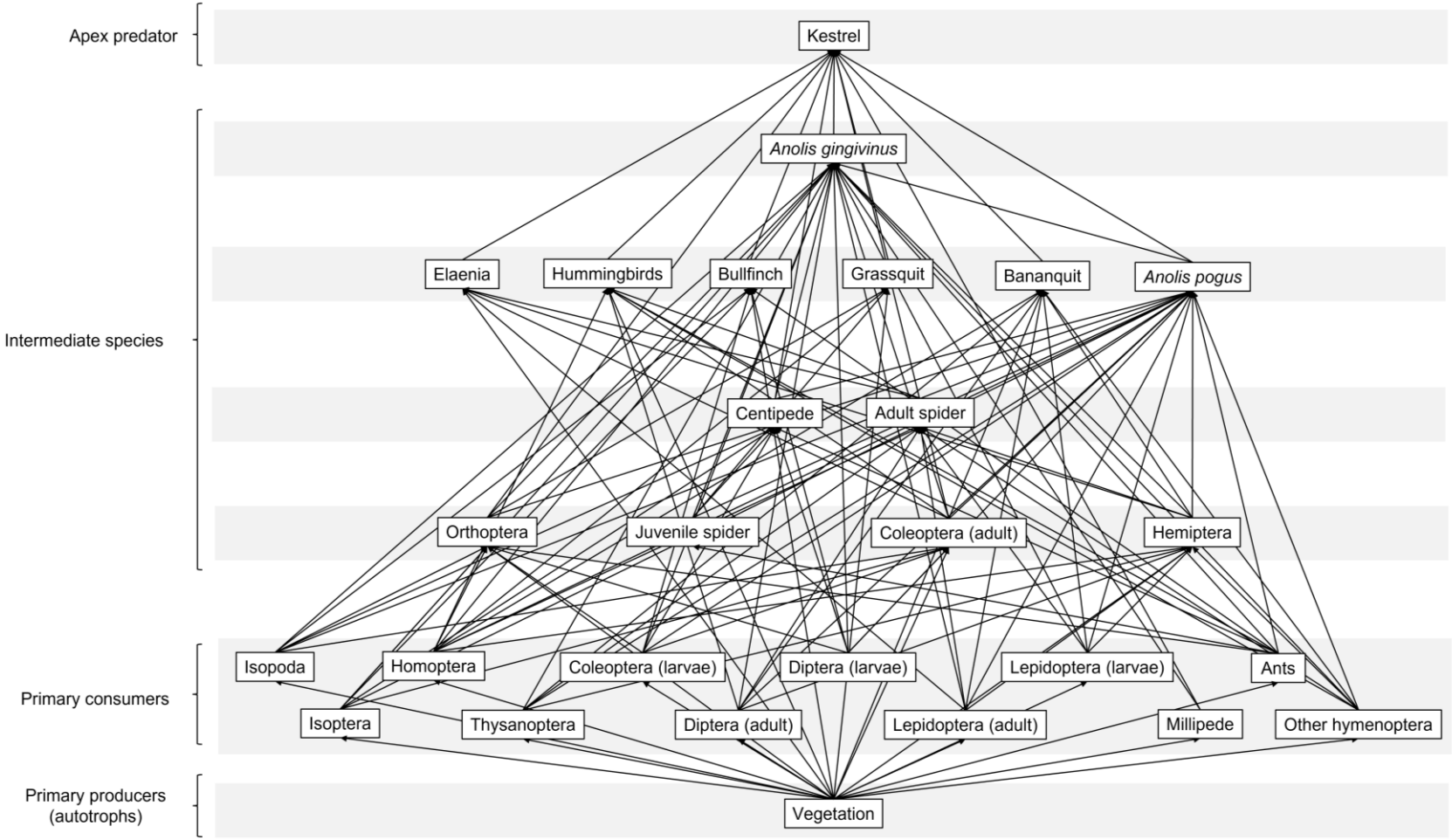


Figure 5.1 Interspecies interactions related to the American kestrel (*Falco sparverius*) on St Martin Island in the Caribbean, based on published work by Goldwasser and Roughgarden (1993). The system is disaggregated by trophic levels following the flow of energy from the bottom trophic level, upwards: (i) vegetation (autotrophs), (ii) primary consumers (i.e., herbivores), (iii) intermediate species, and (iv) the apex predator: the American kestrel (Falco sparverius). To guide the reader, the interactions comprising Fig. 5.1. are presented in tabular form across Tables A6, Appendix 1

Fig. 5.1 describes the ecological system being quantified. With so many interactions between the 25 species (plus 'vegetation') making-up this food web, individual connections are obscured. To guide the reader, the interactions comprising Fig. 5.1. are presented in tabular form in Table A6, 'Section 10.3.1', Appendix 1.

The method used to quantify the resource needs of the ecological system is as follows:

1. The described frequencies of interspecies predation ([acts of predation] km^{-2} day^{-1}) are adjusted using the land use requirement of each consumer (e.g., predator species), L_i (km^2 [individual predator]$^{-1}$) (Eq. 5.1), which are calculated using the corresponding ecological needs equations for land use (Table 3.1, Chapter 3). The adjusted values represent the contribution of each consumed (e.g., prey) species to their consumer's diet within their expected land use range. The units become: ([acts of predation] [individual predator]$^{-1}$ day^{-1}). Here $(predation\ frequency\ per\ km^2)_{i,j}$ is the scaled predation frequency of prey species '*j'* consumed by predator '*i*' (see Table A68, Appendix 6).

$$\text{scaled predation frequency of species } i = (\text{predation frequency per km}^2)_{i,j} \times L_i \qquad (5.1)$$

2. Each value is then converted into 'percentage contribution' (Eq. 5.2) to describe how consumed species contribute to each consumer's diet. Here, $contributuion_{i,j}$ is the contribution of consumed species '*j*' to the diet of consumer species '*i*' as a percentage. $\Sigma(predation\ frequency\ per\ km^2)_j$ is the sum of all consumed species' scaled predation frequency for consumer species '*i*' (see Table A69, Appendix 6).

$$\text{contribution}_{i,j} \text{ as } \% = 100 \times \frac{(\text{predation frequency per km}^2)_{i,j}}{\sum(\text{predation frequency per km}^2)_j} \qquad (5.2)$$

3. The contribution of consumed species to consumer species' diets is then described in terms of energy (kJ [individual consumer]$^{-1}$ day^{-1}) using the energy requirements of each consumer species, E_i (kJ [individual consumer]$^{-1}$ day^{-1}) (Eq. 5.3), which are calculated using the corresponding ecological needs equations for energy requirement (Table 3.1, Chapter 3). However, E_i describes the 'energy required' by an individual at this stage and not the 'energy consumed' by an individual.

To convert 'energy required' by a species acting as the consumer, E_i (kJ individual^{-1}), into 'energy consumed' by that consumer, $E_{c,i}$ (kJ individual^{-1}), all species' metabolic efficiencies, η_i (%) and seasonal increases in species' metabolic rates, $E_{var,i}$ (%), must be accounted for (Eqs (5.3) – (5.4)). Here, seasonal increases in species' metabolic rates are limited to birds: the result of moulting (of feathers) and breeding/chick-rearing. Here, *contribution$_{i,j}$ as energy* is the contribution of consumed species '*j*' to the diet of consumer species '*i*' in terms of energy (i.e., kJ) (see Table A70, Appendix 6).

$$\text{contribution}_{i,j} \text{ as energy} = (\text{contribution}_{i,j} \text{ as } \%) \times E_{c,i} \tag{5.3}$$

Where:

$$E_{c,i} = E_i \times \eta_i \times E_{\text{var},i} \tag{5.4}$$

Metabolic efficiency describes what proportion of consumed energy is metabolised into usable energy. Owing to a lack of species-specific data on metabolic efficiencies, assumptions are made the same as in Chapter 4: $\eta_{carnivory}$ and $\eta_{omnivory}$: 0.75, and $\eta_{herbivory}$: 0.59%. It is assumed that the bird species experience a 9.8% increase in metabolic rate [497] throughout the moulting period and that this period lasts for 4 months for the kestrel, and 5 weeks for the smaller bird species in this study [498]. It is assumed that the bird species moult once per year. Regarding breeding/chick rearing: it is assumed that bird species experience an 11% increase in metabolic rate [499] during this period and that this period lasts on average 5 months for all bird species [500]-[511].

4. The contribution of each consumed species, as 'energy consumed', are next converted into 'individuals equivalent' using assumed calorific values of each consumed species as food, c_j (kJ kg^{-1}) (Eq. 5.5). This requires the adult body mass of consumed species, M_i (kg individual^{-1}), and the aggregated calorific values of them as food: 'meat' (5,767 kJ kg^{-1}) and 'invertebrates' (20,021 kJ kg^{-1}), as calculated in Chapter 5 (Appendix 3). These calorific values are used owing to a lack of species-specific calorific data. Here, *contribution$_{i,j}$ as 'individuals equivalent' (N_e)* is the contribution of consumed species '*j*' to the diet of consumer species '*i*' in terms of 'individuals equivalent' (see Table A71, Appendix 6). N_e is used here to denote 'individuals equivalent'.

$$\text{contribution}_{i,j} \text{ as 'individuals equivalent' } (N_e) = \frac{\left(\text{contribution}_{i,j} \text{ as energy}\right)}{M_i \times c_i} \tag{5.5}$$

For example, if a species of bird of adult body mass (M_i =) 10 g comprised 1,000 kJ kg^{-1}, 1,000 calories worth of that bird species is equivalent to one individual-worth of energy, i.e., 'one individual equivalent'.

5. The flow of energy across the entire trophic system (Fig. 5.1) is then described using a top-down fashion, starting at the kestrel. This method is the same as described in 'Section 4.2.5.2', Chapter 4. The full working is provided in Table A73, Appendix 6. Once completed, species abundances and energy flows (from both predation and herbivory) are explicitly quantified (see 'Section 5.3. Results').

6. With species abundances, N_e ([individuals equivalent] $system^{-1}$), now quantified, the water consumption of each population of species comprising the food web, $W_{p,i}$ (m^3 [population of N_e indviduals]$^{-1}$ day^{-1}), and the water consumption of the entire ecological system, W_s (m^3 $system^{-1}$ day^{-1}), can be quantified (Eq. 5.6 and Eq. 5.7, respectively) using the corresponding ecological needs equation for water consumption (Table 3.1, Chapter 3).

$$W_{p,i} = W_i \times N_e \tag{5.6}$$

$$W_s = \sum W_{p,i} \tag{5.7}$$

Together, the requirements for land, energy, and water constitute the natural resource requirements of the ecological system: the kestrel and the species that underpin it.

7. In another amendment to the method in Chapter 4, the results are scaled from 'daily' to 'lifetime' energy requirements based on a conservative estimation of a (wild) kestrel lifetime: five years (1,825 days) [512]. This is done by multiplying the *contribution$_{i,j}$ as 'individuals equivalent'* values (Table A71, Appendix 6) by 1,825 days, thus scaling energy per day to energy per 'kestrel lifetime' (results in Table A72, Appendix 6). The energy flows across the entire system are then re-calculated (Table A80, Appendix 6). Changes to the system's dynamics will likely occur, due to instability or new or existing environmental pressures for example, but these are not included at this stage. The system is assumed stable throughout the kestrel's anticipated lifetime.

5.3 Results and discussion

5.3.1 Land use requirement of the ecological system

The land use requirement of the kestrel represents the spatial confines of the trophic system (the kestrel and the species underpinning it, Fig. 5.1). Using the ecological needs equation for land use for carnivorous birds from Chapter 3 (Eq. 3.14), and assuming an adult body mass of 0.163 kg [411], the land use requirement of the kestrel, $L_{kestrel}$, is calculated to be 0.32 km^2 (individual [kestrel])$^{-1}$. Thus, the land use requirement of the trophic system, $L_{s,}$ is 0.32 km^2 system^{-1}.

$$L_{carnivorous,bird} = \left(0.65 \times M_i^{-0.868}\right)^{-1} \tag{5.8}$$

5.3.2 Energy and water requirement of the ecological system

5.3.2.1 Daily requirements

Sankey diagrams are produced to illustrate the flows of energy (Fig. 5.2) and water (Fig. 5.3) – of adequate type (i.e., freshwater) and quality (i.e., drinkable) – across the ecological system over a period of one day. The diagrams are supplemented with the following calculated results:

- $N_{e,j}$, 'energy as species richness (number of individuals) of consumed species *j* equivalent': the flow of energy from the consumed species, *j*, to the consumer species, *i*, as 'individuals' based on the calorific value of that consumed species: c_j (kJ individual^{-1}).
- $E_{c,p,i}$ (kJ [population of $N_{e,j}$ indviduals]$^{-1}$ day^{-1}), the energy consumed by the population (of population size $N_{e,j}$) of species *i.*
- $W_{p,i}$ (m^3 [population of $N_{e,j}$ indviduals]$^{-1}$ day^{-1}), water consumption (ingested, excluding water intake from food) of each population (of population size $N_{e,j}$) of species *i*.
- $E_{veg,c,s}$ (kJ system^{-1} day^{-1}), the energy requirement of the ecological system from vegetation over time as 'consumed', i.e., accounting for $E_{var,i,}$ (kJ system^{-1} [time]$^{-1}$). This represents a required rate of bio-productivity.
- W_s (m^3 system^{-1} day^{-1}), the rate of water provision needed to sustain the entire trophic system. For consistency.

For consistency, the system's trophic levels are identified in Fig. 5.2 as they are in Fig. 5.1 (the kestrel food web).

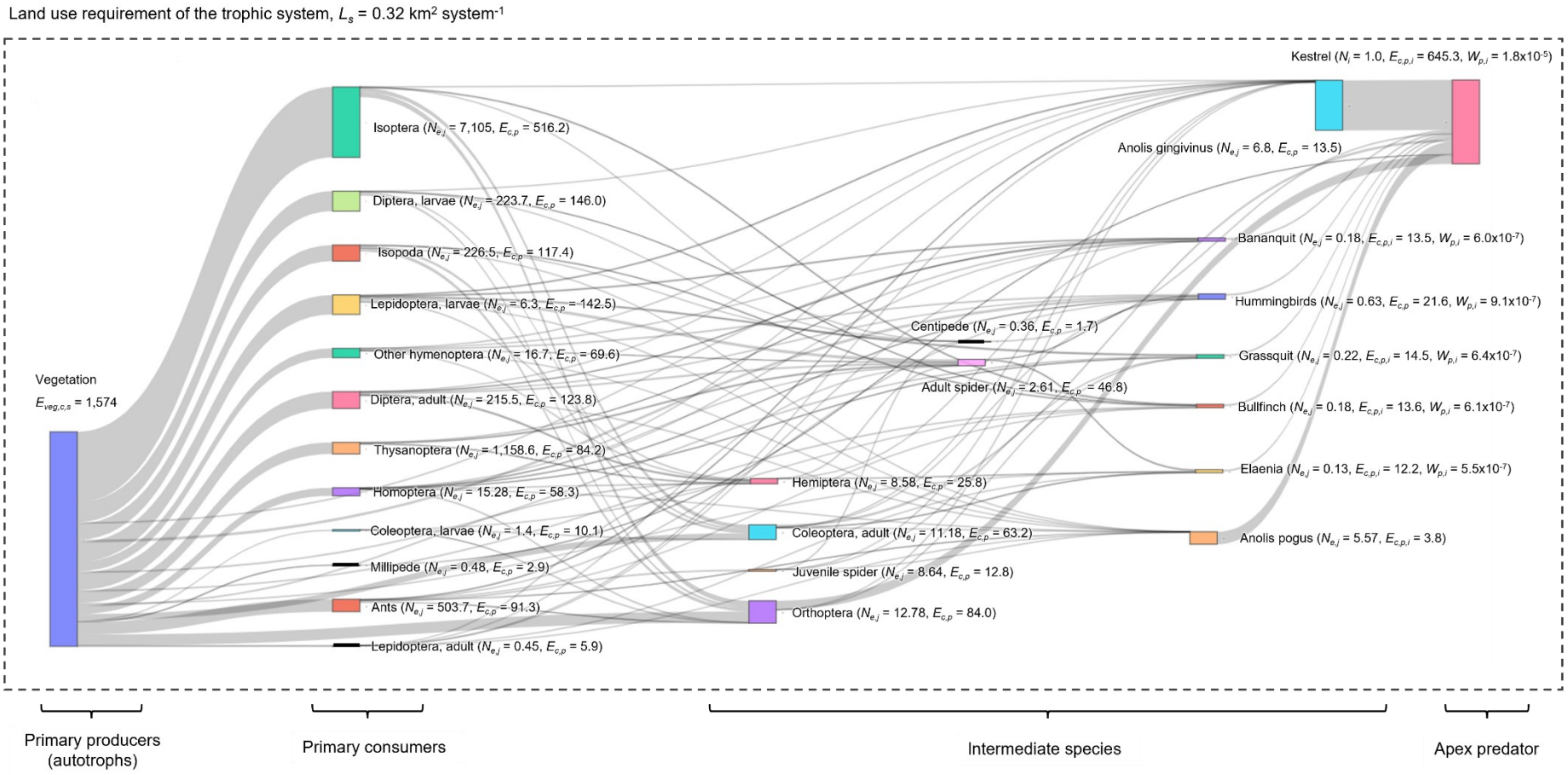


Figure 5.2 The flow of energy between species within the kestrel food web (Fig. 6.1), with one kestrel (N_i = 1.0), over a period of one day (i.e., daily flows within the ecological system). L_s (km² system^{-1}) is the land use requirement of the trophic system, represented by a dashed border, $N_{e,j}$ is 'energy as species richness (number of individuals) of consumed species j equivalent', i.e., the flow of energy from the consumed species, j, to the consumer species, i, as 'individuals' based on the calorific value of that consumed species: c_j (kJ individual^{-1}); $E_{c,p,i}$ (kJ [population of $N_{e,j}$ indviduals]$^{-1}$ day^{-1}) is the energy requirement of each population (of size $N_{e,j}$) of each consumer species, i, accounting for their metabolic efficiencies and seasonal variations in metabolic rate ($E_{var,i}$); and $W_{p,i}$ (m³ [population of $N_{e,j}$ indviduals]$^{-1}$ day^{-1}) is the water consumption of each population (of size $N_{e,j}$) of each species. $E_{veg,c,s}$ (kJ system^{-1} day^{-1}) is the rate of bio-productivity (vegetation production) needed to sustain the ecological system

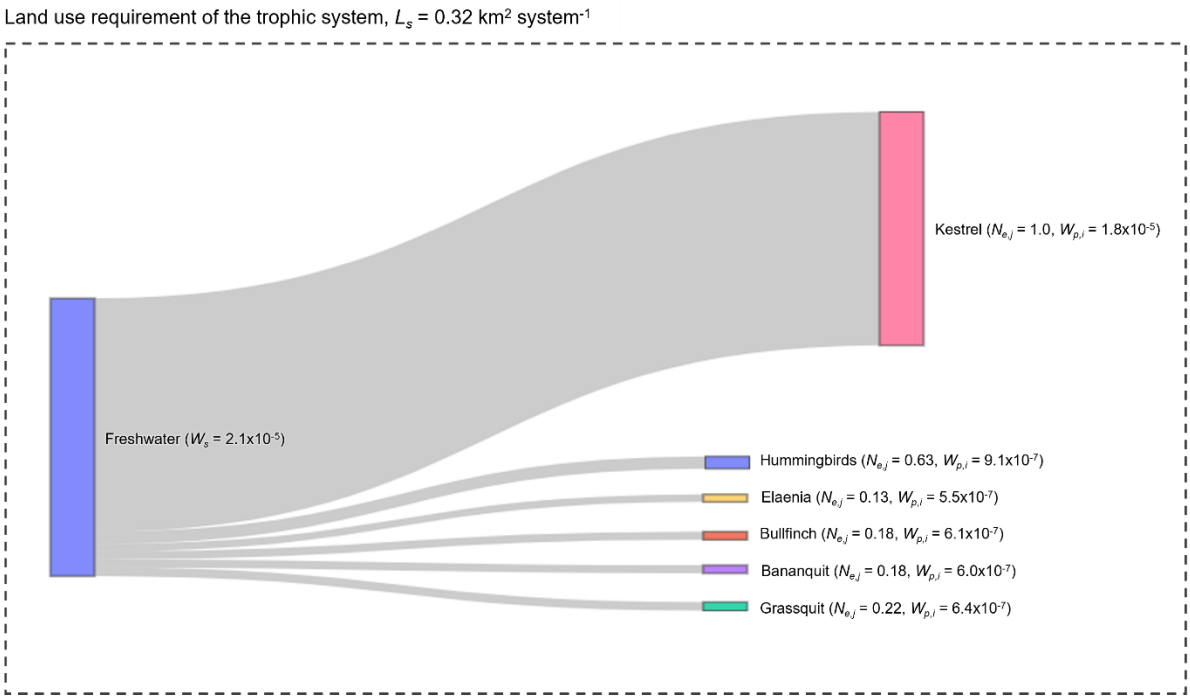


Figure 5.3 The rate of freshwater consumption between species within the kestrel food web (Fig. 6.1), with one kestrel, over a period of one day (i.e., daily flows within the ecological system); and the land use requirement of the trophic system, L_s (km^2 system^{-1}), represented by the dashed border. Invertebrates and reptiles, which have a zero-value for water consumption, are omitted. Here, $N_{e,j}$ is 'energy as species richness (number of individuals) of consumed species j equivalent', i.e., the flow of energy from the consumed species, j, to the consumer species, i, as 'individuals' based on the calorific value of that consumed species: c_j (kJ individual^{-1}); $W_{p,i}$ (m^3 [population of $N_{e,j}$ indviduals]$^{-1}$ day^{-1}) is the water consumption of each population (of size $N_{e,j}$) of each species; and W_s (m^3 system^{-1} day^{-1}) is the rate of freshwater provision needed to sustain the ecological system

The data (results) used to construct Fig. 5.2 and Fig. 5.3 are presented in Table 5.1, overleaf. For full results and working, the reader is directed to Appendix IV: Supplementary data for Chapter 5. The results are presented using Sankey diagrams. Used in lieu of conventional food web models (e.g., Fig. 5.1), Fig. 5.2 shows the direction of energy flows (through interspecies interactions) within the ecological system, as in conventional food webs, but also the relative quantify of energy involved in those interactions. The metabolic value of constituent species to their predators and within the systems is immediately conveyed. Similarly, the relative importance of local freshwater sources to different species within the ecological system can be identified in Fig. 5.3. Figs 5.2 – 5.3 clearly show that the dominant final consumer of energy and water in the described system is the American kestrel.

Table 5.1 Species abundances, N_i (individuals), energy requirements, E_i (kJ individual^{-1} day^{-1}), and rates of water consumption, W_i (m^3 individual^{-1} day^{-1}), of the American kestrel (*Falco sparverius*) and the species underpinning it on St Martin Island, Caribbean. Results are scaled to N_{kesrel} = 1.0 over one day (i.e., daily stocks and flows of the ecological system)

Species [a]	**Species abundance** $N_{e,j}$ **(energy as 'individuals equivalent')**	**Daily water consumption per species** [b, c] $W_{p,i}$ **(m^3 population^{-1} day^{-1})**	**Energy consumed per species** [b, d] $E_{c,p,i}$ **(kJ population^{-1} day^{-1})**				
			From vegetation	From other species			**Total**
				From vertebrates	From invertebrates	Total	
Kestrel	1.0	1.7x10^{-5}	-	53.77	91.49	1.70	645.27
Elaenia	0.1	5.5x10^{-7}	10.47	-	1.70	1.36	12.17
Bullfinch	0.2	6.1x10^{-7}	12.22	-	1.36	1.83	13.58
Grassquit	0.2	6.4x10^{-7}	12.68	-	1.83	2.20	14.51
Hummingbirds	0.6	9.1x10^{-7}	19.36	-	2.20	1.34	21.56
Bananquit	0.2	6.0x10^{-7}	12.11	-	1.34	17.48	13.46
Coleoptera (adult)	11.2	No data [e]	45.72	-	17.48	3.51	63.20
Orthoptera	12.8	No data [e]	80.52	-	3.51	1.71	84.03
Centipede	0.4	No data [e]	-	-	1.71	-	1.71
Millipede	0.5	No data [e]	2.89	-	-	13.48	2.89
Anolis gingivinus	6.8	No data [e]	-	0.01	13.48	3.84	13.48
Anolis pogus	5.6	No data [e]	-	-	3.84	-	3.84
Other hymenoptera	16.7	No data [e]	69.59	-	-	-	69.59
Diptera (adult)	215.5	No data [e]	123.75	-	-	-	123.75
Ants	503.6	No data [e]	91.27	-	-	-	91.27
Lepidoptera (adult)	0.5	No data [e]	5.90	-	-	-	5.90
Lepidoptera (larvae)	6.3	No data [e]	142.51	-	-	12.89	142.51
Hemiptera	8.6	No data [e]	12.89	-	12.89	-	25.79
Homoptera	15.3	No data [e]	58.34	-	-	46.80	58.34
Adult spider	2.6	No data [e]	-	-	46.80	-	46.80
Coleoptera (larvae)	1.4	No data [e]	10.14	-	-	-	10.14
Isopoda	226.5	No data [e]	117.40	-	-	-	117.40

Table 5.1 (continued)

Species [a]	Species abundance $N_{e,j}$ (energy as 'individuals equivalent')	Daily water consumption per species [b] $W_{p,i}$ (m^3 population^{-1} day^{-1})	Energy consumed per species [b, c] $E_{c,p,i}$ (kJ population^{-1} day^{-1})				
			From vegetation	From other species			Total
				From vertebrates	From invertebrates	Total	
Diptera (larvae)	223.6	No data [e]	146.01	-	-	-	146.01
Isoptera	7105.2	No data [e]	516.18	-	-	12.80	516.18
Juvenile spider	8.6	No data [e]	-	-	12.80	-	12.80
Thysanoptera	1158.6	No data [e]	84.17	-	-	1.70	84.17
Total		2.1×10^{-5}	1.6×10^{3}	553.8	212.4	766.2	2.3×10^{3}

[a] species named as listed in the data by Goldwasser and Roughgarden (1993) [87]. [b] 'water consumption' and 'energy requirement' are calculated using the corresponding ecological needs equations, described in Table 3.1, Chapter 3. [c] Water consumption describes only the water ingested by organisms and does not include water consumed as part of their diet (i.e., from prey) or as a by-product of metabolism [d] energy as number of individuals equivalent accounting for annual variations in metabolic rate due to moulting and breeding/rearing. [e] The water consumption of invertebrates is not yet quantified using the ecological needs equations.

5.3.2.2 Requirements of the ecological system over the kestrel's (assumed) lifetime

The stocks and flows of energy (as species and vegetation) and water across the kestrel's anticipated lifetime of five years (1,825 days) [512] are presented in Table 5.2. The same method is used to calculate the results in Table 5.2. (the kestrel-lifetime requirements of the system) as that used to calculate the results in Table 5.1. (the daily requirements of the system); the only change is the study time frame. It is found that while individual energy and water requirements of constituent species scale linearly with time ('per day' to 'per lifetime'), the cumulative energy and water requirements of the trophic system do not. This demonstrates that the natural resource requirements of trophic systems (rather than for individual animals) should be calculated on a case-by-case basis.

'Total water consumption' and 'total energy from vertebrates' only appears to scale with time across Table 5.1. and Table 5.2. because (i) the kestrel is dictating the population sizes of all species that contribute to water consumption, and (ii) the kestrel contributes >99% of the system's 'total energy from vertebrates' value (kestrel = 53.8 kJ [1 kestrel equivalent]$^{-1}$ day^{-1}, *Anolis gingivinus* = 0.01 kJ [6.8 *Anolis gingivinus* equivalent]$^{-1}$ day^{-1}).

Table 5.2 Species abundances, N_i (individuals), energy requirements, E_i (kJ individual^{-1} day^{-1}), and rates of water consumption, W_i (m^3 individual^{-1} day^{-1}), of the American kestrel (*Falco sparverius*) and the species underpinning it on St Martin Island, Caribbean. Results are scaled N_{kesrel} = 1.0 over the kestrel's anticipated lifetime: five years (1,825 days) (i.e., lifetime stocks and flows for the ecological system)

Species [a]	**Species abundance** $N_{e,j}$ **(energy as 'individuals equivalent')**	**Daily water consumption per species** [b, c] $W_{p,i}$ **(m^3 population^{-1} [kestrel lifetime]$^{-1}$)**	**Energy consumed per species** [b, d] $E_{c,p,i}$ **(kJ population^{-1} [kestrel lifetime]$^{-1}$)**				
				From other species			**Total**
			From vegetation	From vertebrates	From invertebrates	Total	
Kestrel	1.0	3.2x10^{-2}	-	1.01x10^6	1.67x10^5	1.18x10^6	1.18x10^6
Elaenia	245.8	1.0x10^{-3}	1.91x10^4	-	3.11x10^3	3.11x10^3	2.22x10^4
Bullfinch	330.1	1.1x10^{-3}	2.23x10^4	-	2.48x10^3	2.48x10^3	2.48x10^4
Grassquit	394.1	1.2x10^{-3}	2.31x10^4	-	3.34x10^3	3.34x10^3	2.65x10^4
Hummingbirds	1.14x10^3	1.7x10^{-3}	3.53x10^4	-	4.02x10^3	4.02x10^3	3.93x10^4
Bananquit	321.9	1.1x10^{-3}	2.21x10^4	-	2.45x10^3	2.45x10^3	2.46x10^4
Coleoptera (adult)	3.55x10^6	No data [d]	1.45x10^7	-	5.55x10^6	5.55x10^6	2.01x10^7
Orthoptera	8.40x10^5	No data [d]	5.29x10^6	-	2.31x10^5	2.31x10^5	5.52x10^6
Centipede	1.16x10^4	No data [d]	-	-	5.46x10^4	5.46x10^4	5.46x10^4
Millipede	9.79x10^3	No data [d]	5.84x10^4	-	-	-	5.84x10^4
Anolis gingivinus	1.25x10^4	No data [d]	-	9.8	2.46x10^4	2.46x10^4	2.46x10^4
Anolis pogus	1.12x10^4	No data [d]	-	-	7.72x10^3	7.72x10^3	7.72x10^3
Other hymenoptera	4.44x10^{10}	No data [d]	1.85x10^{11}	-	-	-	1.85x10^{11}
Diptera (adult)	3.41x10^{12}	No data [d]	1.96x10^{12}	-	-	-	1.96x10^{12}
Ants	2.55x10^{11}	No data [d]	4.62x10^{10}	-	-	-	4.62x10^{10}
Lepidoptera (adult)	1.16x10^9	No data [d]	1.52x10^{10}	-	-	-	1.52x10^{10}
Lepidoptera (larvae)	6.32x10^{10}	No data [d]	1.44x10^{12}	-	-	-	1.44x10^{12}
Hemiptera	3.49x10^8	No data [d]	5.24x10^8	-	5.24x10^8	5.24x10^8	1.05x10^9
Homoptera	3.47x10^{10}	No data [d]	1.32x10^{11}	-	-	-	1.32x10^{11}
Adult spider	1.31x10^6	No data [d]	-	-	2.35x10^7	2.35x10^7	2.35x10^7
Coleoptera (larvae)	7.79x10^7	No data [d]	5.60x10^8	-	-	-	5.60x10^8
Isopoda	1.24x10^{10}	No data [d]	6.42x10^9	-	-	-	6.42x10^9

Table 5.2 (continued)

Species [a]	Species abundance $N_{e,j}$ (energy as 'individuals equivalent')	Daily water consumption per species [b, c] $W_{p,i}$ (m^3 population^{-1} [kestrel lifetime]$^{-1}$)	Energy consumed per species [b, d] $E_{c,p,i}$ (kJ population^{-1} [kestrel lifetime]$^{-1}$)				
			From vegetation	From other species			Total
				From vertebrates	From invertebrates	Total	
Diptera (larvae)	$1.22x10^{10}$	No data [d]	$7.97x10^{9}$	-	-	-	$7.97x10^{9}$
Isoptera	$3.89x10^{11}$	No data [d]	$2.82x10^{10}$	-	-	-	$2.82x10^{10}$
Juvenile spider	$7.89x10^{6}$	No data [d]	-	-	$1.17x10^{7}$	$1.17x10^{7}$	$1.17x10^{7}$
Thysanoptera	$2.02x10^{13}$	No data [d]	$1.47x10^{12}$	-	-	-	$1.47x10^{12}$
Total		$3.8x10^{-2}$	$5.3x10^{12}$	$1.0x10^{6}$	$5.6x10^{8}$	5.7x108	$5.3x10^{12}$

[a] species named as listed in the data by Goldwasser and Roughgarden (1993) [87]. [b] 'water consumption' and 'energy requirement' are calculated using the corresponding ecological needs equations, described in Table 3.1, Chapter 3. [c] Water consumption describes only the water ingested by organisms and does not include water consumed as part of their diet (i.e., from prey) or as a by-product of metabolism [d] Energy as number of individuals equivalent accounting for annual variations in metabolic rate due to moulting and breeding/rearing. [e] The water consumption of invertebrates is not yet quantified using the ecological needs equations.

As the total estimated resource requirements of the system over the kestrel's lifetime, the total results in Table 5.2 represent embodied energy and water requirements of the kestrel on St Martin Island [513], [514]. That is, the kestrel's total direct (i.e., the kestrel's requirements) and indirect (i.e., the requirements of the species underpinning the kestrel) water consumption and energy requirements: 3.8×10^{-2} m^3 $system^{-1}$ [kestrel lifetime]$^{-1}$, and 5.3×10^{12} kJ $system^{-1}$ [kestrel lifetime]$^{-1}$, respectively.

5.4 Perspectives

The aim of this chapter was to demonstrate how the concept and methods developed previously in this thesis can be used to augment quantitative food web data, advancing current understanding of ecological systems (as food webs) through the quantification of land use, energy requirements/flows, and water consumption at the system level. An important aspect of this application is to encourage a holistic view of natural resource use within ecological systems. The ecological needs equations proposed in Chapter 4 of this thesis quantify natural resource use for individual species. However, species exist not in solitude but alongside other species as part of interactive ecological systems. Supporting a (e.g., keystone) species therefore requires that all species underpinning it are also adequately supported. In this case, supporting the American kestrel on St Martin Island requires 25 other animals and vegetation are supported concurrently.

However, several data gaps at the species-level are identified, requiring assumptions to be made. Data on metabolic efficiencies (η_i), periodic changes in species' metabolic rates associated with regular activities such as moulting, reproduction, rearing young, and migration ($E_{var,i}$); and the calorific value of species as prey (c_j) are lacking. A consistent lack of invertebrate data is identified throughout this thesis. To further improve our certainty around the composition of ecological systems (species abundances based on metabolic requirements) and natural resource requirements of at both the individual and system level, these data gaps should be the focus of future work. Furthermore, attempts should be made to model (in)stability within the ecological systems over time though it is not yet clear how this should be done. At present, the system described in Fig. 5.1 – its composition and the species' dynamics – is assumed constant year-round. More dynamic results, accounting for migratory species for example, would further increase the value of these results would offer greater insight into how this ecological system may respond to different human intervention or environmental pressures. A better understanding of natural resource use over time could also improve our capacity to manage ecological and socioeconomic stocks concurrently, in a more intergenerationally secure way.

Metabolic efficiencies are also shown to be narrowly bounded across organisms; there is an upper limit to how efficiently organisms can metabolise energy from food [515]. While it is important to account for metabolic efficiency, as demonstrated in this case study, organisms' rates of consumption are a more important driver of energy transfer through trophic systems [364]. Therefore, understanding the dynamics of species interactions, consumption in particular, holds greater significance than metabolic efficiencies. When describing energy flux, population dynamics warrant greater attention than metabolism at the individual scale.

The results of this chapter are also context-specific, limiting their application across different temporospatial conditions. The system being described is specifically that of the American kestrel on St Martin Island in the Caribbean. While the methods demonstrated can be applied in any temporospatial context (climate, location, habitat type, timespan) and to any ecological system, the results cannot. It may be possible to generalise results, to apply them in temporospatial equivalent contexts, or adjust them in some way, to quantify ecological systems where the requisite data lacks, but neither can be determined at this stage. This requires further work so that different results – modelled ecological systems across different temporospatial contexts and of different population dynamics – can be compared and relationships (or lack thereof) identified. In the meantime, it is recommended that the natural resource requirements of ecological systems are quantified on a case-by-case basis.

This study uses parameters derived from statistical analyses (regression models) rather than mechanistic or process-based theory (e.g., Lotka-Volterra modelling [516], metabolic theory [369]), which limits how well dynamic complexities of ecological systems are captured; particularly where empirical understanding of ecological processes is poor. Therefore, some degree of uncertainty is inherent in the model's results and stability. The model may be sensitive to variations in parameter values, requiring careful consideration and robust ecological understanding where possible, when defining a trophic system and how its constituent species may interact. It is recommended that the relative significance and merit of different parameters, empirical and mechanistic, should be compared in future research.

5.4.1 Relating land use change to potential biodiversity impacts

An area's capacity to support the kestrel as part of the described ecological system (Fig. 5.1.) depends on the potential bio-productivity (energy production by primary producers). To determine how land use change (land transformation and land occupation) – the development of land for commercial use for

example – could affect species diversity local to the change, the bio-productivity of different land cover types found on St Martin Island must be quantified. These must then be related to the energy requirements of the ecological system. A bottom-up method of modelling energy flow through the modelled ecological system, from primary producers (vegetation) to apex predator (kestrel), is demonstrated in Chapter 4. Though beyond the scope of this chapter, the same method can be applied to quantify and characterise (for life cycle impact assessment) the potential life cycle impacts of different land use categories either from (i) current land cover, determined through on-site surveying, or from ecological reports [517] or databases (e.g., from the Dutch Caribbean Biodiversity Database (DCBD) [518]), or (ii) theoretical land cover (e.g., potential net primary productivity [519]), on local species diversity. Until the use of different flora by species within the kestrel food web is quantified, it can be assumed that the land cover type with the highest bio-productivity will best support the kestrel and the species underpinning it.

5.5 Conclusion

[1]This chapter demonstrates how the ecological needs concept can be used to produce comprehensive, empirical models of ecological systems. Importantly, it demonstrates that this can be done without the need for comprehensive ecological input data at the species level: while the better quantified the ecological system in the first instance, the less speculative and more accurate the result, useful models can be built requiring only (i) which species are consumed and (ii) in what approximate proportions. This chapter tested this using data for the American kestrel on St Martin Island, Caribbean. Existing data is improved through the quantification of species abundances (relative to the kestrel) and natural resource (land, energy, and water) use at the individual, species, and system level. Inferences made using this improved food web are less speculative than those made using understanding of the ecological system prior this thesis [87].

It is estimated that supporting an American kestrel on St Martin Island requires 0.32 km^2 land area (L_i), $2.3x10^3$ kJ $system^{-1}$ day^{-1} energy (E_s), and $2.1x10^{-5}$ m^3 $system^{-1}$ day^{-1} water (W_s). This represents the natural resource requirements of the American kestrel and the 25 terrestrial species (and vegetation) identified as underpinning it; natural resources used by socioeconomic systems concurrently. The results clarify the wider ecosystem requirements to support a target species: supporting an American kestrel over the period of a day requires the support of 9,533 individuals as energy equivalent ($N_{e,j}$ – see '6.2 Methodology'). Supporting an American kestrel over its assumed lifetime (five years) requires the support of $2.4x10^{13}$ individual animals (energy equivalent). Regarding scaling, it is found that while

natural resource use of a species scales with population size, $N_{p,i}$ (individuals), natural resource use does not scale linearly at the system level, N_s, (individuals). The results at the system-level must be re-calculated when the timescale is changed.

The method demonstrated in this chapter can be used for almost any ecosystem and any animals, allowing species abundances and their natural resource requirements can be quantified at the species and system level. Prior to this thesis, interspecies interactions and energy flows within ecological systems had often been limited to illustrated qualitatively, as a food web, and described to some extent (predation frequencies, diet compositions) qualitatively [87]. While this model can contribute to the understanding of ecological systems, its uses will be increased through improved ecological and animal specific data, for example the natural resource (water, energy, land etc) of individual species. Additionally, existing results are context-specific, limiting their generalisability in other temporospatial conditions. Finally, the ecological needs concept is currently restricted to physiological requirements: land use, energy requirement, water consumption. Species' needs include other requirements (e.g., social needs, nesting requirements, and nutrition [520]) not yet included in this thesis. It is prudent that natural resource requirements, such as the results of this chapter, inform conservation and design efforts alongside quantitative and qualitative understanding of species' other needs and preferences.

6 Conclusions

The need to quantify the natural resource use of ecological systems under different population dynamics and temporospatial contexts represented a pertinent deficiency in natural resource management, life cycle assessment (LCA), and wildlife-inclusive urban design. The last two fields are strongly interlinked, with LCA results able to inform urban design – especially when designing urban areas to better support urban biodiversity. To address this knowledge gap, this thesis aimed to develop a concept and accompanying method for quantifying the wellbeing needs of ecological systems through which the life cycle impact of biodiversity-supporting urban infrastructure on biodiversity can be quantified more comprehensively and then integrated into the LCA method. To this end, this thesis has provided the first steps of what I hope will turn into a considerable body of work.

This thesis presented, for the first time, a quantified set of ecological needs for terrestrial animals, analogous to human wellbeing needs, and an accompanying method for applying them to model the potential impacts of land/surface cover change on local (terrestrial) species diversity. This greater quantitative understanding of natural resource use in ecological systems can facilitate more informed and more sustainable management of natural resources across socioeconomic and ecological systems, and more comprehensive life cycle assessment of local biodiversity impacts associated with land use change. The thesis' aims were met.

The application of the ecological needs equations, to quantify the natural land, water, and energy use within terrestrial ecological systems at the community level, is demonstrated through two case studies: an investigation of the life cycle local biodiversity impacts of a residential development in London, UK (Chapter 4); and the development of a 'quantitative food web' using species data from St Martin Island (Chapter 5). With this method, it is possible to quantify and hence characterise (through the development of characterisation factors) the potential local biodiversity impacts associated with changing on-site bio-productivity. In Chapter 4, on-site bio-productivity is affected by land use change (urban brownfield to urban sedum roof). The results enable changes to land cover to be causally linked to potential changes in local species richness (i.e., local biodiversity impact). This enables the development of new endpoint characterisation factors so that life cycle local biodiversity impacts of land cover change can be more comprehensively assessed through life cycle assessment (LCA).

Critically, the method is not limited to the urban context nor is it limited to land use change; it may be applied to the conservation or (re)introduction or individual species or entire ecological systems, and to any biodiversity-supporting component; from green or blue infrastructures (e.g., green roofs and wildlife ponds, respectively), or nesting sites to name a few examples. As such, this thesis presents key advancements needed to assess and hence design urban landscapes that can support urban ecological systems alongside humans for mutual and intergenerational benefit.

The literature review presented in Chapter 2 demonstrated that the quantification of the physiological requirements of different species is currently poor, and hence different ecological systems, under different spatial and temporal conditions. This knowledge gap represents a considerable obstacle to modelling natural resource use and assessing the potential impacts of human intervention (by way of resource provision) on biodiversity. Consequently, the capacity to quantitatively manage natural resources, for concurrent human and ecological use, was limited prior to this thesis. This has limited the application of material flow analysis, life cycle assessment, policymaking, footprint model (e.g., ecological footprint) development, and urban and environmental conservation action (to name a few examples). Given that urbanisation is both an important driver of biodiversity loss and a source of increasing conservation value, biodiversity-supporting urban design is chosen as the focus of this thesis.

The allometric equations in Chapter 3 represent the first quantitative set of ecological needs for terrestrial mammals, birds, reptiles, and invertebrates, analogous to human wellbeing needs. New equations for land use were developed for terrestrial mammals, birds, and reptiles that incorporated population density data not previously included in previous versions (e.g., PanTHERIA [211], TetraDensity [212]). An allometric equation for land use of invertebrates was also described. Invertebrates had previously been overlooked despite their importance in ecological systems. Together, these equations represent a means of estimating concurrent land, water, and energy requirements of terrestrial mammals, birds, reptiles, and invertebrates. This method conceptualises, for the first time, a comprehensive ‘dual-system’ approach: modelling natural resource use in socioeconomic and ecological systems simultaneously, thus facilitating more informed, more sustainable natural resource management for the conservation and creation (e.g., rewilding) of biodiversity within ecological systems; in both urban and non-urban contexts. The concept and method also represent an advancement in determining what socioeconomic compromises are needed to achieve contemporary conservation targets alongside ongoing human development.

An application of the ecological needs equations was presented in Chapter 4, centred around a case study of planned urban development in London, UK: the development of residential buildings with either concrete-asphalt roofs or sedum roofs. The aim of this study was to quantify and subsequently characterise the potential life cycle local biodiversity impacts of the sedum roofs, which had not been quantified nor characterised within LCIA. Ecological needs were used to explicitly model ecological systems at the community level, using estimates of land use and energy requirements to quantify proportionate species abundances across different trophic levels. Next, a bottom-up method was demonstrated to relate energy provision (from the sedum roofs atop residential buildings) to the metabolic requirements of each species in the ecological system: first, those related to herbivory (direct consumption of nectar or plant bio-mass), then to acts of predation. Thus, the flow of energy through the trophic system was empirically modelled, and the potential local biodiversity impact (changes in local species richness and local species abundance) of installing sedum roofs on-site was quantified.

This research demonstrated how local biodiversity impacts can be explicitly modelled using ecological needs and then integrated into LCIA. Furthermore, by developing characterisation factors based on those results, local biodiversity impacts associated with different types of land use change can be included within LCIA without the need for modelling in each case. The limited capacity of sedum roofs to offset life cycle biodiversity impacts associated with urbanisation was also illustrated. The results indicated that sedum roofs have limited potential for offsetting the life cycle biodiversity impacts associated with urban developments, with the increase in local species diversity supported by the sedum roofs equal to only 1.3% of the development's life cycle biodiversity impacts. Greater local biodiversity gains can be achieved through more bio-productive designs (e.g., intensive green roofs). However, the upper limit of such impacts is ultimately determined by species richness in the local species population.

Chapter 5 built upon the method demonstrated in Chapter 4 to quantify the natural resource requirements of the ecological system (food web) associated with the American kestrel on St Martin Island, Caribbean. This work used more comprehensive data on interspecies interaction (acts of predation) [87] than was available for the case study in Chapter 4, and incorporated variables that, when omitted in Chapter 4, constituted limitations: metabolic variability, $E_{var,i}$, and a les restricted timeframe. The modelled ecological system and its natural resource requirements are inherently less speculative as a result. From this result, the potential impact of different types of land use change are inferred, demonstrating their conservation value. This understanding is of value in urban design, too, as it offers quantitative data on the provision required to support the American kestrel, and the species underpinning it. Biodiversity-supporting infrastructure can therefore be designed accordingly. Of course, physiological requirements represent only some of the species' prerequisites. Understanding of

species' other requirements (e.g., social needs or nesting requirements) should inform design concurrently.

The work in this thesis represents a means to achieve methodological change, foremost in the application of LCA, but also a fundamental change in perspective to consider ecological systems alongside and analogous to socioeconomic systems. The call is not one for ecocentrism in industrial ecology. Ecological systems play a critical role in supporting socioeconomic systems and human development. To this end, its conservation is inherently anthropocentric. What is evidenced is the need for ongoing transference of ecological knowledge into engineering and meaningful use of that knowledge thereafter. The subsequent chapter, 'Recommendations for Further Work' outlines recommendations on how that may be achieved.

7 Recommendations for future work

7.1 Regarding data limitations

The greatest obstacle to quantifying the ecological wellbeing needs of different organisms is a lack of quantitative data at the species and system level, and a currently poor quantification of ecological system dynamics. Invertebrate data is particularly lacking [26]. Ecosystems are complicated and dynamic systems and there is much we still do not understand about them. The most important data gaps are water consumption of terrestrial animals, and the physiological needs of invertebrates across the board. Collecting information such as native/non-native status is also prudent because theoretical reference states used in LCIA methods would comprise native species (e.g., [19]). While any change in species richness constitutes a biodiversity impact, the support of non-native species would not count towards the restoration of reference-state conditions.

Collecting these data requires ongoing multidisciplinary collaboration. Despite technological advancement, there remains a dependency on field work which is time and labour intensive. Development of generalised equations to predict ecological needs quantitatively will be slow as a result. Better data and greater understanding will come in time, contingent on technological and methodological innovation and ongoing, multidisciplinary collaboration. Here, genetic (eDNA) monitoring techniques [200] is being used to better understand ecological system dynamics in both aquatic and terrestrial habitats (e.g., [521], [522]). Used alongside ecological wellbeing needs, this data offers a means to explicitly model natural resource use between species at the community level (see Chapters 4 and 5). In the meantime, data from captive animals is better than no data when trying to improve our understanding the natural resource use of their non-captive counterparts and the ecological systems they make up. This data can be collected from species within the socioeconomic domain (domestic livestock and animals in zoos) to supplement data from their wild counterparts, with some adjustment.

Captive species are the only means of quantifying resource use for some species. For example, where locations are inaccessible (e.g., due to terrain, political climate, or conservation status), or where species are extinct in the wild. Zoos can provide morphological and physiological data, either observed or inferred. A high level of welfare must be ensured through adequate provision of resources (food and water) and stimulation, which can be quantified. However, captivity is known to affect both morphology [9], [10] and physiology [525]. This reflects the fact captive species are not exposed to the same hardships or resource insecurities as their wild counterparts. Their needs are met through consistent human intervention. Less energy is expended through hunting, foraging, or migrating. By comparing

data from wild and captive specimens of the same species, under equivalent temporal-spatial conditions, a method of scaling between the two conditions may be developed. Studies into the effect of captivity on morphologies focus on soft tissue and bone [523] provide an example of how this could be done for resource use.

The fact species are confined to enclosures makes captive environments poor sources of land use data. Data from wild comparable species under similar spatial-temporal conditions should be used here instead. This will require (dis)similarities between resource use in comparable species to be scrutinised. However, there is uncertainty surrounding the role of individual preference in land use, and the extent to which non-physiological (social) needs and other factors contribute to land use. The accuracy of inter-species generalisations, regarding land use, is therefore difficult to estimate.

Museum specimens offer an additional means to enhance species data. Museum collections have already been used for genetic and morphological data [526], [527], [528]. Here, we suggest they fill the considerable data gap identified for invertebrates. Museum species are of course inanimate. Their value lies in inferring 'living' body mass from their preserved weight. Invertebrates are particularly suited to this given that body water content can be easily measured in small organisms (e.g., [529]). Invertebrates are typically preserved and pinned. They are not preserved in liquids, like aquatic specimens, nor are they mounted (stuffed) like mammals. This means the specimens can be easily weighed and the weight of its trappings (e.g., pins, ID labels) deducted. A living weight can then be deduced be accounting for preservation effects, as described in [530], [531] for invertebrates, and [532] for amphibians. However, limited research comparing living to preserved (specimen) body mass suggests that further research is needed to make use of most museum specimens in this way.

Another important source of species data is 'citizen science': scientific research contributed by the public [533]. Species data is available in citizen science repositories such as iNaturalist [534], which is used to identify species local to the case study site in Chapter 4. These data typically comprise records of species observations alongside geospatial information (e.g., location, date), from which other temporospatial details can be inferred (e.g., land cover, ecosystem quality). Citizen science is useful for collecting large volumes of data in a short time, over a wide area [535]. Individuals can contribute ecological data with limited training, often as well as professional surveyors [536], [537], [538]. However, the need for training and specialist equipment can hamper data collection. The development of mobile apps, 'PlantNet' (for plant ID) [539] and 'Merlin' (for bird ID) [540] for example, are making citizen science more accessible, removing the need for specialist equipment. Even so, there are concerns

about the quality of citizen science data including unequal geographical coverage, and individual bias where common species (seen as not noteworthy) or non-charismatic species are not recorded [536], [541].

Finally, there is opportunity to collect data pertaining urban biodiversity as part of the process of development. The development of urban areas causes land use change, land transformation and land occupation, the biodiversity impacts of which are lacking. It is prudent to encourage, or mandate through policy, developers to collect ecological data preceding land use change. Furthermore, ecological surveys should then be performed periodically to understand (i) local (in-situ) biodiversity change arising from natural restoration (relaxation), and (ii) as post-hoc assessment of the local biodiversity impacts of biodiversity-supporting components on-site; this includes green and blue infrastructure as well as more commonplace contributions: plants growing in pavement cracks and walls, for example. This would improve currently poor quantitative understandings of urban biodiversity and species-infrastructure interactions in different contexts. It would also establish new best practice in urban development with multidisciplinary advantage.

7.1.1 Legislating and incentivising data collection for validating empirical model outputs

Governments and the private sector can play a crucial role in implementing policies that promote investment in biodiversity protection and conservation efforts [542]. This includes the advancement of biodiversity monitoring efforts needed to enhance and validate empirical models, including the ecological wellbeing needs demonstrated in this thesis, as well as their results. This requires on-field measurements of species diversity and abundance (including plants), as well as ecosystem dynamics and functionality, across temporospatial contexts. On-site data collection before and after interventions that affect land cover are important, especially over protracted timeframes. This provides insight into the potential effects of different intervention across key variables.

In the UK, policy and incentives for nature-based solutions and biodiversity gain respond to long-term Environment Act (2021) targets [543], [544] and include grants that encourage, '*habitat creation and restoration at landscape scale'* [545]. Climate mitigation and water management interventions are typically emphasised [542], [546]. Biodiversity gain is encouraged through Biodiversity Net Gain regulation in England [544], [547], [548], [549]. The UK has a relatively high level of environmental legislation pertaining to interventions like nature-based solutions compared to other European countries, but its provision of financial incentives is comparatively very low [542]. There is considerable scope to better encourage biodiversity monitoring and guide best practice to ensure high-quality data collection.

This thesis has identified that encouraging (mandating or incentivising) biodiversity-monitoring by developers is a priority: data should be gathered before and after a site is developed, and throughout its use. Legislation should clarify where the onus then lies for continued monitoring during site its passive relaxation (see Fig. 2.4).

7.2 Characterising local biodiversity impacts across different temporospatial contexts

In this thesis, the potential biodiversity impacts of sedum roofs in London, UK, were explicitly modelled and characterised. However, the methodology proposed is not restricted to the urban context. It is applicable to urban and non-urban contexts, across any temporospatial condition. Future work should focus on characterising the local biodiversity impacts of land use across different contexts such that, conservatively, most biodiversity-supporting infrastructures and components may be included in LCA. This includes 'bug hotels', landfill sites, any surface in fact – even bare walls and pavements can support urban biodiversity. The more comprehensively biodiversity-supporting components can be assessed through LCA, the more effective human intervention can be at supporting or (re)creating biodiversity. This includes the implementation of nature-based solutions. Given their prevalence in the urban landscape, the potential impacts of green infrastructure on local biodiversity should be characterised first. The method demonstrated in Chapter 4 can be applied to any type of land/surface cover comprising vegetation, which expedites the task. Other components will require some creative thinking.

Based on the case study in Chapter 4, it is also recommended that species data is aggregated over a one-year period when the local ecological system is being described. Seasonal variations in physiology (e.g., dietary changes) and population dynamics (e.g., due to migration) must be captured – the former will not be captured in species abundance data but must be incorporated manually.

An important limitation in relating changes in land use type to potential impacts to local species diversity is a lack of understanding on how energy is distributed between species within dynamic trophic systems. In Chapter 4, it was assumed that energy flows from the primary producers (vegetation) to the invertebrate population. Once the invertebrate population had sustained itself, per the dynamics of the modelled ecological system in that case study, the remaining invertebrates were evenly distributed between the species they support (see Table A35, Appendix 5). Whether the energy requirements of intermediate species were adequately supported determined whether their predators could then be sustained through the sedum roofs. However, an equal distribution of energy (as food) is unlikely in ecological systems where different species compete for the same resources. Further work should describe how different species compete for the same resources within different ecological systems. The distribution of energy across different trophic levels can then be modelled in a more representative way.

Exactly how this should be done is not yet clear, but given ecological systems demonstrate temporospatial heterogeneity, the work will likely need to be done on a case-by-case basis.

Finally, this thesis focuses on the supply and flow of energy through terrestrial ecosystems, with the relationship between land cover change and biodiversity change simplified. While simplifying ecosystem dynamics for inclusion in LCA, it complicates accurately predicting how different interventions (e.g., green roofs, gardens) might interact with one another across urban landscapes and how that interactivity will affect biodiversity at the urban scale (i.e., city-wide) or at larger scales (e.g., across regions). The combined potential biodiversity impact of biodiversity-supporting infrastructures is governed by the interactivity between the species and systems they support. Similarly, the ecosystem functionality of cities will depend on the total ecosystem dynamics achieved system-wide. Unfortunately, these aspects are not included in the research at this stage. Currently, the biodiversity impact of interventions scales with territory size until it alters the local species pool. The interplay of urban biodiversity underpinned by human intervention therefore represent considerable opportunity for future research.

7.3 A new environmental pressure category for including biodiversity-supporting infrastructures and biodiversity-supporting components in LCIA

The need to inventory biodiversity-supporting infrastructures and components comprehensively within LCIA, to facilitate the assessment of all potential local biodiversity impacts, was identified in Chapter 4 (‘Section 4.4.1.’). Inventory models should include all biodiversity-supporting surfaces; not only different types of land cover (i.e., land use) but every surface type (material of construction, texture, condition, and orientation) influencing the cumulative potential impact of a building on local species diversity and abundance. This will facilitate more accurate assessments of the potential local life cycle ecological impacts (and hence the overall potential ecological impacts) of socioeconomic activities are possible. Regarding urban design, urban landscapes can then be better engineered to support ecological needs concurrent in addition to human needs, so that ecological and socioeconomic systems can be integrated in a way that mutually beneficial (per the aims of ecological engineering).

While best practice in characterising and inventorying biodiversity-supporting infrastructures and components for LCIA will be design- and context-dependent. Recommendations for including common biodiversity-supporting components within the life cycle inventory are listed in Table 7.1, based on the work undertaking in this thesis. Table 7.1 is not an exhaustive list.

Table 7.1 Recommendations for inventorying common biodiversity-supporting components across two main categories: (1) 'surfaces that provide resources' including (i) horizontal green infrastructure, (ii) non-horizontal green infrastructures, and (iii) surfaces not comprising vegetation; and (2) 'components that provide shelter' including (i) nesting sites

Component	What determines the component's biodiversity-supporting capacity?	Inventoried as
Resources-providing surfaces		
Horizontal green infrastructures (e.g., green roofs, gardens)	The components' total (primary) bio-productivity, i.e., nectar or bio-mass production	Surface area covered [a]
Non-horizontal green infrastructures (e.g., green walls, climbing vines)	The components' total (primary) bio-productivity, i.e., nectar or bio-mass production	Surface area covered by the bioproductive surface [a]
Surfaces not comprising vegetation (e.g., feeding tables, paved surfaces with plants growing in cracks)	Design- and context-dependent. Example 1: Feeding tables Surface area, food type, refill frequency, location, and physiological (e.g., space required), social (e.g., proximity to other species) and temporospatial (e.g., climate, disturbances) factors. Example 2: Paved surfaces Primary productivity inferred from observations of primary succession in comparable locations, then characterised as a land cover type.	Case dependent but relating to surface area of each resources-providing [b] surface
Shelter-providing components		
Nesting sites (e.g., bird boxes, bat roosts, and 'bug hotels')	The number of individuals potentially housed. This is contingent on physiological (e.g., space required), social (e.g., proximity to other species), and temporospatial (e.g., climate, disturbances) factors.	Number of units (e.g., 1 swift nest box)

[a] elevation affects the accessibility of resources to different organisms, but is not included as part of the inventory step; [b] resources provided through bio-productivity

7.4 The co-development of the 'ecological needs' concept and metabolic theory of ecology

The ecological needs concept, proposed in this thesis, and the metabolic theory of ecology (MTE) are contrasted in 'Section 2.4.7', Chapter 2. What is identified is the potential to co-develop both concepts to include greater physiological consideration into MTE while advancing the ecological needs concept from static to dynamic, to capture ecosystem functionality, population stability, and multifactorial effects. Given the intersection between the two concepts, there is rich opportunity for interdisciplinary collaboration to advance the work proposed in this thesis. With MTE laying the groundworks, and this thesis demonstrating the importance of more comprehensive physiological consideration, the two concepts can converge to produce dynamic, empirical models of ecological metabolism that describe

resource use associated with all physiological needs of organisms under different, multifactorial (temporal, spatial, environmental) conditions. This would rank amongst the most ecological comprehensive tool for modelling potential life cycle impacts on species diversity, ecosystem quality, and ecosystem function in a way compatible with LCA and LCIA method. This future research, a culmination of both concepts, is highly encouraged.

8 References

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9 Publications from this thesis

Peer-reviewed publications

1. Mason, A. R.; Gathorne-Hardy, A.; White, C.; Plancherel, Y.; Woods, J.; Myers, R. J. (2022). Resource requirements for ecosystem conservation: A combined industrial and natural ecology approach to quantifying natural resource use in nature. *Ecology and Evolution,* 12 (e9132)

2. Mason, A. R.; Gathorne-Hardy, A.; Myers, R. J.; Smith, B. M.; Puchol-Salort, P. (2024) Local Terrestrial Biodiversity Impacts in Life Cycle Assessment: A Case Study of Sedum Roofs in London, UK. *Journal of Industrial Ecology*.

In preparation

1. Mason, A. R.; Gathorne-Hardy, A.; Smith, B. M.; Myers, R. J. (in preparation) Ecological needs in ecological engineering.

2. Mason, A. R.; Gathorne-Hardy, A.; Smith, B. M.; Myers, R. J. (in preparation) Quantifying the land use, energy requirements, and rates of water consumption of wild species in terrestrial ecosystems: A case study of the American kestrel (*Falco sparverius*) on St Martin Island, Caribbean

Conferences

1. Mason, A. R.; Gathorne-Hardy, A.; White, C.; Plancherel, Y.; Woods, J.; Myers, R. J., 'Quantifying ecosystem wellbeing needs for conservation and resource management', British Ecological Society: Ecology Across Borders 2021, Liverpool, UK, 12-15 December 2021.

10 Appendices

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10.1 Appendix 1: Supplementary information for Chapter 3, 'Quantifying the natural resource requirements (or 'ecological needs') of terrestrial animals analogous to human wellbeing needs'

10.1.1 Regression coefficients for the proposed ecological needs equations

Below are the regression coefficients derived from our regression analyses for $\log_{10}$(population density) versus $\log_{10}$(adult body mass)). These coefficients are all calculated using the 'fitlm' linear regression model [1] in the MATLAB computing environment.

Table A1 Regression coefficients for $\log_{10}$(population density) versus $\log_{10}$(adult body mass) for terrestrial mammals, birds, reptiles, and invertebrates. Where *n* describes the number of observations, and *e* the root mean squared error.

Class	Trophic level	*n*	*e*	R^2	p-value
Mammals	Carnivore	1,323	0.76	0.666	$1.46\text{x}10^{-316}$
Mammals	Herbivore	3,373	0.869	0.667	0
Mammals	Omnivore	1,461	0.796	0.611	$8.64\text{x}10^{-301}$
Birds	Carnivore	6,659	0.842	0.300	0
Birds	Herbivore	1,120	0.854	0.112	$1.28\text{x}10^{-30}$
Birds	Omnivore	2,521	0.808	0.113	$2.43\text{x}10^{-67}$
Reptiles	All	578	1.07	0.205	$1.44\text{x}10^{-30}$
Invertebrates	All	31	0.992	0.377	$2.39\text{x}10^{-4}$

In each case, the p-value is low, indicating that the null hypothesis can be rejected. This confirms what is already known: there is a relationship $\log_{10}$(population density) versus $\log_{10}$(adult body mass) across the four biological classes we analysed. There is great variation in the R^2 values in Table A1. The equations for mammals capture a greater amount of variance than the equations for birds, reptiles, and invertebrates. The remaining variation could be attributed to variations in the habitats in which the species – those described across the data – were observed. Variations in natural resource availability and the rate of its production in one location may vary considerably to those in locations with different climates, habitat qualities, and population dynamics for example. Moreover, this data describes living, autonomous beings – some variation must be attributed to individual behaviour and personal choice, both of which are difficult to describe in a general (i.e., species-level) sense. To improve the analyses in future, it would be useful to investigate the relationship $\log_{10}$(population density) versus $\log_{10}$(adult body mass) disaggregated by habitat-type. While this is not easily done using data from Damuth (1986)

and similar, more recently published databases (e.g., TetraDensity) permit such disaggregation for some datapoints. However, even in the TetraDensity database, many datapoints lack a 'habitat type' value.

In Table A2, we present the biological class-specific coefficients (a_i and b_i) which are described in Eqs. (3.9-3.15). A shown in Eq. 3.8 (main text), a_i corresponds to the intercept of the linear line of best fit in each plot in Figure 3.2, Chapter 3 (red lines); b_i corresponds to the gradient of the line in each case; and $e_{a,i}$ *and* $e_{b,i}$ are the standard error terms for a_i and b_i respectively. These coefficients are all calculated using 'fitlm' linear regression model [387] in the MATLAB computing environment.

Table A2 Biological class-specific coefficients for land area use

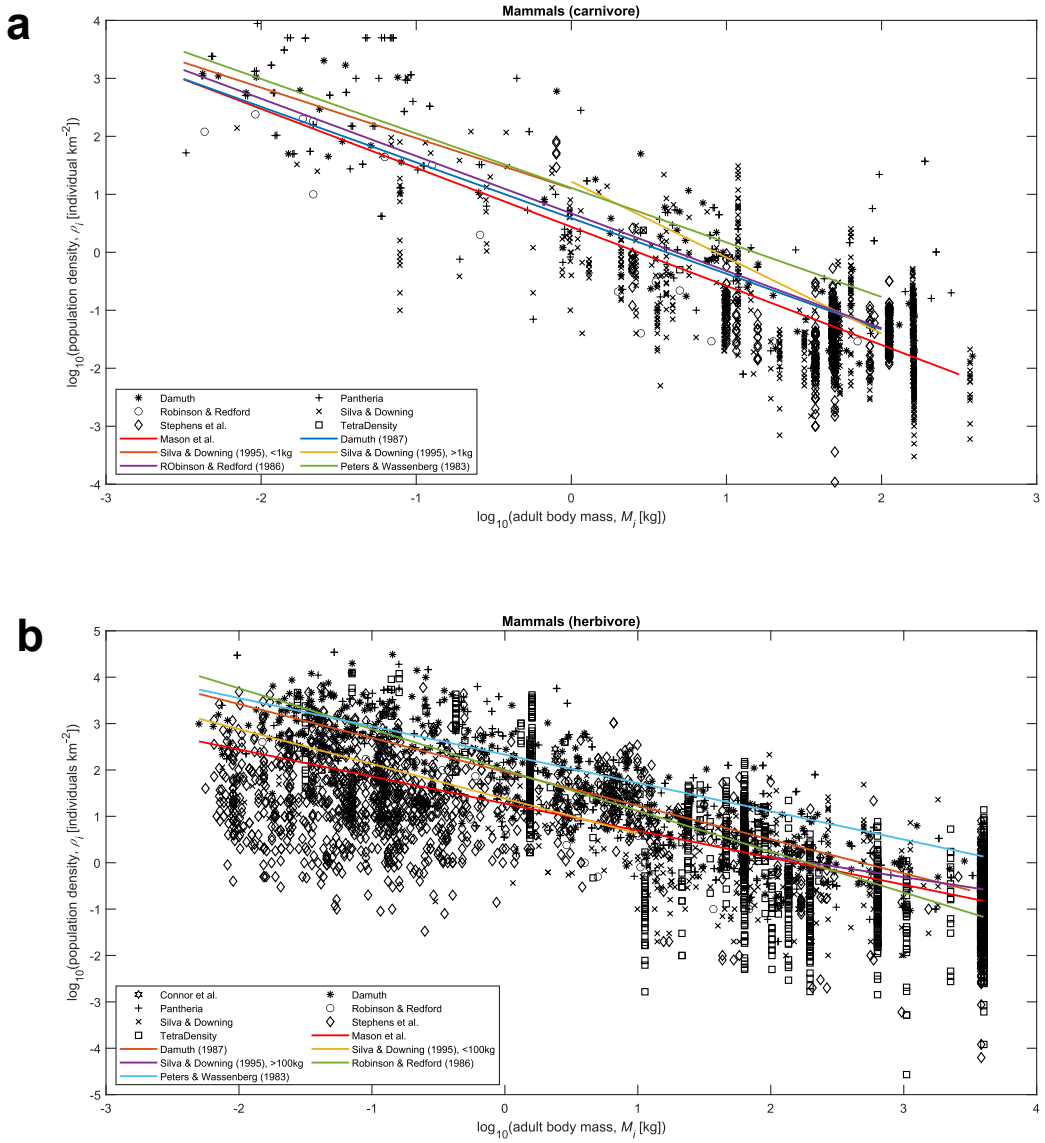
Class	Trophic level	a_i	$e_{a,i}$	b_i	$e_{b,i}$
Mammals	Carnivore	2.74	1.08	-1.018	0.020
Mammals	Herbivore	55.91	1.05	-0.773	0.009
Mammals	Omnivore	20.28	1.05	-0.915	0.019
Birds	Carnivore	0.65	1.06	-0.868	0.016
Birds	Herbivore	6.35	1.11	-0.438	0.037
Birds	Omnivore	5.61	1.07	-0.416	0.023
Reptiles	All	218.3	1.30	-0.656	0.054
Invertebrates	All	7852	5.90	-0.713	0.170

We acknowledge that sampling bias might exist in some of the data used to perform our regression analyses (notably in Figures 3.2a and 3.2d-h, Chapter 3) but addressing this sampling bias is beyond our scope at this stage. The sampling bias observed is due to an asymmetric distribution of data within the sources used to compile the data in Tables A7-A14 (see Appendix 3). Where there is a bias towards individuals of lower $\log_{10}$(adult body mass), e.g., Figs 3.2d-g, the equation for land area use (Table 3.1, Chapter 3) may be an underestimate. Conversely, where there is a bias towards individuals of higher $\log_{10}$(adult body mass), e.g., Figs 3.2a and 3.2h, Chapter 3, the equation for land area use may be an overestimate. In each case, we consider the line of best fit to be sufficiently robust since the relationships between $\log_{10}$(population density) and $\log_{10}$(adult body mass) are generally comparable (see 'Section 11.1.3') to those established in earlier publications: e.g., those published by Damuth (1987), Silva & Downing (1995), Stephen et al., (2019), and Robinson & Redford, (1986).

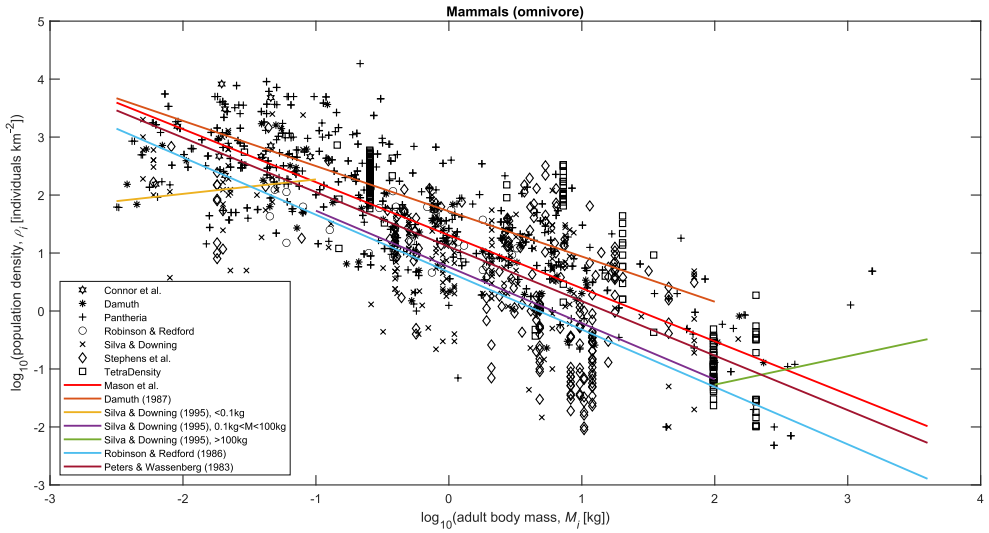
10.1.2 Comparison of proposed land use equations to precedents

In most cases, our proposed equations for the land area use of mammals possess slopes that lie within the ranges of those developed by Damuth (1987) and Silva & Downing (1995), whose equations possess slopes ranging from –0.73 to –0.96, and –0.25 to –1.31, respectively [165], [550]. For mammals, the

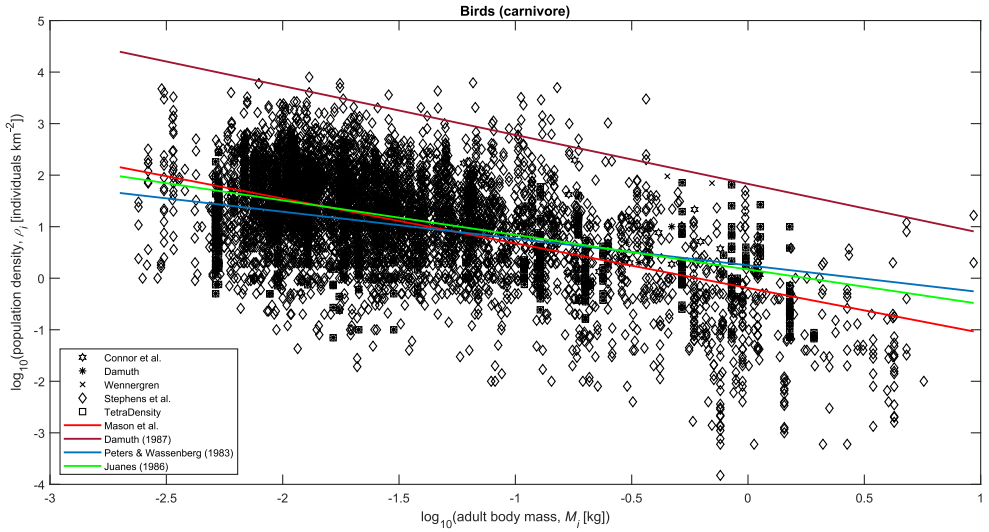
plotted lines have comparable slopes; the exception being the equations from Silva & Downing (1995) for omnivores with adult body mass >100 kg and adult body mass < 0.1 kg (Fig. A1b). We attribute this to the fact that the authors disaggregate the mammalian population density data by body mass as well as trophic level. By reducing the dataset size, the authors were able to identify different relationships between adult body mass and population density [551]. However, it is difficult to determine whether the reduced datasets used were sufficiently large enough to produce accurate allometric equations for population density that may be applied with confidence at the biological class-level.



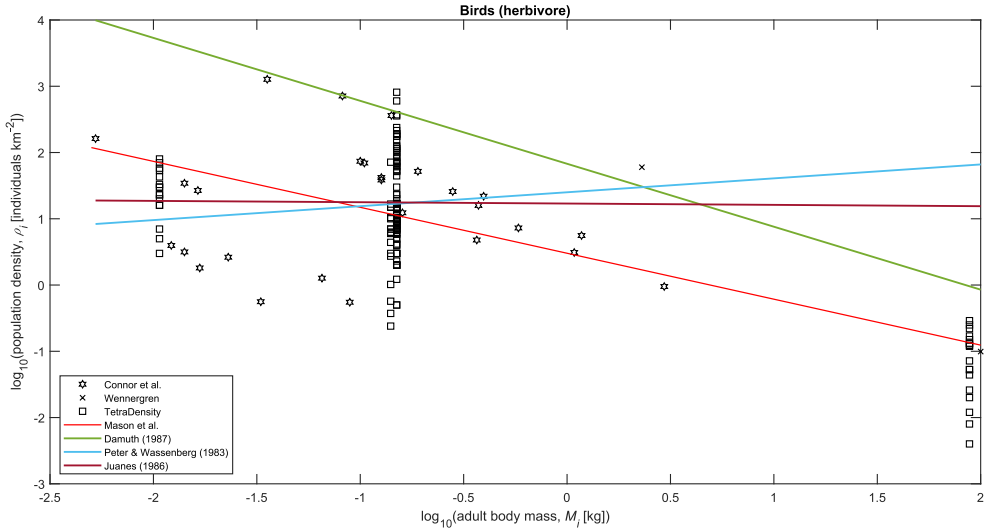
c


Mammals (omnivore)
log10(population density, ρi [individuals km-2])
log10(adult body mass, Mi [kg])
Connor et al.
Damuth
Pantheria
Robinson & Redford
Silva & Downing
Stephens et al.
TetraDensity
Mason et al.
Damuth (1987)
Silva & Downing (1995), <0.1kg
Silva & Downing (1995), 0.1kg<M<100kg
Silva & Downing (1995), >100kg
Robinson & Redford (1986)
Peters & Wassenberg (1983)

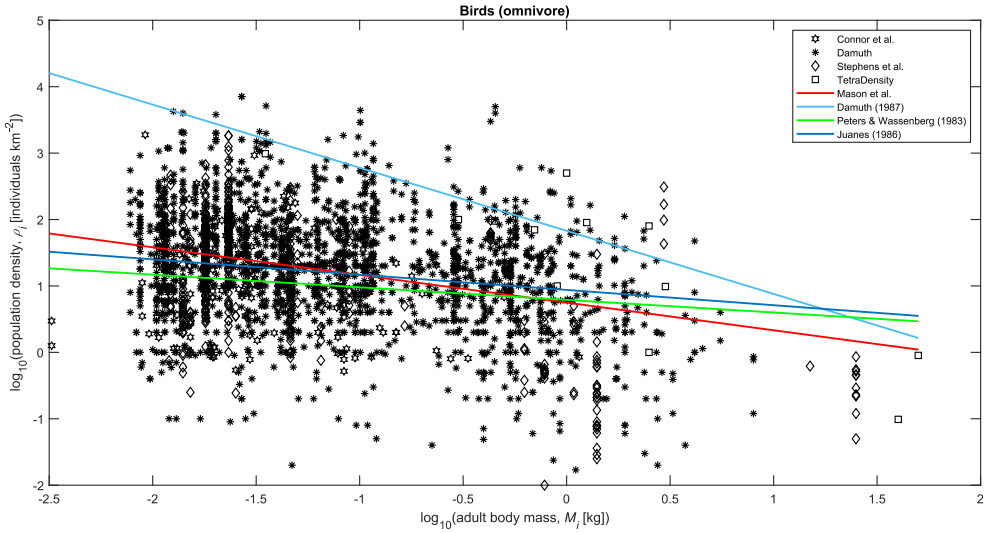
d


Birds (carnivore)
log10(population density, ρi [individuals km-2])
log10(adult body mass, Mi [kg])
Connor et al.
Damuth
Wennergren
Stephens et al.
TetraDensity
Mason et al.
Damuth (1987)
Peters & Wassenberg (1983)
Juanes (1986)

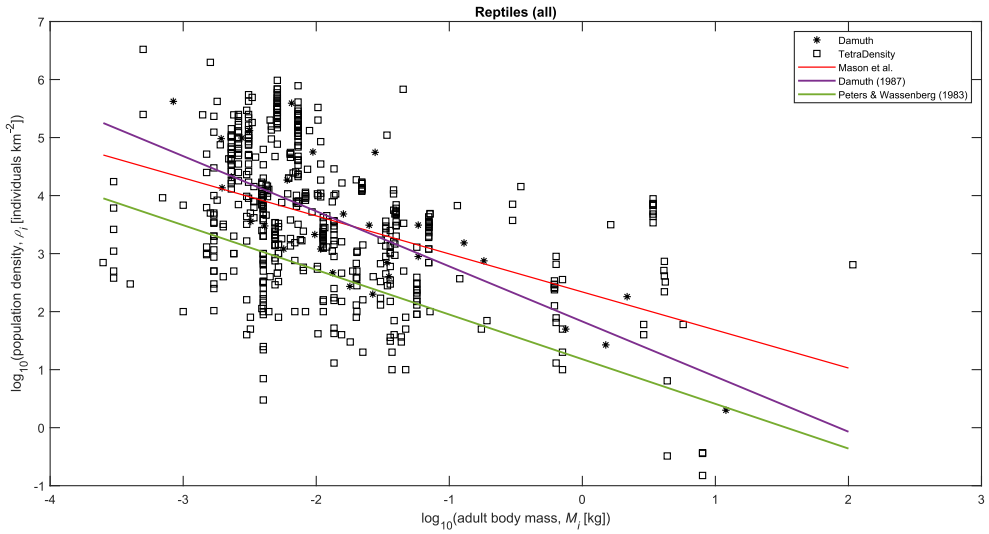
e


Birds (herbivore)
log10(population density, ρi [individuals km-2])
log10(adult body mass, Mi [kg])
Connor et al.
Wennergren
TetraDensity
Mason et al.
Damuth (1987)
Peter & Wassenberg (1983)
Juanes (1986)

f


Birds (omnivore)
Connor et al.
Damuth
Stephens et al.
TetraDensity
Mason et al.
Damuth (1987)
Peters & Wassenberg (1983)
Juanes (1986)
log10(population density, ρi [individuals km-2])
log10(adult body mass, Mi [kg])

g


Reptiles (all)
Damuth
TetraDensity
Mason et al.
Damuth (1987)
Peters & Wassenberg (1983)
log10(population density, ρi [individuals km-2])
log10(adult body mass, Mi [kg])

h

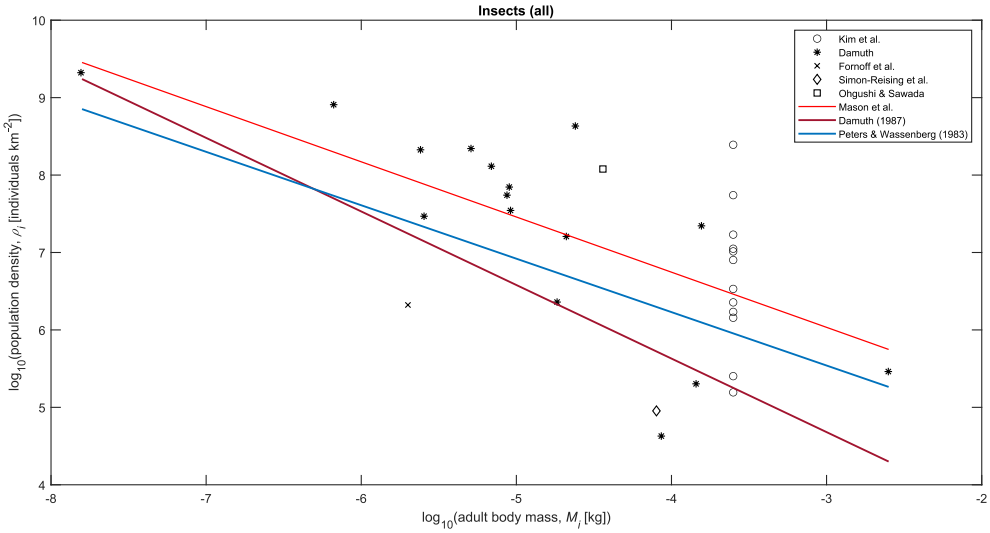

Insects (all)
Kim et al.
Damuth
Fornoff et al.
Simon-Reising et al.
Ohgushi & Sawada
Mason et al.
Damuth (1987)
Peters & Wassenberg (1983)
log10(population density, ρi [individuals km-2])
log10(adult body mass, Mi [kg])

Figure A1 The fit of our proposed allometric equations for land area use (red line, foreground; see Table 4.1, Chapter 4) alongside that of other prominent precedents in the literature, for mammals (**a**, carnivores; **b**, omnivores; **c**, herbivores); birds (**d**, carnivores; **e**, omnivores; **f** herbivores); **g**, reptiles; and **h**, invertebrates. The sources of the data (markers, background) used are shown in the legends. In each case, the full data are available in Appendix 5

In Figure A1c, the allometric equations reported by Damuth (1987) produce results that are an order of magnitude higher relative to the other results. In Fig. A1d, a greater discrepancy is observed; the allometric equations reported by Damuth (1987) produce results that are 2 orders of magnitude higher relative to the other results. It is likely that this difference is the consequence of using a dataset that is much smaller than those used by the other authors referenced in Fig. A1, and substantially smaller than that which is used to produce the allometric equations for land area use presented in Table 3.1, Chapter 3.

There are significant differences between reported allometric equations for birds. Notably, the equations reported by Juanes (1976) and Peters & Wassenberg (1983) possess positive slopes and do not fit the herbivore data [5], [6]. This is contrary to the general trend across all the biological classes, that population density decreases with increasing body size. Additionally, the allometric equation reported by Damuth (1987) apparently over-estimates population density by 1-2 orders of magnitude relative to the others. This may be because Damuth (1987) uses a single equation to describe all non-mammalian terrestrial vertebrates (i.e., birds and reptiles in this scope) and uses a relatively small dataset for non-mammal species, both of which reduce equation accuracy. The limitation of Damuth's small datasets is made more apparent as our expanded dataset contains a considerable quantity of mammal and bird data that was not used in previous studies. In contrast, our expanded dataset contains relatively less additional data for reptiles and invertebrates compared to Damuth (1987). Hence the allometric equations proposed by Damuth (1987) for reptiles and invertebrates are comparable to those we have developed and present in Table 3.1, Chapter 3.

We note that the correlation between body mass and population density across invertebrates species is relatively poor [554], [555], owing to a lack of reported population density data for many invertebrates species. Until such data is collected, we are unable to predict the land area use of invertebrates as accurately as the land area use of mammals, birds, and reptiles. However, it is important that invertebrates are not omitted when considering the natural resource demand of ecosystems. Invertebrates perform critical roles in ecosystem functionality and warrant the same level of consideration as the other biological classes. By including invertebrates in our proposed set of ecosystem wellbeing needs, we set a precedent; we encourage others to describe the natural resource requirements of invertebrates alongside other, better-studied biological classes. Moreover, we encourage further data collection to describe the land area use of invertebrates more accurately than was possible here.

10.2 Appendix 2: Supplementary information for Chapter 4, 'Including local terrestrial biodiversity impacts in life cycle assessment'

10.2.1 Terrestrial species supported by sedum roofs at the Meridian Water Development site in London, UK

Although brownfield sites are not 'natural' habitats, they can support non-zero amounts of urban biodiversity [556]. However, the presence of mammal, bird, reptile, and invertebrate species on the brownfield site is difficult to predict accurately, unless extensive surveys are carried out. Given that no such data is available, we compile a species list (Table A3) using ecological literature, expert opinion, and the online (citizen-science) data repository iNaturalist [452]. This species list will comprise resident and transient species. In line with our scope, we use the data range April-May (2021) when using iNaturalist and include only terrestrial mammals, birds, reptiles, and invertebrates likely to use the MWD site. We also include priority species – those that are identified as threatened and requiring conservation action under the UK Biodiversity Action Plan [454] – in our list. Here, we refer to the Biodiversity Action Plan drawn-up by Enfield council [453]. A full species list is presented below.

Table A3 Terrestrial species expected to make use of Meridian One

Species	Common name	Biological class	Diet type
Vulpes vulpes	Red fox	Mammal	Omnivore
Rattus norvegicus	Brown rat	Mammal	Omnivore
Pipistrellus pipistrellus	Common pipistrelle	Mammal	Carnivore
Pipistrellus pygmaeus	Soprano pipistrelle	Mammal	Carnivore
Pipistrellus nathusii	Nathusius' pipistrelle	Mammal	Carnivore
Nyctalus noctula	Noctule bat	Mammal	Carnivore
Nyctalus leisleri	Leisler's bat	Mammal	Carnivore
Myotis daubentonii	Daubenton's bat	Mammal	Carnivore
Plecotus auritus	Brown long-eared bat	Mammal	Carnivore
Erinaceus europaeus	European hedgehog	Mammal	Omnivore
Falco peregrinus	Peregrine falcon	Bird	Carnivore
Passer domesticus	House sparrow	Bird	Omnivore
Turdus philomelos	Song thrush	Bird	Omnivore
Sturnus vulgaris	European starling	Bird	Omnivore
Apus apus	Common swift	Bird	Carnivore
Aegithalos caudatus	Long-tailed tit	Bird	Omnivore
Carduelis carduelis	European goldfinch	Bird	Herbivore
Columba livia	Wood pigeon	Bird	Herbivore
Corvus corone	Carrion crow	Bird	Omnivore
Cyanistes caeruleus	Blue tit	Bird	Omnivore
Erithacus rubecula	European robin	Bird	Omnivore
Garrulus glandarius	Eurasian jay	Bird	Omnivore
Motacilla cinerea	Grey wagtail	Bird	Carnivore
Parus major	Great tit	Bird	Omnivore
Periparus ater	Coal tit	Bird	Omnivore
Pica pica	Eurasian magpie	Bird	Omnivore

Table A3 (continued)

Species	Common name	Biological class	Diet type
Prunella modularis	Dunnock	Bird	Omnivore
Psittacula krameri	Ring-necked parakeet	Bird	Herbivore
Troglodytes troglodytes	Eurasian wren	Bird	Omnivore
Turdus merula	Common blackbird	Bird	Omnivore
Zootoca vivipara	Common lizard	Reptile	Insectivore
Aglais io	European peacock butterfly	Invertebrate	Herbivore [a]
Agrotis exclamationis	Heart and dart moth	Invertebrate	Herbivore [a]
Amara aenea	Common sun beetle	Invertebrate	Herbivore
Anthocharis cardamines	Orange-tip butterfly	Invertebrate	Herbivore [a]
Anthophora plumipes	Hairy-footed flower bee	Invertebrate	Herbivore [a]
Aphidoidea	Aphids	Invertebrate	Herbivore
Apis mellifera	Western honeybee	Invertebrate	Herbivore [a]
Araneus diadematus	European garden spider	Invertebrate	Insectivore
Bibio marci	St mark's (hawthorn) fly	Invertebrate	Herbivore [a]
Bombus lapidaries	Red-tailed bumblebee	Invertebrate	Herbivore [a]
Bombus lucorum	White-tailed bumblebee	Invertebrate	Herbivore [a]
Bombus pratorum	Early bumblebee	Invertebrate	Herbivore [a]
Bombus terrestris	Buff-tailed bumblebee	Invertebrate	Herbivore [a]
Bombylius major	Greater bee fly	Invertebrate	Herbivore [a]
Chrysoperla carnea	Common green lacewing	Invertebrate	Omnivore [a]
Coccinella septempunctata	Seven-spot ladybird	Invertebrate	Insectivore
Cupido minimus	Small blue butterfly	Invertebrate	Herbivore [a]
Deilephila elpenor	Elephant hawk-moth	Invertebrate	Herbivore [a]
Gonepteryx rhamni	Common brimstone	Invertebrate	Herbivore [a]
Heriades truncorum	Large-headed resin bee	Invertebrate	Herbivore [a]
Ichneumonidae	Ichneumonidae wasp	Invertebrate	Herbivore [a]
Lycaena phlaeas	Small copper butterfly	Invertebrate	Herbivore [a]
Megachilidae?	Leafcutter bee	Invertebrate	Herbivore [a]
Mimas tiliae	Lime hawk-moth	Invertebrate	Herbivore [a]
Noctua comes	Lesser yellow underwing	Invertebrate	Herbivore [a]
Osmia rufa	Mason bee	Invertebrate	Herbivore [a]
Pararge aegeria	Speckled wood butterfly	Invertebrate	Herbivore [a]
Polygonia c-album	Comma butterfly	Invertebrate	Herbivore [a]
Pyrochroa serraticornis	Red-headed cardinal beetle	Invertebrate	Herbivore
Pyrrhosoma nymphula	Large red damselfly	Invertebrate	Insectivore
Salticus scenicus	Zebra jumping spider	Invertebrate	Insectivore
Segestria florentina	Tube-web spider	Invertebrate	Insectivore
Tetragnatha extensa	Stretch spider	Invertebrate	Insectivore
Tytthaspis sedecimpunctata	Sixteen-spot ladybird	Invertebrate	Insectivore
Vanessa cardui	Painted lady butterfly	Invertebrate	Herbivore [a]

[a] *nectivorous herbivore*

10.2.2 Estimating reasonable upper limits for species abundances at Meridian One

The food web proposed in the main text (Fig. 4.4) describes the relative abundances of species expected to make use of the Meridian One. When on-site, these species may be supported (directly or indirectly) by the sedum roofs. To ensure spatial consistency, data describing species abundances must be scaled to the developed site: 0.0709 km^2.

10.2.2.1 Transient bird populations

A 2021 bird survey by the British Trust for Ornithology (BTO) [557], [558] is used to estimate the population size of each bird species listed in Table A3. These survey data are collected annually across eighty 1 km^2 sites across London; it does not describe the absolute abundance of each species, but it does describe their relative abundances. We scale the data by estimating the abundance of one of the bird species listed (the Blackbird: 100 individuals km^{-1} [559]) across an area of 0.07 km^2 (i.e., the area of Meridian One). The scaled data describe the abundances of listed bird species one might expect to observe at Meridian One.

Of the species most likely consumed by the peregrine falcon, the most abundant appear to be the European Starling, Wood Pigeon, and House Sparrow. To overcome the lack of predation data for the Peregrine falcon (which may consume any number of bird species [560], [561]), it is assumed that these three species form the diet of the peregrine falcon.

10.2.2.2 Resident bird populations

Several nesting sites (bird boxes and bug hotels) are featured in the designs for Meridian One. According to current designs for Meridian One, there will be 11 bird boxes installed, which can be expected to house a breeding pair of birds. However, there is no guarantee that nest boxes will be occupied, nor which species will occupy them. The provision of energy, in this case from the sedum roofs, limits what biodiversity, transient or otherwise, is supported by resources on-site. With a focus on energy provision by sedum roofs, the availability of nesting sites does not affect our results at this stage.

10.2.2.3 Bats

There are relatively few studies of London's urban bat populations [562]. There are more data on distributions but fewer on species' abundances. Table A4 describes the relative abundances of the bat species expected to use Meridian One in a transient way. That is, they do not reside on-site, but use its resources as a 'visitor'. These relative abundances are sourced from 'The London Bat Atlas' [562], whereas the proposed abundances of each species are calculated using a 2019 urban bat survey undertaken in East London. In this survey, an average of 128 recordings $night^{-1}$ were made [563] across 14 sites of varied size (0.022 km^2 average). Owing to difficulties in predicting abundances from ecoacoustic data, we assume that each recording represents a single bat.

Table A4 Estimated bat abundances at Meridian One

Species	Common name	Relative species abundance (%)	Species abundance (individuals)	Scaled to Meridian One
Pipistrellus pipistrellus	Common pipistrelle	27	34	108
Pipistrellus pygmaeus	Soprano pipistrelle	25	32	102
Pipistrellus nathusii	Nathusius' pipistrelle	3	3	10
Nyctalus noctula	Noctule bat	23	29	92
Nyctalus leisleri	Leisler's bat	3	3	10
Myotis daubentonii	Daubenton's bat	15	19	60
Plecotus auritus	Brown long-eared bat	5	6	19
		100	**128**	**401**

10.2.2.4 Common lizard

Common lizards live underground, in small burrows, or beneath logs or rocks on the ground. Individuals may occupy sedum roofs, but it is unclear above what elevation sedum roofs becomes inaccessible. Common Lizards have been recorded on brownfield sites and therefore may have been present at the MWD site prior to development. Owing to a lack of pre-development data, a pre-development survey at a similar, but smaller (0.85 ha) site in the London borough of Southwark is used. Here, an average of 3 common lizards were observed [564]. Given that Meridian One is roughly twice the size of the Southwark site, we assume up to 6 common lizards may use resources at Meridian One.

10.2.2.5 European hedgehog

Survey work carried out by the public describes urban hedgehog populations in London [565], offering an insight into how many hedgehogs might use resources at Meridian One. While a data gap exists for the MWD site, owing to a lack of residential access, areas nearby, of similar size, support European hedgehog populations of five or ten individuals. We therefore estimate up to five hedgehogs may use resources at Meridian One. We assume that these hedgehogs satisfy their metabolic requirement through natural predation and are not fed (e.g., pet food) by residents.

10.2.2.6 Red fox

Urban red fox populations are usually made up of two to six individuals [566]. In urban areas, the average population density of red foxes is 16 individuals km^{-2}. Covering 0.07 km^2, Meridian One would be used by one individual. A study on urban foxes in Oxfordshire describes their diet in urban areas [240], which we apply in our calculations: 2% plant biomass, 5% invertebrates, 37% meat, and 56% waste.

10.2.2.7 Invertebrates

Data on the abundance of invertebrates on sedum roofs is used to describe the invertebrate population across the 0.01 km^2 of sedum roofs at Meridian One. In a survey of urban sedum roofs in London [475], species richness (136 species) is recorded, but species abundance is not. Data on the abundance of invertebrates on urban sedum roofs is however reported in a Melbourne (Australia) study with similar climate [567]. This is used here in lieu of case-specific data. The abundances of five genus types are recorded: Hymenoptera, Hemiptera, Coleoptera, Araneae, and Lepidoptera. The invertebrates species named in Table A3 are allocated to these five genus types; three of which correspond to predatory species: Hymenoptera, Coleoptera, and Araneae. By scaling the reported results to our study area, we approximate the relative population sizes for each invertebrate species in Table A3. The results are presented in Table A5:

Table A5 Relative abundance of invertebrates on sedum roofs at Meridian One

Genus	Quantity across 1 km^2 green roof (individuals)	Observed quantity across 0.018 km^2 green roof (individuals)	Number of species per genus expected on-site (individuals)	Minimum expected population per species [a] (individuals)	Relative abundance per species (%)
Araneae	3,846	41	4	11	13
Coleoptera	1,923	21	4	6	7
Hymenoptera	8,974	97	14	7	9
Hemiptera	5,128	56	1	56	68
Lepidoptera	1,282	14	12	2	2
	21,154	**229** [b]	**146**		**100**

[a] Values rounded-up to avoid incomplete individuals; [b] used as a 'minimum expected population' to validate the invertebrate population supported by the sedum roofs in our ecological model.

Using these relative abundances, we inform our calculations when describing the body mass and dietary requirements of the 'aggregated invertebrate'. Our aggregation is therefore representative of the invertebrate population supported by sedum roofs as it accounts for the different relative abundances of each genus.

10.2.3 Adjusting for land use requirements exceeding the case study site

Some animals will use Meridian One in a transient way, meaning that they make use of resources across areas larger than Meridian One. Where an individual uses resources beyond the case study site, its survival does not depend on the production of energy at Meridian One (i.e., by the sedum roofs) alone. In these cases, we adjust individual metabolic requirements to reflect the use of Meridian One as a proportion of the total land use requirement of each species (see Table A29, Appendix 5). For example: Meridian One constitutes only 70% of the brown rat's land area requirement, so only 70% of its

energetic requirements is allocated to Meridian One. We assume that adequate resources are available off-site. Further consideration of off-site resources is beyond the scope of this work at this stage.

10.2.4 Assumptions made when describing the species population at Meridian One

Where explicit predation data is available – such that predator-prey relationships are described at the species level (predation matrices for example) – a local species population can be described without the need for aggregation. Such data is lacking for the species in Table A3. The type of food consumed (e.g., invertebrates) is known, but the exact species of invertebrate(s) being consumed is not. For this reason, we describe the unit population in a more general way, aggregating the four food types are identified in this study: 'plants', 'invertebrates', 'meat', and 'waste' (used here to mean food scraps). Given that most species in Table A3 are either herbivorous or insectivorous, this aggregation affects only 'invertebrates' as prey species. The exception in Table A3 is the Peregrine falcon (*Falco peregrinus*): the Peregrine falcon is the only species that consumes meat (smaller birds). Restricted to the species described in Table A3, we assume the peregrine falcon consumes house sparrow, song thrush, European starling, and common swift in equal amounts (by calorific value).

Aggregation is also necessary when describing the calorific values of 'invertebrates (as prey)', 'meat' and 'waste'. Calorific values of different types of vegetation (comprising 'plants') are documented: leaves (19,710 kJ kg^{-1} [419]), wood (19,330 kJ kg^{-1} [419]), nectar (17,000 kJ kg^{-1} [458]), berries (19,000 kJ kg^{-1} [419]), roots (19,020 kJ kg^{-1} [419]) and seeds (5,000 kJ kg^{-1} [568]). These are aggregated to describe a calorific value for 'plants': 16,510 kJ kg^{-1}. However, less is known about the other food-types; particularly the species comprising Table A3. Owing to a paucity of calorific data, we describe a calorific value for 'invertebrates', 20,021 kJ kg^{-1} [569], [570], [571], [572], [573] (Table A33, Appendix 5), and 'meat', 5,767 kJ kg^{-1} [574] (Table A34, Appendix 5), using calorific data from various sources. Finally, 'waste' (i.e., food waste) will be a combination of meat and plants (assuming a typical human diet of meat, leaves, roots, and berries). Thus, by aggregating the calorific values of 'meat', 'leaves', 'roots' and 'berries', a calorific value for 'waste' is described: 15,874 kJ kg^{-1}.

To determine the metabolic and other requirements of 'invertebrates', we construct an invertebrate that is representative of all those comprising the food web. To ensure this aggregation is representative of our case study, the following relative abundances are applied: Araneae, 13%; Coleoptera, 7%; Hymenoptera, 9%; Hemiptera, 68%; and Lepidoptera, 2% (Table A32, Appendix 5). The adult body mass of this aggregated 'invertebrate' is 1.28×10^{-4} kg. Similarly, its dietary needs are based on the relative proportion of carnivorous, herbivorous, and nectivorous invertebrate species in the food web:

9% from plants (vegetation), 69% from plants (nectar), 23% from invertebrates, 0% from meat, and 0% from waste.

We aggregate 'birds' and 'bats' in a similar way, using the data in the food web. For birds: average body mass = 0.10 kg individual^{-1}, and diet: 35% from plant biomass, 63% from invertebrates, 2% from waste. For bats: average body mass = 0.01 kg individual^{-1}, and diet: insectivorous; 100% from invertebrates.

10.2.5 Energy (nectar and plant biomass) production on the sedum roofs

Assuming each plant has a 60 mm radial spread at substrate level [575], 812,389 plants cover the sedum roofs. Next, assuming the sedums have an average plant-biomass of 5.39x10^{-5} kg plant^{-1} day^{-1} [576]: 43.8 kg day^{-1} (8.6x10^{5} kJ plant biomass day^{-1}) plant-biomass is produced by the 18 green roofs. Finally, sedum can produce 0.058 m^{3} nectar km^{-2} day^{-1} in temperate climates [459]. This value corresponds to milder spring temperatures but is used owing to a lack of data specific to London. This equates 6.2x10^{-4} m^{3} nectar day^{-1} across the total area of sedum roofs. This equates to, assuming a density and calorific value of 1,175 kg m^{-3} and 17,000 kJ kg^{-1} respectively [457], [458].

Of the two types of energy produced by the sedum roofs (nectar and plant-biomass), nectar production is most limited. Nectar production therefore constrains the local species population they support. Ample vegetation exists to satisfy herbivory alongside nectar consumption. In Table A27, Appendix 5, it is determined that an individual (aggregated) invertebrate requires 19 kJ individual^{-1} day^{-1} plant biomass. Nectar represents 69% of this need with an assimilation efficiency of 0.59. Each individual invertebrate therefore requires 22.2 kJ nectar individual^{-1} day^{-1}. The production of 12,458 kJ nectar day^{-1} on the sedum roofs satisfies the metabolic needs of 560 individual invertebrates.

These 560 invertebrates first support the invertebrate population itself, then other species in higher trophic levels (Fig. 4.6, 'Section 4.3', Chapter 4). With 23% of their energy coming from the consumption of other invertebrates, with an assimilation efficiency of 0.75%, 267 invertebrates day^{-1} are consumed by the invertebrate population itself. 233 individuals day^{-1} remain, which are consumed by species occupying higher trophic levels. Equal distribution amongst consumers is assumed. Those species are then themselves distributed equally amongst their predators, and so on (see Fig. 4.6, 'Section 4.3', Chapter 4). Where the production of nectar does not limit a local species' population size (i.e.,

herbivorous birds: wood pigeon, rink-necked parakeet, and European goldfinch), typical population density values (Table A29, Appendix 5) are used as upper limits of population size. This is also done where an abundant wood pigeon population may supplement energy sources supporting the red fox and peregrine falcon: 1 red fox and 1 peregrine falcon is the reasonable upper limit for Meridian One, based on its size and location [557], [558], [577].

10.2.6 Endpoint characterisation factor for the sedum roofs at Meridian One

Endpoint characterisation factors ($CF_{endpoint}$) are used to convert actions (environmental interventions) to quantitative end-point impacts (e.g., damage to ecosystems). Here we are interested in end-point impacts pertaining to biodiversity loss. Fig. A2 describes the relevant impact pathway for land use in our case study, showing the midpoint and endpoint impact stages.

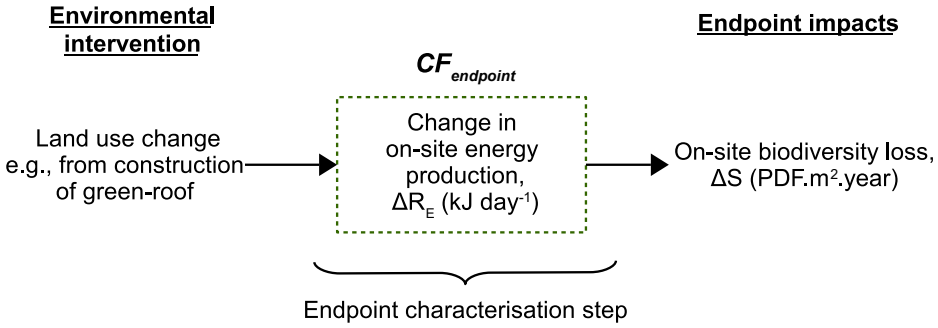


Figure A2 Impact pathway for the creation of a green roof, as land use change, to potential impacts to local (i.e., in-situ) terrestrial biodiversity change

It is reasonable to attribute the biodiversity gain associated with the extensive green roof to the 'use phase' at Meridian One. That is: when Meridian One is occupied, after the transformation stage and before end-of-life demolition. Biodiversity – particularly species in high taxa – are unlikely to be supported upon the completion of land transformation. It will take time for the green roof to become established and for species to make use of it.

Taking inspiration from Figure 11.2 (p. 90) in Huijbregts et al. (2017) [84], Fig. A3 illustrates the differences between the two development scenarios at Meridian One, showing that the two development scenarios represent different states of occupation. Occupation as extensive green roof, $Q_{green\ roof}$, represents a state of higher ecosystem quality than occupation as urban area, Q_{urban} (i.e., the conventional roof). It is important to note that while higher quality ecosystems may support more biodiversity than lower quality ecosystems, higher ecosystem quality does not necessarily mean lower ecological impact. Even if the ecological impact associated with sedum roofs is found to be equal to

that of other urban areas, the creation of distinctive characterisation factors will help to achieve greater granularity in applying LCA.

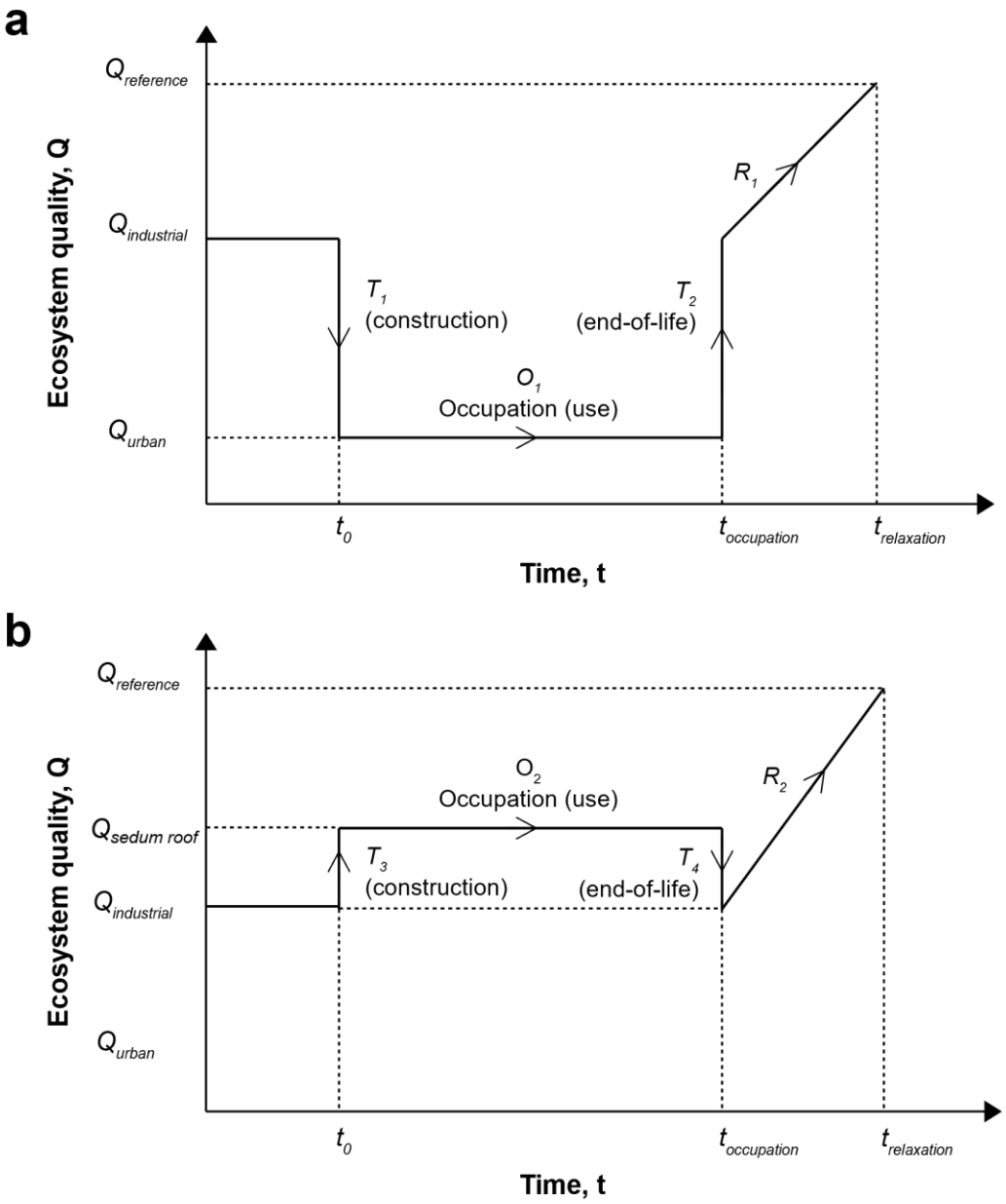


Figure A3 Land use during the development of Meridian One and its impacts on ecosystem quality. (**a**) describes the land transformation and occupation associated with concrete roofs and the areas surrounding the 18 residential buildings at Meridian One. It involves two transformation stages (T_1: 'from urban/industrial fallow' + 'to urban'; and T_2: 'from urban' + 'to urban/industrial fallow'). (**b**) describes the land transformation and occupation associated with sedum roofs (which we aim to define). It involves two transformation stages (T_3: 'from urban/industrial fallow' + 'to urban, extensive green roof'; and T_4: 'from urban, extensive green roof' + 'to urban/industrial fallow'). The land transformations, equal and opposite, cancel to net zero. This is because Meridian One is assumed to return to its starting land use type at end-of-life. O represents occupation, with O_1 and O_2 both describing 'occupation as urban'. R_i represents relaxation – natural restoration to the site's reference habitat type (broadleaf forest). The sedum roofs are attributed a higher ecosystem quality, $Q_{sedum\ roof}$, than the pre-developed, industrial site, $Q_{industrial}$ and the urban areas, Q_{urban}. $Q_{reference}$ represents the ecosystem quality of the reference habitat type: European broadleaf forest. The figure is illustrative and not to scale

In Fig. A3 and in our case study, modelled land transformations, in each design scenario (i.e., T_1 and T_2, and T_3 and T_4) produce a lifetime net zero impact. This is because it is assumed that the site returns to its original state (urban/industrial fallow) at end-of-life. Consequently, land transformations do not contribute to the overall ecological impacts calculated here. Different end-of-life land use scenarios can offset portions of lifetime ecological impacts at end-of-life; however, the offsetting capabilities of transformation to and from feasible alternatives ('arable, unspecified use'; pasture, 'man-made') were found to be insignificant – orders of magnitude less – when compared to the ex-situ ecological impacts of the development.

10.2.7 Alpha (α) diversity index based on calculated species abundance

A terrestrial population of 53 species is expected to be supported by the sedum roofs at Meridian One. This population has an abundance of 673 individuals. While species abundance is not included in the IMPACT World+ assessment method, it provides information on the quality of the population supported. This terrestrial population has an alpha (local) diversity index of 0.4, according to the Shannon entropy index [471]; this value represents a low species diversity. A poorly diverse population is likely to be less resilient to disturbances than a richly diverse population. Furthermore, a population with low diversity will likely contribute less to ecosystem functionality at the system level. This should be considered at the design stage of biodiversity-supporting infrastructures, especially where the aim of their employment is to contribute to the creation of an ecosystem. For example, by means of ecological engineering [56].

10.3. Appendices for Chapter 5, 'Quantifying the land, water, and energy requirements of wild species in terrestrial ecosystems: A case study of the American kestrel (Falco sparverius) on St Martin Island, Caribbean'

10.3.1 Interspecies interactions as depicted in the kestrel food web

To guide the reader, the interspecies interactions that make-up the kestrel food web (Fig. 6.1) in Chapter 5 are detailed in Table A6. These interactions are described in Goldwasser and Roughgarden (1993) [87].

Table A6 Interspecies interactions as depicted in the kestrel food web. A cross (**X**) indicates that the species in the 'consumed' row is consumed by the corresponding species in the 'consumer' column, constituting a flow of energy, from consumed species *j* to consumer species *i*, in the food web

'Consumer' (Energy flows to…)	**'Consumed' (Energy flows from…)**									
	Elaenia	Bullfinch	Grassquit	Hummingbirds	Bananquit	Coleoptera (adult)	Orthoptera	Centipede	Millipede	*Anolis gingivinus*
Kestrel	X	X	X	X	X	X	X	X	X	X
Elaenia						X				
Bullfinch										
Grassquit							X			
Hummingbirds						X				
Bananquit										
Anolis gingivinus						X	X	X	X	
Anolis pogus						X	X	X		
Orthoptera										
Isoptera										
Hemiptera										
Homoptera										
Thysanoptera										
Coleoptera (adult)										
Coleoptera (larvae)										
Diptera (adult)										
Diptera (larvae)										
Adult spider										
Juvenile spider										
Ants										
Other hymenoptera										
Lepidoptera (adult)										
Lepidoptera (larvae)										
Centipede										
Millipede										

Table A6 (continued)

'Consumer' (Energy flows to…)	**'Consumed' (Energy flows from…)**									
	*Anolis pogu*s	Other hymenoptera	Diptera (adult)	Ants	Lepidoptera (adult)	Lepidoptera (larvae)	Hemiptera	Homoptera	Adult spider	Coleoptera (larvae)
Kestrel	X									
Elaenia					X		X		X	
Bullfinch				X						X
Grassquit						X		X		
Hummingbirds		X	X	X				X	X	
Bananquit		X	X		X	X	X	X		
Anolis gingivinus	X	X	X	X	X	X	X	X		X
Anolis pogus		X	X	X	X	X	X	X		X
Orthoptera				X				X		X
Isoptera										
Hemiptera		X	X		X	X		X		
Homoptera										
Thysanoptera										
Coleoptera (adult)										X
Coleoptera (larvae)										
Diptera (adult)										
Diptera (larvae)										
Adult spider		X	X		X	X	X	X		
Juvenile spider				X						
Ants										
Other hymenoptera										
Lepidoptera (adult)										
Lepidoptera (larvae)										
Centipede				X						X
Millipede										

Table A6 (continued)

'Consumer' (Energy flows to…)	**'Consumed' (Energy flows from…)**					
	Isopoda	Diptera (larvae)	Isoptera	Juvenile spider	Thysanoptera	Vegetation
Kestrel						
Elaenia						X
Bullfinch	X	X	X			X
Grassquit				X		X
Hummingbirds						X
Bananquit					X	X
Anolis gingivinus	X	X	X	X	X	
Anolis pogus	X	X	X	X	X	
Orthoptera	X	X	X			X
Isoptera						X
Hemiptera					X	X
Homoptera						X
Thysanoptera						X
Coleoptera (adult)	X	X	X			X
Coleoptera (larvae)						X
Diptera (adult)						X
Diptera (larvae)						X
Adult spider					X	
Juvenile spider						
Ants						X
Other hymenoptera						X
Lepidoptera (adult)						X
Lepidoptera (larvae)						X
Centipede	X	X	X	X		
Millipede						X

10.4 Appendix 4: Supplementary data for Chapter 3, ‘Quantifying the natural resource requirements (or ‘ecological needs’) of terrestrial animals analogous to human wellbeing needs’

The regression analyses presented in Chapter 3, ‘Quantifying ecological needs analogous to human (wellbeing) needs’, used data from both digital repositories and literature. The data is predominantly aggregated from TetraDensity and PanTHERIA, describing two variables, adult body mass (kg), and population density (individuals km^{-2}), across four biological classes: mammals, birds, reptiles, and invertebrates. Mammals and birds were further disaggregated by dietary guild: carnivore, omnivore, herbivore; reptiles and invertebrates were not.

Including these data (~17,000 datapoints) physically in this thesis is impractical. The reader is instead directed to the data in the form of an online repository:

- Mason, A. R. (2023). Appendix 4: Supplementary data for Chapter 3 in the thesis: 'Quantifying the natural resource requirements of terrestrial ecosystems for the management of biodiversity alongside human development' [Data set]. Zenodo. https://doi.org/10.5281/zenodo.11217968

Tables A15, containing the Natural Resource Laboratory (NCL) results, and Table A16, containing data on the quantification of a ‘bioproductive land area’ value based on herbivory at the NCL, are more manageable and presented overleaf; however, they are also included in the online repository alongside Tables A7-A14.

Table A15 Species observed at the NCL site, and the land area needed to support that population

Species	Biological class	Adult mass (kg)	Dietary guild	Land use (km^2)	Water consumption (m^3 day^{-1} $individual^{-1}$)	Energy requirement (kJ day^{-1} $individual^{-1}$)
Meles meles	Mammal	12.00	Omnivore	0.5	0.00	3516
Lepus timidus	Mammal	3.10	Herbivore	0.0	0.00	1430
Martes martes	Mammal	1.30	Carnivore	0.5	0.00	989
Sciurus vulgaris	Mammal	0.33	Herbivore	0.0	0.00	336
Sciurus vulgaris	Mammal	0.33	Herbivore	0.0	0.00	336
Cervus nippon	Mammal	42.00	Herbivore	0.3	0.00	7699
Cervus nippon	Mammal	42.00	Herbivore	0.3	0.00	7699
Apodemus sylvaticus	Mammal	0.02	Omnivore	0.0	0.00	51
Apodemus sylvaticus	Mammal	0.02	Omnivore	0.0	0.00	51
Hirundo rustica	Bird	0.02	Carnivore	0.0	0.00	74
Hirundo rustica	Bird	0.02	Carnivore	0.0	0.00	74
Turdus merula	Bird	0.10	Omnivore	0.1	0.00	169
Turdus merula	Bird	0.10	Omnivore	0.1	0.00	169
Cyanistes caeruleus	Bird	0.01	Omnivore	0.0	0.00	40
Cyanistes caeruleus	Bird	0.01	Omnivore	0.0	0.00	40
Cyanistes caeruleus	Bird	0.01	Omnivore	0.0	0.00	40
Cyanistes caeruleus	Bird	0.01	Omnivore	0.0	0.00	40
Fringilla coelebs	Bird	0.02	Carnivore	0.1	0.00	83
Fringilla coelebs	Bird	0.02	Carnivore	0.1	0.00	83
Fringilla coelebs	Bird	0.02	Carnivore	0.1	0.00	83
Fringilla coelebs	Bird	0.02	Carnivore	0.1	0.00	83
Fringilla coelebs	Bird	0.02	Carnivore	0.1	0.00	83
Fringilla coelebs	Bird	0.02	Carnivore	0.1	0.00	83
Fringilla coelebs	Bird	0.02	Carnivore	0.1	0.00	83
Fringilla coelebs	Bird	0.02	Carnivore	0.1	0.00	83
Fringilla coelebs	Bird	0.02	Carnivore	0.1	0.00	83
Fringilla coelebs	Bird	0.02	Carnivore	0.1	0.00	83
Fringilla coelebs	Bird	0.02	Carnivore	0.1	0.00	83
Fringilla coelebs	Bird	0.02	Carnivore	0.1	0.00	83
Fringilla coelebs	Bird	0.02	Carnivore	0.1	0.00	83
Fringilla coelebs	Bird	0.02	Carnivore	0.1	0.00	83
Phylloscopus collybita	Bird	0.01	Carnivore	0.0	0.00	43
Phylloscopus collybita	Bird	0.01	Carnivore	0.0	0.00	43
Periparus ater	Bird	0.01	Omnivore	0.0	0.00	38
Periparus ater	Bird	0.01	Omnivore	0.0	0.00	38
Periparus ater	Bird	0.01	Omnivore	0.0	0.00	38
Periparus ater	Bird	0.01	Omnivore	0.0	0.00	38
Loxia curvirostra	Bird	0.04	Herbivore	0.0	0.00	125
Loxia curvirostra	Bird	0.04	Herbivore	0.0	0.00	125
Cuculus canorus	Bird	0.11	Carnivore	0.2	0.00	267
Cuculus canorus	Bird	0.11	Carnivore	0.2	0.00	267

Table A15 (continued)

Species	Biological class	Adult mass (kg)	Dietary guild	Land use (km^2)	Water consumption (m^3 day^{-1} $individual^{-1}$)	Energy requirement (kJ day^{-1} $individual^{-1}$)
Cuculus canorus	Bird	0.11	Carnivore	0.2	0.00	267
Cuculus canorus	Bird	0.11	Carnivore	0.2	0.00	267
Cuculus canorus	Bird	0.11	Carnivore	0.2	0.00	267
Cuculus canorus	Bird	0.11	Carnivore	0.2	0.00	267
Cuculus canorus	Bird	0.11	Carnivore	0.2	0.00	267
Cuculus canorus	Bird	0.11	Carnivore	0.2	0.00	267
Phasianus colchicus	Bird	1.05	Omnivore	0.2	0.00	739
Numenius arquata	Bird	0.73	Omnivore	0.2	0.00	588
Cinclus cinclus	Bird	0.06	Carnivore	0.1	0.00	178
Regulus regulus	Bird	0.01	Carnivore	0.0	0.00	32
Regulus regulus	Bird	0.01	Carnivore	0.0	0.00	32
Regulus regulus	Bird	0.01	Carnivore	0.0	0.00	32
Regulus regulus	Bird	0.01	Carnivore	0.0	0.00	32
Mergus merganser	Bird	1.50	Omnivore	0.2	0.00	924
Parus major	Bird	0.02	Omnivore	0.0	0.00	53
Parus major	Bird	0.02	Omnivore	0.0	0.00	53
Parus major	Bird	0.02	Omnivore	0.0	0.00	53
Parus major	Bird	0.02	Omnivore	0.0	0.00	53
Motacilla cinerea	Bird	0.02	Carnivore	0.0	0.00	71
Delichon urbicum	Bird	0.01	Carnivore	0.0	0.00	62
Delichon urbicum	Bird	0.01	Carnivore	0.0	0.00	62
Garrulus glandarius	Bird	0.16	Omnivore	0.1	0.00	227
Acanthis cabaret	Bird	0.01	Omnivore	0.0	0.00	41
Acanthis cabaret	Bird	0.01	Omnivore	0.0	0.00	41
Aegithalos caudatus	Bird	0.01	Omnivore	0.0	0.00	36
Turdus viscivorus	Bird	0.12	Omnivore	0.1	0.00	189
Turdus viscivorus	Bird	0.12	Omnivore	0.1	0.00	189
Turdus viscivorus	Bird	0.12	Omnivore	0.1	0.00	189
Turdus viscivorus	Bird	0.12	Omnivore	0.1	0.00	189
Milvus milvus	Bird	1.10	Carnivore	1.7	0.00	1352
Phoenicurus phoenicurus	Bird	0.02	Omnivore	0.0	0.00	51
Phoenicurus phoenicurus	Bird	0.02	Omnivore	0.0	0.00	51
Erithacus rubecula	Bird	0.02	Omnivore	0.0	0.00	59
Erithacus rubecula	Bird	0.02	Omnivore	0.0	0.00	59
Erithacus rubecula	Bird	0.02	Omnivore	0.0	0.00	59
Erithacus rubecula	Bird	0.02	Omnivore	0.0	0.00	59
Erithacus rubecula	Bird	0.02	Omnivore	0.0	0.00	59
Erithacus rubecula	Bird	0.02	Omnivore	0.0	0.00	59
Erithacus rubecula	Bird	0.02	Omnivore	0.0	0.00	59
Erithacus rubecula	Bird	0.02	Omnivore	0.0	0.00	59

Table A15 (continued)

Species	Biological class	Adult mass (kg)	Dietary guild	Land use (km^2)	Water consumption (m^3 day^{-1} $individual^{-1}$)	Energy requirement (kJ day^{-1} $individual^{-1}$)
Erithacus rubecula	Bird	0.02	Omnivore	0.0	0.00	59
Erithacus rubecula	Bird	0.02	Omnivore	0.0	0.00	59
Erithacus rubecula	Bird	0.02	Omnivore	0.0	0.00	59
Spinus spinus	Bird	0.01	Herbivore	0.0	0.00	60
Spinus spinus	Bird	0.01	Herbivore	0.0	0.00	60
Spinus spinus	Bird	0.01	Herbivore	0.0	0.00	60
Spinus spinus	Bird	0.01	Herbivore	0.0	0.00	60
Spinus spinus	Bird	0.01	Herbivore	0.0	0.00	60
Turdus philomelos	Bird	0.07	Omnivore	0.1	0.00	132
Turdus philomelos	Bird	0.07	Omnivore	0.1	0.00	132
Turdus philomelos	Bird	0.07	Omnivore	0.1	0.00	132
Turdus philomelos	Bird	0.07	Omnivore	0.1	0.00	132
Turdus philomelos	Bird	0.07	Omnivore	0.1	0.00	132
Muscicapa striata	Bird	0.02	Carnivore	0.0	0.00	65
Muscicapa striata	Bird	0.02	Carnivore	0.0	0.00	65
Muscicapa striata	Bird	0.02	Carnivore	0.0	0.00	65
Muscicapa striata	Bird	0.02	Carnivore	0.0	0.00	65
Muscicapa striata	Bird	0.02	Carnivore	0.0	0.00	65
Certhia familiaris	Bird	0.01	Omnivore	0.0	0.00	37
Certhia familiaris	Bird	0.01	Omnivore	0.0	0.00	37
Certhia familiaris	Bird	0.01	Omnivore	0.0	0.00	37
Anthus trivialis	Bird	0.02	Omnivore	0.0	0.00	65
Anthus trivialis	Bird	0.02	Omnivore	0.0	0.00	65
Anthus trivialis	Bird	0.02	Omnivore	0.0	0.00	65
Phylloscopus trochilus	Bird	0.01	Carnivore	0.0	0.00	45
Phylloscopus trochilus	Bird	0.01	Carnivore	0.0	0.00	45
Phylloscopus trochilus	Bird	0.01	Carnivore	0.0	0.00	45
Phylloscopus trochilus	Bird	0.01	Carnivore	0.0	0.00	45
Phylloscopus trochilus	Bird	0.01	Carnivore	0.0	0.00	45
Phylloscopus trochilus	Bird	0.01	Carnivore	0.0	0.00	45
Phylloscopus trochilus	Bird	0.01	Carnivore	0.0	0.00	45
Phylloscopus trochilus	Bird	0.01	Carnivore	0.0	0.00	45
Phylloscopus trochilus	Bird	0.01	Carnivore	0.0	0.00	45
Phylloscopus trochilus	Bird	0.01	Carnivore	0.0	0.00	45

Table A15 (continued)

Species	Biological class	Adult mass (kg)	Dietary guild	Land use (km^2)	Water consumption (m^3 day^{-1} $individual^{-1}$)	Energy requirement (kJ day^{-1} $individual^{-1}$)
Phylloscopus trochilus	Bird	0.01	Carnivore	0.0	0.00	45
Phylloscopus trochilus	Bird	0.01	Carnivore	0.0	0.00	45
Phylloscopus trochilus	Bird	0.01	Carnivore	0.0	0.00	45
Phylloscopus trochilus	Bird	0.01	Carnivore	0.0	0.00	45
Columba palumbus	Bird	0.49	Omnivore	0.1	0.00	458
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Troglodytes troglodytes	Bird	0.01	Omnivore	0.0	0.00	37
Zootoca vivipara	Reptile	0.01	No data	0.0	No data	2
Zootoca vivipara	Reptile	0.01	No data	0.0	No data	2
Scarabaeidae sp.	Invertebrate	0.00	No data	0.0	No data	46
Euthrix potatoria	Invertebrate	0.00	No data	0.0	No data	96
Scarabaeidae sp.	Invertebrate	0.00	No data	0.0	No data	46
Scarabaeidae sp.	Invertebrate	0.00	No data	0.0	No data	46
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17

Table A15 (continued)

Species	Biological class	Adult mass (kg)	Dietary guild	Land use (km^2)	Water consumption (m^3 day^{-1} $individual^{-1}$)	Energy requirement (kJ day^{-1} $individual^{-1}$)
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pieris napi	Invertebrate	0.00	No data	0.0	No data	17
Pyrrhosoma nymphula	Invertebrate	0.00	No data	0.0	No data	4
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Anthocharis cardamines	Invertebrate	0.00	No data	0.0	No data	17
Pararge aegeria	Invertebrate	0.00	No data	0.0	No data	17
				Total	**0.01**	**37,158**

Table A16 Quantifying a 'bio-productive land area requirement' based on herbivory across species at the NCL site

Species	Biological class	Adult body mass (kg)	Dietary guild	Energy use (kJ individual^{-1} day^{-1})	Assumed portion from plants	
					(%)	(kJ individual^{-1} day^{-1})
Zootoca vivipara	Reptile	0.01	Herbivore	2	100%	2
Zootoca vivipara	Reptile	0.01	Herbivore	2	100%	2
Scarabaeidae sp.	Invertebrate	0.00	Herbivore	46	100%	46
Euthrix potatoria	Invertebrate	0.00	Herbivore	96	100%	96
Scarabaeidae sp.	Invertebrate	0.00	Herbivore	46	100%	46
Scarabaeidae sp.	Invertebrate	0.00	Herbivore	46	100%	46
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pieris napi	Invertebrate	0.00	Herbivore	17	100%	17
Pyrrhosoma nymphula	Invertebrate	0.00	Herbivore	4	100%	4
Anthocharis cardamines	Invertebrate	0.00	Herbivore	17	100%	17
Anthocharis cardamines	Invertebrate	0.00	Herbivore	17	100%	17
Anthocharis cardamines	Invertebrate	0.00	Herbivore	17	100%	17

Table A16 (continued)

Species	Biological class	Adult body mass (kg)	Dietary guild	Energy use (kJ individual^{-1} day^{-1})	Assumed portion from plants	
					(%)	(kJ individual^{-1} day^{-1})
Anthocharis cardamines	Invertebrate	0.00	Herbivore	17	100%	17
Anthocharis cardamines	Invertebrate	0.00	Herbivore	17	100%	17
Anthocharis cardamines	Invertebrate	0.00	Herbivore	17	100%	17
Anthocharis cardamines	Invertebrate	0.00	Herbivore	17	100%	17
Anthocharis cardamines	Invertebrate	0.00	Herbivore	17	100%	17
Anthocharis cardamines	Invertebrate	0.00	Herbivore	17	100%	17
Anthocharis cardamines	Invertebrate	0.00	Herbivore	17	100%	17
Anthocharis cardamines	Invertebrate	0.00	Herbivore	17	100%	17
Anthocharis cardamines	Invertebrate	0.00	Herbivore	17	100%	17
Anthocharis cardamines	Invertebrate	0.00	Herbivore	17	100%	17
Anthocharis cardamines	Invertebrate	0.00	Herbivore	17	100%	17
Pararge aegeria	Invertebrate	0.00	Herbivore	17	100%	17
Lepus timidus	Mammal	3.10	Herbivore	1430	100%	1430
Sciurus vulgaris	Mammal	0.33	Herbivore	336	100%	336
Sciurus vulgaris	Mammal	0.33	Herbivore	336	100%	336
Cervus nippon	Mammal	42.00	Herbivore	7699	100%	7699
Cervus nippon	Mammal	42.00	Herbivore	7699	100%	7699
Loxia curvirostra	Bird	0.04	Herbivore	125	100%	125
Loxia curvirostra	Bird	0.04	Herbivore	125	100%	125
Spinus spinus	Bird	0.01	Herbivore	60	100%	60
Spinus spinus	Bird	0.01	Herbivore	60	100%	60
Spinus spinus	Bird	0.01	Herbivore	60	100%	60
Spinus spinus	Bird	0.01	Herbivore	60	100%	60
Spinus spinus	Bird	0.01	Herbivore	60	100%	60
Meles meles	Mammal	12.00	Omnivore	3516	75%	2637
Apodemus sylvaticus	Mammal	0.02	Omnivore	51	75%	38
Apodemus sylvaticus	Mammal	0.02	Omnivore	51	75%	38
Turdus merula	Bird	0.10	Omnivore	169	75%	127
Turdus merula	Bird	0.10	Omnivore	169	75%	127
Cyanistes caeruleus	Bird	0.01	Omnivore	40	75%	30
Cyanistes caeruleus	Bird	0.01	Omnivore	40	75%	30
Cyanistes caeruleus	Bird	0.01	Omnivore	40	75%	30
Cyanistes caeruleus	Bird	0.01	Omnivore	40	75%	30
Periparus ater	Bird	0.01	Omnivore	38	75%	28
Periparus ater	Bird	0.01	Omnivore	38	75%	28
Periparus ater	Bird	0.01	Omnivore	38	75%	28
Periparus ater	Bird	0.01	Omnivore	38	75%	28
Phasianus colchicus	Bird	1.05	Omnivore	739	75%	554
Numenius arquata	Bird	0.73	Omnivore	588	75%	441
Mergus merganser	Bird	1.50	Omnivore	924	75%	693
Parus major	Bird	0.02	Omnivore	53	75%	40

Table A16 (continued)

Species	Biological class	Adult body mass (kg)	Dietary guild	Energy use (kJ individual^{-1} day^{-1})	Assumed portion from plants	
					(%)	(kJ individual^{-1} day^{-1})
Parus major	Bird	0.02	Omnivore	53	75%	40
Parus major	Bird	0.02	Omnivore	53	75%	40
Parus major	Bird	0.02	Omnivore	53	75%	40
Garrulus glandarius	Bird	0.16	Omnivore	227	75%	170
Acanthis cabaret	Bird	0.01	Omnivore	41	75%	31
Acanthis cabaret	Bird	0.01	Omnivore	41	75%	31
Aegithalos caudatus	Bird	0.01	Omnivore	36	75%	27
Turdus viscivorus	Bird	0.12	Omnivore	189	75%	142
Turdus viscivorus	Bird	0.12	Omnivore	189	75%	142
Turdus viscivorus	Bird	0.12	Omnivore	189	75%	142
Turdus viscivorus	Bird	0.12	Omnivore	189	75%	142
Phoenicurus phoenicurus	Bird	0.02	Omnivore	51	75%	38
Phoenicurus phoenicurus	Bird	0.02	Omnivore	51	75%	38
Erithacus rubecula	Bird	0.02	Omnivore	59	75%	45
Erithacus rubecula	Bird	0.02	Omnivore	59	75%	45
Erithacus rubecula	Bird	0.02	Omnivore	59	75%	45
Erithacus rubecula	Bird	0.02	Omnivore	59	75%	45
Erithacus rubecula	Bird	0.02	Omnivore	59	75%	45
Erithacus rubecula	Bird	0.02	Omnivore	59	75%	45
Erithacus rubecula	Bird	0.02	Omnivore	59	75%	45
Erithacus rubecula	Bird	0.02	Omnivore	59	75%	45
Erithacus rubecula	Bird	0.02	Omnivore	59	75%	45
Erithacus rubecula	Bird	0.02	Omnivore	59	75%	45
Turdus philomelos	Bird	0.07	Omnivore	132	75%	99
Turdus philomelos	Bird	0.07	Omnivore	132	75%	99
Turdus philomelos	Bird	0.07	Omnivore	132	75%	99
Turdus philomelos	Bird	0.07	Omnivore	132	75%	99
Turdus philomelos	Bird	0.07	Omnivore	132	75%	99
Certhia familiaris	Bird	0.01	Omnivore	37	75%	28
Certhia familiaris	Bird	0.01	Omnivore	37	75%	28
Certhia familiaris	Bird	0.01	Omnivore	37	75%	28
Anthus trivialis	Bird	0.02	Omnivore	65	75%	49
Anthus trivialis	Bird	0.02	Omnivore	65	75%	49
Anthus trivialis	Bird	0.02	Omnivore	65	75%	49
Columba palumbus	Bird	0.49	Omnivore	458	75%	343
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28

Table A16 (continued)

Species	Biological class	Adult body mass (kg)	Dietary guild	Energy use (kJ individual^{-1} day^{-1})	Assumed portion from plants	
					(%)	(kJ individual^{-1} day^{-1})
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
Troglodytes troglodytes	Bird	0.01	Omnivore	37	75%	28
			Total	**29,705**		**27,051**

10.5 Appendix 5: Supplementary data for Chapter 4, 'Including local terrestrial biodiversity impacts in life cycle assessment'

In this appendix, the data supplementary to Chapter 4, 'Including local terrestrial biodiversity impacts in life cycle assessment', is presented across Tables A17-A64. Table A29 and Table A30 are impractical to present here due to their size; however, these and all supplementary data for Chapter 4 are presented in the form of an online repository:

- Mason, A. R. (2023). Appendix 5: Supplementary data for Chapter 4 in the thesis: 'Quantifying the natural resource requirements of terrestrial ecosystems for managing biodiversity alongside human development' [Data set]. Zenodo. https://doi.org/10.5281/zenodo.11217971

Table A17 Dimensions of buildings to be constructed at Meridian One

Plot	Phase 1A or 1B	Building Footprint Area (m^2)	Number of floors	Total Area (slabs) (m^2)	External perimeter (m)	Total façade area (m^2)	Volume of concrete for columns (m^3) [a, b, c]
E 2.3B	A	440	3	1,319	120	1,044	148
E 2.3A	A	541	3	1,622	145	1,262	183
E 2.2	A	277	5	1,387	95	1,378	260
E 2.1	A	503	4	2,011	170	1,972	302
E 1.2	A	917	4	3,670	280	3,248	550
E 1.1	A	676	4	2,703	210	2,436	405
A2	A	1,277	6	7,664	325	5,655	1,724
A1	A	432	8	3,456	135	3,132	1,037
A3	A	493	12	5,921	140	4,872	2,665
C1	B	1,178	7	8,243	320	6,496	2,164
C2	B	524	8	4,191	165	3,828	1,257
C3	B	298	12	3,573	100	3,480	1,608
D1	B	1,022	8	8,176	285	6,612	2,453
D2	B	552	30	16,558	120	10,440	18,627
B1	B	524	24	12,582	130	9,048	11,323
B2	B	367	9	3,301	120	3,132	1,114
B3	B	374	11	4,110	105	3,350	1,695
B4	B	408	8	3,265	115	2,668	980
		10,803		**93,753**		**74,052**	**48,496**

[a] assume 4 columns of 30x30 cm every 24 m^2 = area of 0.36 m^2 of concrete in 24m^2 = 1.5% of the total area.

[b] Volume of concrete = total area of slabs x 0.015 x 2.5 m (height from slab to slab) x number of floors in each building.

[c] We assume: 2.9 m height per floor (including the false ceiling and slabs), 2.5 m floor to ceiling height, no basements.

Table A18 Materials for building frames at Meridian One

External perimeter (m)	Total façade area (m^2)	Volume of concrete for columns (m^3) [a]	Volume of concrete for slabs (m^3)	Cement (kg) [b]	Sand (kg) [c]	Gravel (kg) [d]	Water (kg) [e]
120	1,044	49	330	546,157	606,841	910,262	379,276
145	1,262	61	406	671,674	746,304	1,119,456	466,440
95	1,378	52	347	574,259	638,066	957,099	398,791
170	1,972	75	503	832,438	924,931	1,387,397	578,082
280	3,248	138	917	1,519,214	1,688,016	2,532,024	1,055,010
210	2,436	101	676	1,119,125	1,243,472	1,865,208	777,170
325	5,655	287	1,916	3,172,962	3,525,514	5,288,270	2,203,446
135	3,132	130	864	1,430,585	1,589,539	2,384,309	993,462
140	4,872	222	1,480	2,451,410	2,723,789	4,085,683	1,702,368
320	6,496	309	2,061	3,412,656	3,791,840	5,687,760	2,369,900
165	3,828	157	1,048	1,734,958	1,927,731	2,891,597	1,204,832
100	3,480	134	893	1,479,371	1,643,746	2,465,618	1,027,341
285	6,612	307	2,044	3,384,996	3,761,107	5,641,661	2,350,692
120	10,440	621	4,139	6,854,846	7,616,496	11,424,744	4,760,310
130	9,048	472	3,145	5,208,749	5,787,499	8,681,249	3,617,187
120	3,132	124	825	1,366,697	1,518,552	2,277,828	949,095
105	3,350	154	1,028	1,701,648	1,890,720	2,836,079	1,181,700
115	2,668	122	816	1,351,826	1,502,029	2,253,043	938,768
		3,516	**23,438**	**38,813,572**	**43,126,191**	**64,689,287**	**26,953,870**

[a] assume 4 columns of 30x30 cm every 24 m^2 = area of 0.36 m^2 of concrete in 24m^2 = 1.5% of the total area,
[b] Density: 1440 kg m^{-3}, [c] Density: 1600 kg m^{-3}, [d] Density: 2400 kg m^{-3}, [e] Density: 1000 kg m^{-3}

Table A19 Materials for building walls at Meridian One

External perimeter (m)	Total façade area (m^2)	Total wall area [a]	Brick (kg)	Insulation (rock wool) (kg)	Flashing membrane (kg)	Concrete bricks (kg)	Plastering (kg)			Glass (kg)	Aluminium frame (kg)
							Cement	Sand	Water		
120	1,044	714	158,426	14,855	571	143,443	4,410	15,266	3,928	5,277	8,905
145	1,262	856	189,860	17,803	685	171,904	5,285	18,295	4,707	6,490	10,951
95	1,378	1,031	228,640	21,439	825	207,017	6,365	22,032	5,669	5,548	9,363
170	1,972	1,469	325,931	30,562	1,175	295,107	9,073	31,407	8,081	8,043	13,572
280	3,248	2,331	516,985	48,476	1,864	468,091	14,391	49,817	12,818	14,678	24,770
210	2,436	1,760	390,456	36,612	1,408	353,529	10,869	37,624	9,681	10,813	18,247
325	5,655	3,739	829,394	77,770	2,991	750,954	23,088	79,920	20,564	30,657	51,733
135	3,132	2,268	503,125	47,177	1,814	455,542	14,006	48,481	12,475	13,822	23,325
140	4,872	3,392	752,358	70,547	2,713	681,204	20,944	72,497	18,654	23,685	39,969
320	6,496	4,435	983,841	92,253	3,548	890,794	27,387	94,803	24,394	32,973	55,641
165	3,828	2,780	616,744	57,831	2,224	558,415	17,168	59,429	15,292	16,763	28,287
100	3,480	2,587	573,785	53,803	2,069	519,519	15,973	55,290	14,227	14,293	24,120
285	6,612	4,568	1,013,277	95,013	3,654	917,447	28,207	97,639	25,124	32,705	55,190
120	10,440	6,301	1,397,629	131,052	5,040	1,265,448	38,906	134,675	34,653	66,230	111,764
130	9,048	5,903	1,309,347	122,774	4,722	1,185,516	36,449	126,169	32,464	50,326	84,925
120	3,132	2,307	511,683	47,979	1,845	463,291	14,244	49,306	12,687	13,205	22,283
105	3,350	2,322	515,063	48,296	1,858	466,351	14,338	49,631	12,771	16,441	27,744
115	2,668	1,852	410,748	38,515	1,481	371,902	11,434	39,580	10,184	13,061	22,041
			10,816,543	**1,014,243**	**39,009**	**9,793,571**	**301,103**	**10,42,281**	**268,189**	**361,949**	**610,789**

[a] assuming a 25% window-to-floor ratio

Table A20 Materials for floors at Meridian One

Floor Construction Materials					
Concrete & Screed (kg)				polythene membrane (kg)	Flooring (kg)
Cement	Sand	Gravel	Water		
1,155,773	2,577,541	3,664,388	695,185	1,055	1,780,947
1,421,390	3,169,905	4,506,529	854,951	1,298	2,190,240
1,215,243	2,710,167	3,852,938	730,956	1,110	1,872,585
1,761,598	3,928,619	5,585,162	1,059,583	1,609	2,714,472
3,214,948	7,169,801	10,193,022	1,933,758	2,936	4,953,960
2,368,282	5,281,612	7,508,659	1,424,497	2,163	3,649,320
6,714,594	14,974,520	21,288,682	4,038,759	6,131	10,346,616
3,027,392	6,751,523	9,598,373	1,820,945	2,764	4,664,952
5,187,652	11,569,217	16,447,497	3,120,319	4,737	7,993,728
7,221,832	16,105,733	22,896,883	4,343,857	6,595	11,128,226
3,671,503	8,187,984	11,640,533	2,208,371	3,353	5,657,472
3,130,632	6,981,763	9,925,696	1,883,043	2,859	4,824,036
7,163,299	15,975,197	22,711,305	4,308,651	6,541	11,038,032
14,506,165	32,350,853	45,991,926	8,725,309	13,246	22,352,760
11,022,709	24,582,241	34,947,598	6,630,046	10,065	16,985,052
2,892,192	6,450,007	9,169,719	1,739,623	2,641	4,456,620
3,601,012	8,030,778	11,417,040	2,165,971	3,288	5,548,851
2,860,722	6,379,825	9,069,945	1,720,695	2,612	4,408,128
79,276,213	**176,797,463**	**251,345,950**	**47,683,826**	**72,390**	**122,157,869**

Table A21 Materials for ceilings at Meridian One

Ceiling Construction Materials	
Plasterboard (kg)	**Insulation (rock wool) (kg)**
1,055	27,440
1,298	0
1,110	0
1,609	12,014,052
2,936	5,174
2,163	56,227
6,131	574,812
2,764	1,243,987
4,737	0
6,595	49,252,702
3,353	1,257
2,859	2,859
6,541	10,629
13,246	19,869
10,065	1,887
2,641	330,120
3,288	41,103
2,612	261,222
72,390	63,582,117
147,392	**127,425,457**

Table A22 Materials for conventional roofs at Meridian One

Conventional Roof Construction Materials						
Concrete & Roof topping (gravel) (kg)				Vapour Barrier (kg)	Insulation (Rock Wool) (kg)	Waterproof membrane (asphalt) (kg)
Cement	Sand	Gravel	Water			
370,174	813,931	1,379,769	221,878	620	9,147	32,981
455,247	1,000,987	1,696,864	272,870	763	11,249	40,560
233,533	513,487	870,459	139,977	391	5,770	20,807
423,158	930,429	1,577,255	253,636	709	10,456	37,701
772,270	1,698,050	2,878,519	462,890	1,294	19,082	68,805
568,891	1,250,863	2,120,453	340,986	953	14,057	50,685
1,075,286	2,364,313	4,007,963	644,514	1,801	26,569	95,802
363,608	799,494	1,355,295	217,943	609	8,984	32,396
415,379	913,326	1,548,263	248,974	696	10,264	37,008
991,299	2,179,645	3,694,916	594,173	1,660	24,494	88,319
440,970	969,595	1,643,649	264,312	739	10,896	39,288
250,672	551,172	934,342	150,250	420	6,194	22,334
860,356	1,891,732	3,206,848	515,688	1,441	21,258	76,653
464,608	1,021,569	1,731,755	278,481	778	11,480	41,394
441,298	970,317	1,644,873	264,509	739	10,904	39,317
308,773	678,924	1,150,906	185,075	517	7,629	27,510
314,548	691,621	1,172,430	188,537	527	7,772	28,025
343,590	755,478	1,280,681	205,944	576	8,490	30,612
8,750,072	**19,239,455**	**32,614,560**	**5,244,691**	**14,656**	**216,204**	**779,583**

Table A23 Materials for sedum roofs at Meridian One

Sedum-Roof Construction Materials												
Concrete (kg)				Protection layer (kg)	Root barrier (kg)	Drainage layer (kg)	Filter layer (kg)	Water retention layer (kg)	Substrate (kg)			Vegetation (kg)
Cement	Sand	Gravel	Water						Clay (10%)	Crushed brick (80%)	Compost (10%)	
370,174	813,931	172,089	221,878	132	352	572	528	264	4,397	35,179	4,397	6,464
455,247	1,000,987	211,638	272,870	162	433	703	649	324	5,408	43,264	5,408	7,950
233,533	513,487	108,566	139,977	83	222	361	333	166	2,774	22,194	2,774	4,078
423,158	930,429	196,720	253,636	151	402	653	603	302	5,027	40,214	5,027	7,389
772,270	1,698,050	359,017	462,890	275	734	1,193	1,101	550	9,174	73,392	9,174	13,486
568,891	125,0863	264,469	340,986	203	541	879	811	405	6,758	54,064	6,758	9,934
1,075,286	2,364,313	499,885	644,514	383	1,022	1,661	1,533	766	12,774	102,189	12,774	18,777
363,608	799,494	169,036	217,943	130	346	562	518	259	4,319	34,555	4,319	6,350
415,379	913,326	193,104	248,974	148	395	641	592	296	4,934	39,475	4,934	7,254
991,299	2,179,645	460,841	594,173	353	942	1,531	1,413	707	11,776	94,207	11,776	17,311
440,970	969,595	205,001	264,312	157	419	681	629	314	5,238	41,907	5,238	7,700
250,672	551,172	116,534	150,250	89	238	387	357	179	2,978	23,822	2,978	4,377
860,356	1,891,732	399,968	515,688	307	818	1,329	1,226	613	10,220	81,763	10,220	15,024
464,608	1,021,569	215,990	278,481	166	442	717	662	331	5,519	44,154	5,519	8,113
441,298	970,317	205,153	264,509	157	419	681	629	315	5,242	41,938	5,242	7,706
308,773	678,924	143,544	185,075	110	293	477	440	220	3,668	29,344	3,668	5,392
314,548	691,621	146,229	188,537	112	299	486	448	224	3,737	29,893	3,737	5,493
343,590	755,478	159,730	205,944	122	327	531	490	245	4,082	32,653	4,082	6,000
8,750,072	**19,239,455**	**4,067,784**	**5,244,691**	**3,118**	**8,316**	**13,513**	**12,473**	**6,482**	**103,944**	**831,555**	**103,944**	**158,798**

Table A25 Materials used throughout Meridian One's lifetime (metric kilotons)

Materials	Mass (metric kiloton, kt)											
	Construction of residential buildings					construction of concrete roofs	Production of vegetation	construction of sedum roofs	Maintenance [a,b,c] (use phase)		Overall total	
	Concrete frame	Wall	Floor	Ceiling	Total				Concrete roofs	Sedum roofs	Concrete roofs	Sedum roofs
Aluminium	-	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] concrete roof requires four replacements, sedum roof requires one replacement, [b] construction of sedum roof 60mm substrate, but production of vegetation is 10mm, [c] does not include utilities (gas, water, electricity) -- these are added in OpenLCA

Table A26 Transport of materials for Meridian One

Conventional roof scenario										
Material	**Delivery distance (km)**	**Recycled (%)**	**Disposal distance (recycle)**	**To landfill (%)**	**Disposal distance (landfill)**	**Transport for delivery (t.km)**	**Transport for maintenance (t.km)**	**Transport for recycle (t.km)**	**Transport for landfill (t.km)**	**Total transport (t.km)**
Aluminium	50	70%	21	30%	21	$3.05x10^{7}$	0	$8.98\ x10^{6}$	$3.85\ x10^{6}$	**$4.34\ x10^{7}$**
Asphalt	50	0%	21	100%	21	$3.90\ x10^{7}$	$1.56x10^{8}$	0	$8.19\ x10^{7}$	**$2.77\ x10^{8}$**
Brick	50	87%	21	13%	21	$5.41\ x10^{8}$	0	$1.98\ x10^{8}$	$2.95\ x10^{7}$	**$7.68\ x10^{8}$**
Brick (crushed)	60	0%	21	100%	21	0	0.	0	0	**0**
Cement	50	100%	21	0%	21	$6.36\ x10^{9}$	0	$2.67\ x10^{9}$	0	**$9.03\ x10^{9}$**
Clay	60	0%	21	100%	21	0	0	0	0	**0**
Compost	60	0%	21	100%	21	0	0	0	0	**0**
Concrete brick	50	87%	21	13%	21	$4.90\ x10^{8}$	0	$1.79\ x10^{8}$	$2.67\ x10^{7}$	**$6.95\ x10^{8}$**
Flashing membrane	50	0%	21	100%	21	$1.95\ x10^{6}$	0	0	$8.19\ x10^{5}$	**$2.77\ x10^{6}$**
Flooring (wood)	50	81%	21	19%	21	$6.11x10^{9}$	0	$2.08\ x10^{9}$	$4.87\ x10^{8}$	**$8.67\ x10^{9}$**
Glass	50	30%	21	70%	21	$1.81x10^{7}$	0	$2.28\ x10^{6}$	$5.32\ x10^{6}$	**$2.57\ x10^{7}$**
Gravel	50	100%	21	0%	21	$1.74\ x10^{10}$	$6.52\ x10^{9}$	$1.01\ x10^{10}$	0	**$3.40\ x10^{10}$**
Plasterboard	50	0%	21	100%	21	$3.62\ x10^{6}$	0	0	$1.52\ x10^{6}$	**$5.14\ x10^{6}$**
Polyethylene	50	81%	21	19%	21	$4.35x10^{6}$	$2.93\ x10^{6}$	$2.48\ x10^{6}$	$5.81\ x10^{5}$	**$1.03\ x10^{7}$**
Polypropylene	50	0%	21	100%	21	0	0	0	0	**0**
Polystyrene foam slab	50	0%	21	100%	21	0	0	0	0	**0**
Rock wool	50	0%	21	100%	21	$3.24x10^{9}$	0	0	$1.36\ x10^{9}$	**$4.60\ x10^{9}$**
Sand	50	100%	21	0%	21	$1.20\ x10^{10}$	0	$5.04\ x10^{9}$	0	**$1.71\ x10^{10}$**
Vegetation	60	0%	21	100%	21	0	0	0	0	**0**
Water	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	**n/a**
						$4.63x10^{10}$	**$6.68x10^{9}$**	**$2.02x10^{10}$**	**$2.00x10^{9}$**	**$7.52x10^{10}$**

Table A26 (continued)

Sedum roof scenario										
Material	**Delivery distance (km)**	**Recycled (%)**	**Disposal distance (recycle)**	**To landfill (%)**	**Disposal distance (landfill)**	**Transport for delivery (t.km)**	**Transport for maintenance (t.km)**	**Transport for recycle (t.km)**	**Transport for landfill (t.km)**	**Total transport (t.km)**
Aluminium	50	70%	21	30%	21	3.05×10^{7}	0	8.98×10^{6}	3.85×10^{6}	**4.34×10^{7}**
Asphalt	50	0%	21	100%	21	0	0	0	0	**0**
Brick	50	87%	21	13%	21	5.41×10^{8}	0	1.98×10^{8}	2.95×10^{7}	**7.68×10^{8}**
Brick (crushed)	60	0%	21	100%	21	3.18×10^{6}	3.18×10^{6}	0	2.23×10^{6}	**8.59×10^{6}**
Cement	50	100%	21	0%	21	6.36×10^{9}	0	2.67×10^{9}	0	**9.03×10^{9}**
Clay	60	0%	21	100%	21	3.18×10^{6}	3.18×10^{6}	0	2.23×10^{6}	**8.59×10^{6}**
Compost	60	0%	21	100%	21	3.86×10^{7}	2.54×10^{6}	0	1.44×10^{7}	**5.55×10^{7}**
Concrete brick	50	87%	21	13%	21	4.90×10^{8}	0	1.79×10^{8}	2.67×10^{7}	**6.95×10^{8}**
Flashing membrane	50	0%	21	100%	21	1.95×10^{6}	0	0	8.19×10^{5}	**2.77×10^{6}**
Flooring (wood)	50	81%	21	19%	21	6.11×10^{9}	0	2.08×10^{9}	4.87×10^{8}	**8.67×10^{9}**
Glass	50	30%	21	70%	21	1.81×10^{7}	0	2.28×10^{6}	5.32×10^{6}	**2.57×10^{7}**
Gravel	50	100%	21	0%	21	1.60×10^{10}	0	6.72×10^{9}	0	**2.27×10^{10}**
Plasterboard	50	0%	21	100%	21	3.62×10^{6}	0	0	1.52×10^{6}	**5.14×10^{6}**
Polyethylene	50	81%	21	19%	21	4.04×10^{6}	4.16×10^{5}	1.51×10^{6}	3.55×10^{5}	**6.32×10^{6}**
Polypropylene	50	0%	21	100%	21	7.80×10^{5}	7.80×10^{5}	0	6.55×10^{5}	**2.21×10^{6}**
Polystyrene foam slab	50	0%	21	100%	21	6.76×10^{5}	6.76×10^{5}	0	5.68×10^{5}	**1.92×10^{6}**
Rock wool	50	0%	21	100%	21	3.23×10^{9}	7.80×10^{4}	0	1.36×10^{9}	**4.59×10^{9}**
Sand	50	100%	21	0%	21	1.20×10^{10}	0	5.04×10^{9}	0	**1.71×10^{10}**
Vegetation	60	0%	21	100%	21	9.53×10^{6}	9.53×10^{6}	0	6.67×10^{6}	**2.57×10^{7}**
Water	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	**n/a**
						4.49×10^{10}	**2.04×10^{7}**	**1.69×10^{10}**	**1.94×10^{9}**	**6.37×10^{10}**

Table A27 Assumed diets for species comprising the population unit. If species-specific data is unavailable, it is assumed that omnivores consume 75% meat and 25% vegetation.

Species	Biological class	Diet type [a]	Proportion from plants	Proportion from invertebrates	Proportion from vertebrates	Proportion from waste
Pipistrellus pipistrellus	Mammal	Carnivore	0	1	0	0
Pipistrellus pygmaeus	Mammal	Carnivore	0	1	0	0
Pipistrellus nathusii	Mammal	Carnivore	0	1	0	0
Nyctalus noctula	Mammal	Carnivore	0	1	0	0
Nyctalus leisleri	Mammal	Carnivore	0	1	0	0
Myotis daubentonii	Mammal	Carnivore	0	1	0	0
Plecotus auritus	Mammal	Carnivore	0	1	0	0
Erinaceus europaeus	Mammal	Omnivore	0.25	0.75	0	0
Passer domesticus	Bird	Omnivore	0.25	0.75	0	0
Turdus philomelos	Bird	Carnivore	0	1	0	0
Sturnus vulgaris	Bird	Carnivore	0	1	0	0
Falco peregrinus	Bird	Carnivore	0	0	1	0
Apus apus	Bird	Carnivore	0	1	0	0
Natrix natrix	Reptile	Carnivore	0	1	0	0
Anguis fragilis	Reptile	Carnivore	0	1	0	0
Zootoca vivipara	Reptile	Carnivore	0	1	0	0
Aglais io	Invertebrate	Herbivore	1	0	0	0
Agrotis exclamationis	Invertebrate	Herbivore	1	0	0	0
Amara aenea	Invertebrate	Herbivore	1	0	0	0
Anthocharis cardamines	Invertebrate	Herbivore	1	0	0	0
Anthophora plumipes	Invertebrate	Herbivore	1	0	0	0
Aphidoidea	Invertebrate	Herbivore	1	0	0	0
Apis mellifera	Invertebrate	Herbivore	1	0	0	0
Araneus diadematus	Invertebrate	Carnivore	0	1	0	0
Bibio marci	Invertebrate	Herbivore	1	0	0	0
Bombus lapidaries	Invertebrate	Herbivore	1	0	0	0
Bombus lucorum	Invertebrate	Herbivore	1	0	0	0
Bombus pratorum	Invertebrate	Herbivore	1	0	0	0
Bombus terrestris	Invertebrate	Herbivore	1	0	0	0

Table A27 (continued)

Species	Biological class	Diet type [a]	Proportion from plants	Proportion from invertebrates	Proportion from vertebrates	Proportion from waste
Chrysoperla carnea	Invertebrate	Herbivore	1	0	0	0
Coccinella septempunctata	Invertebrate	Carnivore	0	1	0	0
Cupido minimus	Invertebrate	Herbivore	1	0	0	0
Deilephila elpenor	Invertebrate	Herbivore	1	0	0	0
Diplolepis rosae	Invertebrate	Herbivore	1	0	0	0
Gonepteryx rhamni	Invertebrate	Herbivore	1	0	0	0
Heriades truncorum	Invertebrate	Herbivore	1	0	0	0
Ichneumonidae	Invertebrate	Herbivore	1	0	0	0
Lycaena phlaeas	Invertebrate	Herbivore	1	0	0	0
Megachilidae?	Invertebrate	Herbivore	1	0	0	0
Mimas tiliae	Invertebrate	Herbivore	1	0	0	0
Neuropterus quercusbaccarum	Invertebrate	Herbivore	1	0	0	0
Noctua comes	Invertebrate	Herbivore	1	0	0	0
Osmia rufa (most common)	Invertebrate	Herbivore	1	0	0	0
Pararge aegeria	Invertebrate	Herbivore	1	0	0	0
Polygonia c-album	Invertebrate	Herbivore	1	0	0	0
Pyrochroa serraticornis	Invertebrate	Herbivore	1	0	0	0
Pyrrhosoma nymphula	Invertebrate	Carnivore	0	1	0	0
Salticus scenicus	Invertebrate	Carnivore	0	1	0	0
Segestria florentina	Invertebrate	Carnivore	0	1	0	0
Tetragnatha extensa	Invertebrate	Carnivore	0	1	0	0
Tytthaspis sedecimpunctata	Invertebrate	Carnivore	0	1	0	0
Vanessa cardui	Invertebrate	Herbivore	1	0	0	0
Insecta[*]	Invertebrate	Herbivore [c]	1	0	0	0
Rattus norvegicus	Mammal	Omnivore	0.33	0.33	0	0.33
Vulpes vulpes	Mammal	Omnivore	0.02	0.05	0.37	0.56
Aegithalos caudatus	Bird	Carnivore	0	1	0	0
Carduelis carduelis	Bird	Herbivore	1	0	0	0
Columba palumbus	Bird	Herbivore	1	0	0	0
Corvus corone	Bird	Omnivore	0.25	0.5	0	0.25
Cyanistes caeruleus	Bird	Omnivore	0.2	0.8	0	0

Table A27 (continued)

Species	Biological class	Diet type [a]	Proportion from plants	Proportion from invertebrates	Proportion from vertebrates	Proportion from waste
Garrulus glandarius	Bird	Omnivore	0.25	0.75	0	0
Motacilla cinerea	Bird	Carnivore	0	1	0	0
Parus major	Bird	Carnivore	0	1	0	0
Periparus ater	Bird	Omnivore	0.25	0.75	0	0
Pica pica	Bird	Omnivore	0.6	0.4	0	0
Prunella modularis	Bird	Carnivore	0	1	0	0
Psittacula krameri	Bird	Herbivore	1	0	0	0
Troglodytes troglodytes	Bird	Omnivore	0.25	0.75	0	0
Turdus merula	Bird	Omnivore	0.25	0.75	0	0

* Invertebrates (aggregated)

Table A28 The aggregated diet of bird species at Meridian One

Species	Biological class	Diet type [a]	Proportion from plants	Proportion from invertebrates [c]	Proportion from vertebrates	Proportion from waste	Assumed abundance	Proportion from plants	Proportion from invertebrates [c]	Proportion from vertebrates	Proportion from waste
Passer domesticus	Bird	Omnivore	0.25	0.75	0	0	56	14	42	0	0
Turdus philomelos	Bird	Carnivore	0	1	0	0	29	0	29	0	0
Sturnus vulgaris	Bird	Carnivore	0	1	0	0	41	0	41	0	0
Apus apus	Bird	Carnivore	0	1	0	0	27	0	27	0	0
Aegithalos caudatus	Bird	Carnivore	0	1	0	0	29	0	29	0	0
Carduelis carduelis	Bird	Herbivore	1	0	0	0	31	31	0	0	0
Columba palumbus	Bird	Herbivore	1	0	0	0	26	26	0	0	0
Corvus corone	Bird	Omnivore	0.25	0.5	0	0.25	11	2.75	5.5	0	2.75
Cyanistes caeruleus	Bird	Omnivore	0.2	0.8	0	0	34	6.8	27.2	0	0
Erithacus rubecula	Bird	Omnivore	0.2	0.8	0	0	35	7	28	0	0
Garrulus glandarius	Bird	Omnivore	0.25	0.75	0	0	1	0.25	0.75	0	0
Motacilla cinerea	Bird	Carnivore	0	1	0	0	27	0	27	0	0
Parus major	Bird	Carnivore	0	1	0	0	32	0	32	0	0
Periparus ater	Bird	Omnivore	0.25	0.75	0	0	28	7	21	0	0
Pica pica	Bird	Omnivore	0.6	0.4	0	0	7	4.2	2.8	0	0
Prunella modularis	Bird	Carnivore	0	1	0	0	29	0	29	0	0
Psittacula krameri	Bird	Herbivore	1	0	0	0	6	6	0	0	0
Troglodytes troglodytes	Bird	Omnivore	0.25	0.75	0	0	33	8.25	24.75	0	0
Turdus merula	Bird	Omnivore	0.25	0.75	0	0	7	1.75	5.25	0	0

Table A29, ‘Terrestrial species expected to use Meridian One’, and **Table A30**, ‘Terrestrial species of birds, mammals, reptiles, and invertebrates expected at the Meridian Water Development (MWD) site in late spring (April-May), and their anticipated requirement for land area, water, and energy’, is too large to produce here in physical form. The reader is instead directed to these data in the form of an online repository:

- Mason, A. R. (2023). Appendix 5: Supplementary data for Chapter 4 in the thesis: 'Quantifying the natural resource requirements of terrestrial ecosystems for managing biodiversity alongside human development' [Data set]. Zenodo. https://doi.org/10.5281/zenodo.11217971

Table A31 Quantifying the energy requirements of constituent units (based on Table A3, Appendix 1). Here, contributions of plant biomass and nectar to invertebrate (aggregated) diets described in Appendix 5: 9% plant biomass, 69% nectar

Biological class	Total energy (kJ day^{-1})	From plant biomass (kJ day^{-1})			Nectar (kJ day^{-1})	From invertebrates (kJ day^{-1})	From meat (kJ day^{-1})	From waste (kJ day^{-1})
		Total	Species that can access the sedum roof	Other species				
Sum mammals	11,789	488	0	488	0	6,217	166	331
Sum birds	60,738	46,461	46,461	0	0	25,150	638	0
Sum reptiles	4	0	0	0	0	28	0	0
Sum invertebrates	178,081	235,429	21,189	0	162,446	54,611	0	0
	14,439	**282,378**	**67,650**	**488**	**162,446**	**86,007**	**805**	**331**

Table A32 Observed abundances of invertebrates by genus on sedum roofs [596]

Genus	Minimum count (median if zero)	Abundance (individuals km^{-2})	Scaled abundance (individuals km^{-2})	Number of species in each genus expected on-site	Abundance per species (individuals km^{-2})
Hymenoptera	7.0	8,974	628	14	45
Hemiptera	4.0	5,128	359	1	359
Coleoptera	1.5	1,923	135	4	34
Araneae	3.0	3,846	269	4	68
Lepidoptera	1.0	1,282	90	12	8
		21,154	**1,481**	**35**	**512**

[a] scaled to sedum roof area (0.0108 km2), [b] ([scaled abundance] / [number of species in genus…]) rounded up to next whole number; this avoids the presence of a partial individual.

Table A33 Calorific values of different invertebrates

Species	Calorific value (kJ kg^{-1})	Reference
Aulacophora flareicollis Lucas	29,793	J. S. Singh and P. S. Yadava (1973)
Cantharis tenuicollis Pall	31,527	J. S. Singh and P. S. Yadava (1973)
Chiloloba accuta Wied	31,147	J. S. Singh and P. S. Yadava (1973)
Diapromorpha turciea Thunb…	24,321	J. S. Singh and P. S. Yadava (1973)
Mylabris pustulata Thunb…	25,263	J. S. Singh and P. S. Yadava (1973)
Onthophagus catta F.	30,401	J. S. Singh and P. S. Yadava (1973)
Pachystrachelus cribioceps Chd.	29,556	J. S. Singh and P. S. Yadava (1973)
Pheroperophys consularis Schm. Goeb.	26,138	J. S. Singh and P. S. Yadava (1973)
Scarabidae sp.	26,601	J. S. Singh and P. S. Yadava (1973)
Mantis sp.	21,793	J. S. Singh and P. S. Yadava (1973)
Apis sp.	25,617	J. S. Singh and P. S. Yadava (1973)
Pieris sp. (1)	30,401	J. S. Singh and P. S. Yadava (1973)
Pieris sp. (2)	25,248	J. S. Singh and P. S. Yadava (1973)
Crocothmis sp.	25,615	J. S. Singh and P. S. Yadava (1973)
Gryllus sp.	31,361	J. S. Singh and P. S. Yadava (1973)
Truxalis indica Thunb.	24,256	J. S. Singh and P. S. Yadava (1973)
Chortoicetes terminifera	20,878	Lenka Kouřimská and Anna Adámkov (2016)
Oecophylla smaragdina	53,220	Lenka Kouřimská and Anna Adámkov (2016)
Tenebrio molitor	8,619	Lenka Kouřimská and Anna Adámkov (2016)
Tenebrio molitor	5,774	Lenka Kouřimská and Anna Adámkov (2016)
Atta mexicna	16,903	Lenka Kouřimská and Anna Adámkov (2016)
Gryllus bimaculatus	5,021	Lenka Kouřimská and Anna Adámkov (2016)
Oxya japonica	6,234	Lenka Kouřimská and Anna Adámkov (2016)
Cyrtacanthacris tatarica	3,724	Lenka Kouřimská and Anna Adámkov (2016)
Bombyx mori	3,933	Lenka Kouřimská and Anna Adámkov (2016)
Locusta migratoria	7,489	Lenka Kouřimská and Anna Adámkov (2016)
Macrotermes falciger Gerstacker	31,840	Phelps et al. (1975)
Macrotermes subhyalinus Rambur	25,648	Oliveira et al. (1976)
Anaphe venata Butler	25,564	Ashiru (1988)
Various (23), mostly *Saturniidae*	19,121	Malaisse and Parent (1980)
Various (16)	6,594	Ramos-Elorduy and Pino (1990)
Various (17)	5,964	Ramos-Elorduy and Pino (1990)
Various (14)	5,646	Ramos-Elorduy and Pino (1990)
Various (24)	5,361	Ramos-Elorduy and Pino (1990)
Various (20)	4,168	Ramos-Elorduy and Pino (1990)
Total	**20,021**	

Table A34 Calorific value of commonly consumed meats

Meat type	Calorific value (kJ kg^{-1})	Reference
Beef (muscle)	4,812	R. Chizzolini et al. (1999)
Veal (muscle)	4,226	R. Chizzolini et al. (1999)
Pork (muscle)	4,770	R. Chizzolini et al. (1999)
Mutton (fillet)	5,104	R. Chizzolini et al. (1999)
Chicken	6,025	R. Chizzolini et al. (1999)
Turkey	9,665	R. Chizzolini et al. (1999)
Total	**5,767**	

Table A35 The distribution of energy through the species expected to use Meridian One. Nectar production (limiting energy provision) supports 950 invertebrates. It is assumed that consumers have equal access to invertebrates as prey once the energy requirements of insectivorous invertebrates are satisfied

Trophic level	Comment	Predator	Prey	Invertebrates consumed directly (invertebrates day^{-1} individual^{-1})	Invertebrates distributed [a]	Number of consumers supported by invertebrates [b]	Abundance limit [c]	Total supported by vegetation stocks [d] (individuals)
4	Plants	Plants	Not applicable	0	0	0		0
3a	Direct consumption of sedum roof vegetation by 950 invertebrates.	Invertebrates (27 species)	Vegetation	0	0	950		950
3b	Direct consumption of sedum roof vegetation by vertebrates	Ring-necked parakeet	Vegetation	0	0	0	6	6
3b	Direct consumption of sedum roof vegetation by vertebrates	European goldfinch	Vegetation	0	0	0	29	29
3b	Direct consumption of sedum roof vegetation by vertebrates	Wood pigeon	Vegetation	0	0	0	24	24
2a	Consumption of invertebrates by invertebrates	Invertebrates (8 species)	Invertebrates	1	218	218		218
2b	Consumption of invertebrates by vertebrates	Blue tit	Invertebrates	11	27	2		2
2b		Eurasian magpie	Invertebrates	27	27	1		1
2b		Coal tit	Invertebrates	10	27	2		2
2b		Eurasian wren	Invertebrates	9	27	3		3
2b		European Robin	Invertebrates	16	27	1		1
2b		Common blackbird	Invertebrates	41	27	0		0
2b		Eurasian jay	Invertebrates	46	27	0		0

Table A35 (continued)

Trophic level	Comment	Predator	Prey	Invertebrates consumed directly (invertebrates day^{-1} individual^{-1})	Invertebrates distributed [a]	Number of consumers supported by invertebrates [b]	Abundance limit [c]	Total supported by vegetation stocks [d] (individuals)
2b		Great tit	Invertebrates	19	27	1		1
2b		Common swift	Invertebrates	41	27	0		0
2b		Long-tailed tit	Invertebrates	12	27	2		2
2b		Song thrush	Invertebrates	43	27	0		0
2b		Dunnock	Invertebrates	20	27	1		1
2b		European starling	Invertebrates	47	27	0		0
2b		Brown long-eared bat	Invertebrates	5	27	5		5
2b		Nathuslus' pipistrelle	Invertebrates	4	27	6		6
2b		Noctule bat	Invertebrates	13	27	2		2
2b		Daubenton's bat	Invertebrates	5	27	5		5
2b		Common pipistrelle	Invertebrates	3	27	9		9
2b		Soprano pipistrelle	Invertebrates	3	27	9		9
2b		Leisler's bat	Invertebrates	6	27	4		4
2b		Grey wagtail	Invertebrates	23	27	1		1
2b		European hedgehog	Invertebrates	58	27	0		0
2b		Carrion crow	Invertebrates	40	27	0		0
2b		Common lizard	Invertebrates	1	27	27		27
2b		Brown rat	Invertebrates	20	27	1		1
1	Consumption of other species	Red fox	Invertebrates (29), brown rat (1), wood pigeon (1)	6	27	4	1	1
1		Peregrine falcon	Wood pigeon (1), European starling (2), house sparrow (5)			0	1	1
				Total	**947**	**1,255**		**1,312**

[a] based on a population of 950 invertebrates, [b] rounded down, [c] based on observations in the literature, [d] plant biomass and nectar

Table A36 life cycle impacts of the concrete roofs design scenario at Meridian One

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$5.12x10^{10}$	$PDF.m^2.yr$
Climate change, ecosystem quality, long term	$6.54\ x10^{9}$	$PDF.m^2.yr$
Climate change, ecosystem quality, short term	$1.92\ x10^{9}$	$PDF.m^2.yr$
Marine acidification, long term	$1.58\ x10^{9}$	$PDF.m^2.yr$
Land transformation, biodiversity	$1.02\ x10^{9}$	$PDF.m^2.yr$
Terrestrial acidification	$7.99\ x10^{8}$	$PDF.m^2.yr$
Land occupation, biodiversity	$3.98\ x10^{8}$	$PDF.m^2.yr$
Freshwater ecotoxicity, short term	$1.80\ x10^{8}$	$PDF.m^2.yr$
Marine acidification, short term	$1.71\ x10^{8}$	$PDF.m^2.yr$
Freshwater acidification	$1.11\ x10^{8}$	$PDF.m^2.yr$
Marine eutrophication	$2.04\ x10^{7}$	$PDF.m^2.yr$
Freshwater eutrophication	$3.35\ x10^{6}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$1.33\ x10^{5}$	$PDF.m^2.yr$
Climate change, human health, long term	$2.98\ x10^{4}$	DALY
Water availability, freshwater ecosystem	$9.42x10^{3}$	$PDF.m^2.yr$
Climate change, human health, short term	$8.89x10^{3}$	DALY
Particulate matter formation	$7.88x10^{3}$	DALY
Thermally polluted water	$5.33\ x10^{3}$	$PDF.m^2.yr$
Water availability, human health	$3.99\ x10^{3}$	DALY
Human toxicity cancer, short term	$1.94\ x10^{3}$	DALY
Human toxicity non-cancer, short term	$1.41\ x10^{3}$	DALY
Human toxicity non-cancer, long term	$1.20\ x10^{3}$	DALY
Human toxicity cancer, long term	34.53	DALY
Ionizing radiation, human health	13.40	DALY
Ozone layer depletion	3.79	DALY
Photochemical oxidant formation	3.54	DALY
Ionizing radiation, ecosystem quality	2.04	$PDF.m^2.yr$
Total ecological impacts	**$6.39x10^{10}$**	**$PDF.m^2.yr$**

Table A37 Life cycle impacts of the sedum roofs design scenario at Meridian One

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$4.37x10^{10}$	$PDF.m^2.yr$
Climate change, ecosystem quality, long term	$5.59x10^{9}$	$PDF.m^2.yr$
Climate change, ecosystem quality, short term	$1.64x10^{9}$	$PDF.m^2.yr$
Marine acidification, long term	$1.35x10^{9}$	$PDF.m^2.yr$
Land transformation, biodiversity	$8.73x10^{8}$	$PDF.m^2.yr$
Terrestrial acidification	$6.82x10^{8}$	$PDF.m^2.yr$
Land occupation, biodiversity	$3.39x10^{8}$	$PDF.m^2.yr$
Freshwater ecotoxicity, short term	$1.54x10^{8}$	$PDF.m^2.yr$
Marine acidification, short term	$1.46x10^{8}$	$PDF.m^2.yr$
Freshwater acidification	$9.50x10^{7}$	$PDF.m^2.yr$
Marine eutrophication	$1.74x10^{7}$	$PDF.m^2.yr$
Freshwater eutrophication	$2.86x10^{6}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$1.15x10^{5}$	$PDF.m^2.yr$
Climate change, human health, long term	$2.54x10^{4}$	DALY
Water availability, freshwater ecosystem	$8.08x10^{3}$	$PDF.m^2.yr$
Climate change, human health, short term	$7.60x10^{3}$	DALY
Particulate matter formation	$6.72x10^{3}$	DALY
Thermally polluted water	$4.56x10^{3}$	$PDF.m^2.yr$
Water availability, human health	$3.42x10^{3}$	DALY
Human toxicity cancer, short term	$1.66x10^{3}$	DALY
Human toxicity non-cancer, short term	$1.21x10^{3}$	DALY
Human toxicity non-cancer, long term	1.03x103	DALY
Human toxicity cancer, long term	29.55	DALY
Ionizing radiation, human health	11.45	DALY
Ozone layer depletion	3.23	DALY
Photochemical oxidant formation	3.02	DALY
Ionizing radiation, ecosystem quality	1.74	$PDF.m^2.yr$
Total ecological impacts	**$5.46x10^{10}$**	**$PDF.m^2.yr$**

Table A38 Impacts of the construction phase, concrete roofs scenario

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$3.18x10^{10}$	$PDF.m^2.yr$
Climate change, ecosystem quality, long term	$4.08x10^{9}$	$PDF.m^2.yr$
Climate change, ecosystem quality, short term	$1.20x10^{9}$	$PDF.m^2.yr$
Marine acidification, long term	$9.83x10^{8}$	$PDF.m^2.yr$
Land transformation, biodiversity	$6.35x10^{8}$	$PDF.m^2.yr$
Terrestrial acidification	$4.97x10^{8}$	$PDF.m^2.yr$
Land occupation, biodiversity	$2.44x10^{8}$	$PDF.m^2.yr$
Freshwater ecotoxicity, short term	$1.12x10^{8}$	$PDF.m^2.yr$
Marine acidification, short term	$1.07x10^{8}$	$PDF.m^2.yr$
Freshwater acidification	$6.92x10^{7}$	$PDF.m^2.yr$
Marine eutrophication	$1.27x10^{7}$	$PDF.m^2.yr$
Freshwater eutrophication	$2.08x10^{6}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$8.63x10^{4}$	$PDF.m^2.yr$
Climate change, human health, long term	$1.86x10^{4}$	DALY
Water availability, freshwater ecosystem	$5.92x10^{3}$	$PDF.m^2.yr$
Climate change, human health, short term	$5.54x10^{3}$	DALY
Particulate matter formation	$4.89x10^{3}$	DALY
Thermally polluted water	$3.34x10^{3}$	$PDF.m^2.yr$
Water availability, human health	$2.50x10^{3}$	DALY
Human toxicity cancer, short term	$1.20x10^{3}$	DALY
Human toxicity non-cancer, short term	875.86	DALY
Human toxicity non-cancer, long term	749.01	DALY
Human toxicity cancer, long term	21.61	DALY
Ionizing radiation, human health	8.30	DALY
Ozone layer depletion	2.35	DALY
Photochemical oxidant formation	2.19	DALY
Ionizing radiation, ecosystem quality	1.26	$PDF.m^2.yr$
Total ecological impacts	**3.98E+10**	**$PDF.m^2.yr$**

Table A39 Impacts of the use phase, concrete roofs scenario

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$4.49x10^9$	$PDF.m^2.yr$
Climate change, ecosystem quality, long term	$5.59x10^8$	$PDF.m^2.yr$
Climate change, ecosystem quality, short term	$1.67x10^8$	$PDF.m^2.yr$
Marine acidification, long term	$1.35x10^8$	$PDF.m^2.yr$
Land transformation, biodiversity	$8.88x10^7$	$PDF.m^2.yr$
Terrestrial acidification	$6.90x10^7$	$PDF.m^2.yr$
Land occupation, biodiversity	$3.69x10^7$	$PDF.m^2.yr$
Freshwater ecotoxicity, short term	$1.54x10^7$	$PDF.m^2.yr$
Marine acidification, short term	$1.46x10^7$	$PDF.m^2.yr$
Freshwater acidification	$9.60x10^6$	$PDF.m^2.yr$
Marine eutrophication	$1.77x10^6$	$PDF.m^2.yr$
Freshwater eutrophication	$2.88x10^5$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$1.09x10^4$	$PDF.m^2.yr$
Climate change, human health, long term	$2.55x10^3$	DALY
Water availability, freshwater ecosystem	849.74	$PDF.m^2.yr$
Climate change, human health, short term	774.15	DALY
Particulate matter formation	680.45	DALY
Thermally polluted water	465.04	$PDF.m^2.yr$
Water availability, human health	359.54	DALY
Human toxicity cancer, short term	171.03	DALY
Human toxicity non-cancer, short term	122.64	DALY
Human toxicity non-cancer, long term	103.77	DALY
Human toxicity cancer, long term	2.98	DALY
Ionizing radiation, human health	1.21	DALY
Ozone layer depletion	0.33	DALY
Ionizing radiation, ecosystem quality	0.18	$PDF.m^2.yr$
Total ecological impacts	**$5.58x10^9$**	**$PDF.m^2.yr$**

Table A40 Impacts of the end-of-life phase, concrete roofs scenario

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$1.48x10^{10}$	$PDF.m^2.yr$
Climate change, ecosystem quality, long term	$1.89x10^{9}$	$PDF.m^2.yr$
Climate change, ecosystem quality, short term	$5.55x10^{8}$	$PDF.m^2.yr$
Marine acidification, long term	$4.56x10^{8}$	$PDF.m^2.yr$
Land transformation, biodiversity	$2.99x10^{8}$	$PDF.m^2.yr$
Terrestrial acidification	$2.32x10^{8}$	$PDF.m^2.yr$
Land occupation, biodiversity	$1.16x10^{8}$	$PDF.m^2.yr$
Freshwater ecotoxicity, short term	$5.22x10^{7}$	$PDF.m^2.yr$
Marine acidification, short term	$4.94x10^{7}$	$PDF.m^2.yr$
Freshwater acidification	$3.23x10^{7}$	$PDF.m^2.yr$
Marine eutrophication	$5.97x10^{6}$	$PDF.m^2.yr$
Freshwater eutrophication	$9.83x10^{5}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$3.59x10^{4}$	$PDF.m^2.yr$
Climate change, human health, long term	$8.60x10^{3}$	DALY
Water availability, freshwater ecosystem	$2.63x10^{3}$	$PDF.m^2.yr$
Climate change, human health, short term	$2.56x10^{3}$	DALY
Particulate matter formation	$2.30x10^{3}$	DALY
Thermally polluted water	$1.52x10^{3}$	$PDF.m^2.yr$
Water availability, human health	$1.13x10^{3}$	DALY
Human toxicity cancer, short term	564.58	DALY
Human toxicity non-cancer, short term	411.13	DALY
Human toxicity non-cancer, long term	345.54	DALY
Human toxicity cancer, long term	9.89	DALY
Ionizing radiation, human health	3.88	DALY
Ozone layer depletion	1.11	DALY
Ionizing radiation, ecosystem quality	0.60	$PDF.m^2.yr$
Total ecological impacts	**$1.85x10^{10}$**	**$PDF.m^2.yr$**

Table A41 Impacts of the construction phase, sedum roofs scenario

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$3.11x10^{10}$	$PDF.m^2.yr$
Climate change, ecosystem quality, long term	$3.97x10^{9}$	$PDF.m^2.yr$
Climate change, ecosystem quality, short term	$1.17x10^{9}$	$PDF.m^2.yr$
Marine acidification, long term	$9.57x10^{8}$	$PDF.m^2.yr$
Land transformation, biodiversity	$6.17x10^{8}$	$PDF.m^2.yr$
Terrestrial acidification	$4.84x10^{8}$	$PDF.m^2.yr$
Land occupation, biodiversity	$2.37x10^{8}$	$PDF.m^2.yr$
Freshwater ecotoxicity, short term	$1.09x10^{8}$	$PDF.m^2.yr$
Marine acidification, short term	$1.04x10^{8}$	$PDF.m^2.yr$
Freshwater acidification	$6.74x10^{7}$	$PDF.m^2.yr$
Marine eutrophication	$1.23x10^{7}$	$PDF.m^2.yr$
Freshwater eutrophication	$2.02x10^{6}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$8.43x10^{4}$	$PDF.m^2.yr$
Climate change, human health, long term	$1.81x10^{4}$	DALY
Water availability, freshwater ecosystem	$5.82x10^{3}$	$PDF.m^2.yr$
Climate change, human health, short term	$5.41x10^{3}$	DALY
Particulate matter formation	$4.76x10^{3}$	DALY
Thermally polluted water	$3.26x10^{3}$	$PDF.m^2.yr$
Water availability, human health	$2.45x10^{3}$	DALY
Human toxicity cancer, short term	1173.95	DALY
Human toxicity non-cancer, short term	853.14	DALY
Human toxicity non-cancer, long term	730.48	DALY
Human toxicity cancer, long term	21.08	DALY
Ionizing radiation, human health	8.13	DALY
Ozone layer depletion	2.28	DALY
Photochemical oxidant formation	2.13	DALY
Ionizing radiation, ecosystem quality	1.23	$PDF.m^2.yr$
Total ecological impacts	**$3.88x10^{10}$**	**$PDF.m^2.yr$**

Table A42 Impacts of the use phase, sedum roofs scenario

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$1.61x10^{8}$	$PDF.m^2.yr$
Climate change, ecosystem quality, long term	$8.82x10^{6}$	$PDF.m^2.yr$
Climate change, ecosystem quality, short term	$5.79x10^{6}$	$PDF.m^2.yr$
Land occupation, biodiversity	$3.10x10^{6}$	$PDF.m^2.yr$
Marine acidification, long term	$2.02x10^{6}$	$PDF.m^2.yr$
Terrestrial acidification	$1.34x10^{6}$	$PDF.m^2.yr$
Land transformation, biodiversity	$8.11x10^{5}$	$PDF.m^2.yr$
Marine acidification, short term	$2.19x10^{5}$	$PDF.m^2.yr$
Freshwater acidification	$1.88x10^{5}$	$PDF.m^2.yr$
Freshwater ecotoxicity, short term	$1.71x10^{5}$	$PDF.m^2.yr$
Marine eutrophication	$3.08x10^{4}$	$PDF.m^2.yr$
Freshwater eutrophication	$2.13x10^{3}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	278.00	$PDF.m^2.yr$
Water availability, freshwater ecosystem	67.60	$PDF.m^2.yr$
Climate change, human health, long term	39.96	DALY
Climate change, human health, short term	26.79	DALY
Water availability, human health	25.13	DALY
Thermally polluted water	19.52	$PDF.m^2.yr$
Particulate matter formation	10.79	DALY
Human toxicity cancer, short term	6.23	DALY
Human toxicity non-cancer, long term	3.00	DALY
Human toxicity non-cancer, short term	2.87	DALY
Human toxicity cancer, long term	0.09	DALY
Ionizing radiation, human health	0.08	DALY
Ionizing radiation, ecosystem quality	0.01	$PDF.m^2.yr$
Ozone layer depletion	0.01	DALY
Photochemical oxidant formation	0.01	DALY
Total ecological impacts	**$1.84x10^{8}$**	**$PDF.m^2.yr$**

Table A43 Impacts of the end-of-life phase, sedum roofs scenario

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$1.26x10^{10}$	$PDF.m^2.yr$
Climate change, ecosystem quality, long term	$1.61x10^{9}$	$PDF.m^2.yr$
Climate change, ecosystem quality, short term	$4.73x10^{8}$	$PDF.m^2.yr$
Marine acidification, long term	$3.88x10^{8}$	$PDF.m^2.yr$
Land transformation, biodiversity	$2.55x10^{8}$	$PDF.m^2.yr$
Terrestrial acidification	$1.98x10^{8}$	$PDF.m^2.yr$
Land occupation, biodiversity	$9.90x10^{7}$	$PDF.m^2.yr$
Freshwater ecotoxicity, short term	$4.44x10^{7}$	$PDF.m^2.yr$
Marine acidification, short term	$4.21x10^{7}$	$PDF.m^2.yr$
Freshwater acidification	$2.75x10^{7}$	$PDF.m^2.yr$
Marine eutrophication	$5.08x10^{6}$	$PDF.m^2.yr$
Freshwater eutrophication	$8.37x10^{5}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$3.06x10^{4}$	$PDF.m^2.yr$
Climate change, human health, long term	$7.33x10^{3}$	DALY
Water availability, freshwater ecosystem	$2.24x10^{3}$	$PDF.m^2.yr$
Climate change, human health, short term	$2.18x10^{3}$	DALY
Particulate matter formation	$1.96x10^{3}$	DALY
Thermally polluted water	$1.29x10^{3}$	$PDF.m^2.yr$
Water availability, human health	961.80	DALY
Human toxicity cancer, short term	480.76	DALY
Human toxicity non-cancer, short term	350.09	DALY
Human toxicity non-cancer, long term	294.24	DALY
Human toxicity cancer, long term	8.43	DALY
Ionizing radiation, human health	3.30	DALY
Ozone layer depletion	0.95	DALY
Photochemical oxidant formation	0.88	DALY
Ionizing radiation, ecosystem quality	0.51	$PDF.m^2.yr$
Total ecological impacts	**$1.58x10^{10}$**	**$PDF.m^2.yr$**

Table A44 Life cycle contributions to freshwater ecotoxicity (long term) in the concrete roofs scenario

Contribution	Process	Amount	Unit
97.35%	Transport	5.02×10^{10}	PDF.m^2.yr
1.55%	Rock wool	5.76×10^{8}	PDF.m^2.yr
0.40%	Electricity	1.48×10^{8}	PDF.m^2.yr
0.36%	market for cement, Portland | cement, Portland | Cutoff, S - Europe without Switzerland	1.32×10^{8}	PDF.m^2.yr
0.13%	window frame production, aluminium, U=1.6 W/m2K | window frame, aluminium, U=1.6 W/m2K | Cutoff, S - RoW	4.69×10^{7}	PDF.m^2.yr
0.08%	gravel production, crushed | gravel, crushed | Cutoff, S - RoW	2.95×10^{7}	PDF.m^2.yr
0.04%	market for sealing tape, aluminium/PE, 50 mm wide | sealing tape, aluminium/PE, 50 mm wide | Cutoff, S - GLO	1.35×10^{7}	PDF.m^2.yr
0.03%	autoclaved aerated concrete block production | autoclaved aerated concrete block | Cutoff, S - RoW	1.05×10^{7}	PDF.m^2.yr
0.02%	clay brick production | clay brick | Cutoff, S - RoW	8.29×10^{6}	PDF.m^2.yr
0.02%	mastic asphalt production | mastic asphalt | Cutoff, S - RoW	7.28×10^{6}	PDF.m^2.yr
0.02%	gravel and sand quarry operation | sand | Cutoff, S - RoW	6.92×10^{6}	PDF.m^2.yr
0.01%	fleece production, polyethylene | fleece, polyethylene | Cutoff, S - RoW	2.97×10^{6}	PDF.m^2.yr
0.00%	flat glass production, uncoated | flat glass, uncoated | Cutoff, S - RoW	1.78×10^{6}	PDF.m^2.yr
0.00%	market for sawnwood, board, hardwood, dried (u=10%), planed |sawnwood, board, hardwood, dried (u=10%), planed | Cutoff, S - Europe without Switzerland	8.17×10^{5}	PDF.m^2.yr
0.00%	gypsum plasterboard production | gypsum plasterboard | Cutoff, S - RoW	1.70×10^{5}	PDF.m^2.yr
0.00%	market for natural gas, low pressure | natural gas, low pressure | Cutoff, S - RoW	5.38×10^{4}	PDF.m^2.yr

Table A45 Life cycle contributions to freshwater ecotoxicity (long term) in the sedum roofs scenario

Contribution	Process	Amount	Unit
96.78%	Transport	4.28×10^{10}	PDF.m^2.yr
1.92%	Rock wool	5.74×10^{8}	PDF.m^2.yr
0.49%	Electricity	1.48×10^{8}	PDF.m^2.yr
0.44%	Cement	1.32×10^{8}	PDF.m^2.yr
0.16%	Aluminium	4.69×10^{7}	PDF.m^2.yr
0.07%	gravel production, crushed | gravel, crushed | Cutoff, S - RoW	1.97×10^{7}	PDF.m^2.yr
0.04%	market for sealing tape, aluminium/PE, 50 mm wide | sealing tape, aluminium/PE, 50 mm wide | Cutoff, S - GLO	1.35×10^{7}	PDF.m^2.yr
0.04%	autoclaved aerated concrete block production | autoclaved aerated concrete block | Cutoff, S - RoW	1.05×10^{7}	PDF.m^2.yr
0.03%	clay brick production | clay brick | Cutoff, S - RoW	8.29×10^{6}	PDF.m^2.yr
0.02%	gravel and sand quarry operation | sand | Cutoff, S - RoW	6.92×10^{6}	PDF.m^2.yr
0.01%	fleece production, polyethylene | fleece, polyethylene | Cutoff, S - RoW	1.81×10^{6}	PDF.m^2.yr
0.01%	flat glass production, uncoated | flat glass, uncoated | Cutoff, S - RoW	1.78×10^{6}	PDF.m^2.yr
0.00%	market for sawnwood, board, hardwood, dried (u=10%), planed | sawnwood, board, hardwood, dried (u=10%), planed | Cutoff, S - Europe without Switzerland	8.17×10^{5}	PDF.m^2.yr
0.00%	market for textile, nonwoven polypropylene | textile, nonwoven polypropylene | Cutoff, S - GLO	5.01×10^{5}	PDF.m^2.yr
0.00%	gypsum plasterboard production | gypsum plasterboard | Cutoff, S - RoW	1.70×10^{5}	PDF.m^2.yr
0.00%	market for polystyrene, general purpose | polystyrene, general purpose | Cutoff, S - GLO	5.68×10^{4}	PDF.m^2.yr
0.00%	market for natural gas, low pressure | natural gas, low pressure | Cutoff, S - RoW	5.38×10^{4}	PDF.m^2.yr
0.00%	market for clay | clay | Cutoff, S - RoW	2.36×10^{4}	PDF.m^2.yr
0.00%	market for compost | compost | Cutoff, S - GLO	5.56×10^{3}	PDF.m^2.yr
0.00%	market for waste brick | waste brick | Cutoff, S - Europe without Switzerland	-5.92×10^{5}	PDF.m^2.yr

Table A46 Contributions to freshwater ecotoxicity (long term) in the concrete roofs scenario: construction phase

Contribution	Process	Amount	Unit
97.33%	Transport	$3.10x10^{10}$	$PDF.m^2.yr$
1.87%	Rock wool	$5.76x10^{8}$	$PDF.m^2.yr$
0.43%	Cement	$1.32x10^{8}$	$PDF.m^2.yr$
0.24%	window frame production, aluminium, U=1.6 W/m2K | window frame, aluminium, U=1.6 W/m2K | Cutoff, S - RoW	$4.69x10^{7}$	$PDF.m^2.yr$
0.05%	gravel production, crushed | gravel, crushed | Cutoff, S - RoW	$2.14x10^{7}$	$PDF.m^2.yr$
0.04%	market for sealing tape, aluminium/PE, 50 mm wide | sealing tape, aluminium/PE, 50 mm wide | Cutoff, S - GLO	$1.35x10^{7}$	$PDF.m^2.yr$
0.02%	autoclaved aerated concrete block production | autoclaved aerated concrete block | Cutoff, S - RoW	$1.05x10^{7}$	$PDF.m^2.yr$
0.02%	clay brick production | clay brick | Cutoff, S - RoW	$8.29x10^{6}$	$PDF.m^2.yr$
0.01%	gravel and sand quarry operation | sand | Cutoff, S - RoW	$6.92x10^{6}$	$PDF.m^2.yr$
0.01%	flat glass production, uncoated | flat glass, uncoated | Cutoff, S - RoW	$1.78x10^{6}$	$PDF.m^2.yr$
0.01%	fleece production, polyethylene | fleece, polyethylene | Cutoff, S - RoW	$1.77x10^{6}$	$PDF.m^2.yr$
0.00%	mastic asphalt production | mastic asphalt | Cutoff, S - RoW	$1.46x10^{6}$	$PDF.m^2.yr$
0.00%	market for sawnwood, board, hardwood, dried (u=10%), planed | sawnwood, board, hardwood, dried (u=10%), planed | Cutoff, S - Europe without Switzerland	$8.17x10^{5}$	$PDF.m^2.yr$
0.00%	gypsum plasterboard production | gypsum plasterboard | Cutoff, S - RoW	$1.70x10^{5}$	$PDF.m^2.yr$

Table A47 Contributions to freshwater ecotoxicity (long term) in the concrete roofs scenario: use phase

Contribution	Process	Amount	Unit
96.36%	Transport	$4.32x10^{9}$	$PDF.m^2.yr$
3.30%	Electricity	$1.48x10^{8}$	$PDF.m^2.yr$
0.18%	Gravel	$8.02x10^{6}$	$PDF.m^2.yr$
0.13%	mastic asphalt production | mastic asphalt | Cutoff, S - RoW	$5.83x10^{6}$	$PDF.m^2.yr$
0.03%	fleece production, polyethylene | fleece, polyethylene | Cutoff, S - RoW	$1.19x10^{6}$	$PDF.m^2.yr$
0.00%	market for natural gas, low pressure | natural gas, low pressure | Cutoff, S - RoW	$5.38x10^{4}$	$PDF.m^2.yr$

Table A48 Contributions to freshwater ecotoxicity (long term) in the concrete roofs scenario: end of life phase

Contribution	Process	Amount	Unit
100.00%	Transport	$5.22x10^{7}$	$PDF.m^2.yr$

Table A49 Contributions to freshwater ecotoxicity (long term) in the sedum roofs scenario: construction phase

Contribution	Process	Amount	Unit
96.78%	Transport	3.01E+10	PDF.m^2.yr
1.92%	Rock wool	5.74E+08	PDF.m^2.yr
0.49%	Electricity	1.48E+08	PDF.m^2.yr
0.44%	Cement	1.32E+08	PDF.m^2.yr
0.16%	window frame production, aluminium, U=1.6 W/m2K | window frame, aluminium, U=1.6 W/m2K | Cutoff, S - RoW	4.69E+07	PDF.m^2.yr
0.07%	gravel production, crushed | gravel, crushed | Cutoff, S - RoW	1.97E+07	PDF.m^2.yr
0.04%	market for sealing tape, aluminium/PE, 50 mm wide | sealing tape, aluminium/PE, 50 mm wide | Cutoff, S - GLO	1.35E+07	PDF.m^2.yr
0.04%	autoclaved aerated concrete block production | autoclaved aerated concrete block | Cutoff, S - RoW	1.05E+07	PDF.m^2.yr
0.03%	clay brick production | clay brick | Cutoff, S - RoW	8.29E+06	PDF.m^2.yr
0.02%	gravel and sand quarry operation | sand | Cutoff, S - RoW	6.92E+06	PDF.m^2.yr
0.01%	flat glass production, uncoated | flat glass, uncoated | Cutoff, S - RoW	1.78E+06	PDF.m^2.yr
0.01%	fleece production, polyethylene | fleece, polyethylene | Cutoff, S - RoW	1.64E+06	PDF.m^2.yr
0.00%	market for sawnwood, board, hardwood, dried (u=10%), planed | sawnwood, board, hardwood, dried (u=10%), planed | Cutoff, S - Europe without Switzerland	8.17E+05	PDF.m^2.yr
0.00%	market for textile, nonwoven polypropylene | textile, nonwoven polypropylene | Cutoff, S - GLO	2.50E+05	PDF.m^2.yr
0.00%	gypsum plasterboard production | gypsum plasterboard | Cutoff, S - RoW	1.70E+05	PDF.m^2.yr
0.00%	market for natural gas, low pressure | natural gas, low pressure | Cutoff, S - RoW	5.38E+04	PDF.m^2.yr
0.00%	market for polystyrene, general purpose | polystyrene, general purpose | Cutoff, S - GLO	2.84E+04	PDF.m^2.yr
0.00%	market for clay | clay | Cutoff, S - RoW	1.18E+04	PDF.m^2.yr
0.00%	market for compost | compost | Cutoff, S - GLO	5220.86023	PDF.m^2.yr
0.00%	market for waste brick | waste brick | Cutoff, S - Europe without Switzerland	-2.96E+05	PDF.m^2.yr

Table A50 Contributions to freshwater ecotoxicity (long term) in the sedum roofs scenario: use phase

Contribution	Process	Amount	Unit
95.31%	Electricity	$1.48x10^{8}$	$PDF.m^2.yr$
4.54%	Transport	$1.32x10^{7}$	$PDF.m^2.yr$
0.16%	Polypropylene	$2.50x10^{5}$	$PDF.m^2.yr$
0.11%	fleece production, polyethylene | fleece, polyethylene | Cutoff, S - RoW	$1.69x10^{5}$	$PDF.m^2.yr$
0.03%	market for natural gas, low pressure | natural gas, low pressure | Cutoff, S - RoW	$5.38x10^{4}$	$PDF.m^2.yr$
0.02%	market for polystyrene, general purpose | polystyrene, general purpose | Cutoff, S - GLO	$2.84x10^{4}$	$PDF.m^2.yr$
0.01%	market for stone wool | stone wool | Cutoff, S - GLO	$1.39x10^{4}$	$PDF.m^2.yr$
0.01%	market for clay | clay | Cutoff, S - RoW	$1.18x10^{4}$	$PDF.m^2.yr$
0.00%	market for compost | compost | Cutoff, S - GLO	344.48	$PDF.m^2.yr$
-0.19%	market for waste brick | waste brick | Cutoff, S - Europe without Switzerland	$-2.96x10^{5}$	$PDF.m^2.yr$

Table A51 Contributions to freshwater ecotoxicity (long term) in the sedum roofs scenario: end of life phase

Contribution	Process	Amount	Unit
100.00%	Transport	$4.44x10^{7}$	$PDF.m^2.yr$

10.5.1 Data pertaining to the sensitivity analysis in Chapter 4

The following data (Tables A52-A65) supplement the sensitivity analysis performed as part of Chapter 4. Here the focus is the change in life cycle ecological impacts and value of freshwater ecotoxicity with changing transport and end of life land transformation.

Table A52 Concrete roofs: 25% less transportation

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$3.97x10^{10}$	$PDF.m^2.yr$
Climate change, ecosystem quality, long term	$5.07x10^{9}$	$PDF.m^2.yr$
Climate change, ecosystem quality, short term	$1.49x10^{9}$	$PDF.m^2.yr$
Marine acidification, long term	$1.22x10^{9}$	$PDF.m^2.yr$
Land transformation, biodiversity	$7.92x10^{8}$	$PDF.m^2.yr$
Terrestrial acidification	$6.19x10^{8}$	$PDF.m^2.yr$
Land occupation, biodiversity	$3.07x10^{8}$	$PDF.m^2.yr$
Freshwater ecotoxicity, short term	$1.40x10^{8}$	$PDF.m^2.yr$
Marine acidification, short term	$1.33x10^{8}$	$PDF.m^2.yr$
Freshwater acidification	$8.61x10^{7}$	$PDF.m^2.yr$
Marine eutrophication	$1.58x10^{7}$	$PDF.m^2.yr$
Freshwater eutrophication	$2.59x10^{6}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$1.05x10^{5}$	$PDF.m^2.yr$
Climate change, human health, long term	$2.31x10^{4}$	DALY
Water availability, freshwater ecosystem	$7.37x10^{3}$	$PDF.m^2.yr$
Climate change, human health, short term	$6.90x10^{3}$	DALY
Particulate matter formation	$6.09x10^{3}$	DALY
Thermally polluted water	$4.15x10^{3}$	$PDF.m^2.yr$
Water availability, human health	$3.11x10^{3}$	DALY
Human toxicity cancer, short term	$1.50x10^{3}$	DALY
Human toxicity non-cancer, short term	$1.09x10^{3}$	DALY
Human toxicity non-cancer, long term	931.54	DALY
Human toxicity cancer, long term	26.84	DALY
Ionizing radiation, human health	10.39	DALY
Ozone layer depletion	2.93	DALY
Photochemical oxidant formation	2.73	DALY
Ionizing radiation, ecosystem quality	1.58	$PDF.m^2.yr$
Total ecological impacts	**$4.96x10^{10}$**	**$PDF.m^2.yr$**

Table A53 Concrete roofs: 25% more transportation

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$6.27x10^{10}$	PDF.m^2.yr
Climate change, ecosystem quality, long term	$8.00x10^{9}$	PDF.m^2.yr
Climate change, ecosystem quality, short term	$2.35x10^{9}$	PDF.m^2.yr
Marine acidification, long term	$1.93x10^{9}$	PDF.m^2.yr
Land transformation, biodiversity	$1.26x10^{9}$	PDF.m^2.yr
Terrestrial acidification	$9.79x10^{8}$	PDF.m^2.yr
Land occupation, biodiversity	$4.88x10^{8}$	PDF.m^2.yr
Freshwater ecotoxicity, short term	$2.21x10^{8}$	PDF.m^2.yr
Marine acidification, short term	$2.09x10^{8}$	PDF.m^2.yr
Freshwater acidification	$1.36x10^{8}$	PDF.m^2.yr
Marine eutrophication	$2.50x10^{7}$	PDF.m^2.yr
Freshwater eutrophication	$4.11x10^{6}$	PDF.m^2.yr
Water availability, terrestrial ecosystem	$1.61x10^{5}$	PDF.m^2.yr
Climate change, human health, long term	$3.64x10^{4}$	DALY
Water availability, freshwater ecosystem	$1.15x10^{4}$	PDF.m^2.yr
Climate change, human health, short term	$1.09x10^{4}$	DALY
Particulate matter formation	$9.66x10^{3}$	DALY
Thermally polluted water	$6.50x10^{3}$	PDF.m^2.yr
Water availability, human health	$4.86x10_{3}$	DALY
Human toxicity cancer, short term	$2.38x10^{3}$	DALY
Human toxicity non-cancer, short term	$1.73x10^{3}$	DALY
Human toxicity non-cancer, long term	$1.47x10^{3}$	DALY
Human toxicity cancer, long term	42.18	DALY
Ionizing radiation, human health	16.40	DALY
Ozone layer depletion	4.65	DALY
Photochemical oxidant formation	4.34	DALY
Ionizing radiation, ecosystem quality	2.50	PDF.m^2.yr
Total ecological impacts	**$7.83x10^{10}$**	**PDF.m^2.yr**

Table A54 Concrete roofs: 7-16 tonne vehicles

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$7.07x10^{10}$	PDF.m^2.yr
Climate change, ecosystem quality, long term	$9.28x10^{9}$	PDF.m^2.yr
Climate change, ecosystem quality, short term	$2.72x10^{9}$	PDF.m^2.yr
Marine acidification, long term	$2.24x10^{9}$	PDF.m^2.yr
Land transformation, biodiversity	$1.24x10^{9}$	PDF.m^2.yr
Terrestrial acidification	$8.62x10^{8}$	PDF.m^2.yr
Land occupation, biodiversity	$4.07x10^{8}$	PDF.m^2.yr
Marine acidification, short term	$2.43x10^{8}$	PDF.m^2.yr
Freshwater ecotoxicity, short term	$2.26x10^{8}$	PDF.m^2.yr
Freshwater acidification	$1.24x10^{8}$	PDF.m^2.yr
Marine eutrophication	$1.83x10^{7}$	PDF.m^2.yr
Freshwater eutrophication	$4.51x10^{6}$	PDF.m^2.yr
Water availability, terrestrial ecosystem	$1.69x10^{5}$	PDF.m^2.yr
Climate change, human health, long term	$4.23x10^{4}$	DALY
Climate change, human health, short term	$1.26x10^{4}$	DALY
Water availability, freshwater ecosystem	$1.15x10^{4}$	PDF.m^2.yr
Particulate matter formation	$8.06x10^{3}$	DALY
Thermally polluted water	$6.84x10^{3}$	PDF.m^2.yr
Water availability, human health	$5.09x10^{3}$	DALY
Human toxicity cancer, short term	$2.40x10^{3}$	DALY
Human toxicity non-cancer, short term	$1.88x10^{3}$	DALY
Human toxicity non-cancer, long term	$1.48x10^{3}$	DALY
Human toxicity cancer, long term	44.62	DALY
Ionizing radiation, human health	17.62	DALY
Ozone layer depletion	5.10	DALY
Photochemical oxidant formation	3.24	DALY
Ionizing radiation, ecosystem quality	2.72	PDF.m^2.yr
Total ecological impacts	**$8.81x10^{10}$**	**PDF.m^2.yr**

Table A55 Concrete roofs: >32 tonne vehicles

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$2.28x10^{10}$	$PDF.m^2.yr$
Land transformation, biodiversity	$6.55x10^{9}$	$PDF.m^2.yr$
Climate change, ecosystem quality, long term	$4.52x10^{9}$	$PDF.m^2.yr$
Climate change, ecosystem quality, short term	$1.33x10^{9}$	$PDF.m^2.yr$
Marine acidification, long term	$1.09x10^{9}$	$PDF.m^2.yr$
Land occupation, biodiversity	$3.90x10^{8}$	$PDF.m^2.yr$
Terrestrial acidification	$3.18x10^{8}$	$PDF.m^2.yr$
Freshwater ecotoxicity, short term	$1.32x10^{8}$	$PDF.m^2.yr$
Marine acidification, short term	$1.18x10^{8}$	$PDF.m^2.yr$
Freshwater acidification	$4.51x10^{7}$	$PDF.m^2.yr$
Marine eutrophication	$7.41x10^{6}$	$PDF.m^2.yr$
Freshwater eutrophication	$3.69x10^{6}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$1.74x10^{5}$	$PDF.m^2.yr$
Climate change, human health, long term	$2.06x10^{4}$	DALY
Water availability, freshwater ecosystem	$8.18x10^{3}$	$PDF.m^2.yr$
Climate change, human health, short term	$6.16x10^{3}$	DALY
Particulate matter formation	$5.28x10^{3}$	DALY
Thermally polluted water	$4.39x10^{3}$	$PDF.m^2.yr$
Water availability, human health	$3.35x10^{3}$	DALY
Human toxicity cancer, short term	$1.55x10^{3}$	DALY
Human toxicity non-cancer, long term	$1.17x10^{3}$	DALY
Human toxicity non-cancer, short term	890.16	DALY
Human toxicity cancer, long term	28.97	DALY
Ionizing radiation, human health	12.98	DALY
Ionizing radiation, ecosystem quality	2.04	$PDF.m^2.yr$
Ozone layer depletion	1.82	DALY
Photochemical oxidant formation	1.59	DALY
Total ecological impacts	**$3.73x10^{10}$**	**$PDF.m^2.yr$**

Table A56 Sedum roofs: 25% less transportation

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$3.31x10^{10}$	$PDF.m^2.yr$
Climate change, ecosystem quality, long term	$4.23x10^{9}$	$PDF.m^2.yr$
Climate change, ecosystem quality, short term	$1.24x10^{9}$	$PDF.m^2.yr$
Marine acidification, long term	$1.02x10^{9}$	$PDF.m^2.yr$
Land transformation, biodiversity	$6.57x10^{8}$	$PDF.m^2.yr$
Terrestrial acidification	$5.15x10^{8}$	$PDF.m^2.yr$
Land occupation, biodiversity	$2.55x10^{8}$	$PDF.m^2.yr$
Freshwater ecotoxicity, short term	$1.16x10^{8}$	$PDF.m^2.yr$
Marine acidification, short term	$1.11x10^{8}$	$PDF.m^2.yr$
Freshwater acidification	$7.17x10^{7}$	$PDF.m^2.yr$
Marine eutrophication	$1.31x10^{7}$	$PDF.m^2.yr$
Freshwater eutrophication	$2.15x10^{6}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$8.92x10^{4}$	$PDF.m^2.yr$
Climate change, human health, long term	$1.92x10^{4}$	DALY
Water availability, freshwater ecosystem	$6.18x10^{3}$	$PDF.m^2.yr$
Climate change, human health, short term	$5.75x10^{3}$	DALY
Particulate matter formation	$5.07x10^{3}$	DALY
Thermally polluted water	$3.47x10^{3}$	$PDF.m^2.yr$
Water availability, human health	$2.60x10^{3}$	DALY
Human toxicity cancer, short term	$1.25x10^{3}$	DALY
Human toxicity non-cancer, short term	908.81	DALY
Human toxicity non-cancer, long term	777.27	DALY
Human toxicity cancer, long term	22.42	DALY
Ionizing radiation, human health	8.66	DALY
Ozone layer depletion	2.43	DALY
Photochemical oxidant formation	2.27	DALY
Ionizing radiation, ecosystem quality	1.31	$PDF.m^2.yr$
Total ecological impacts	**$4.13x10^{10}$**	**$PDF.m^2.yr$**

Table A57 Sedum roofs: 25% more transportation

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$5.44x10^{10}$	$PDF.m^2.yr$
Climate change, ecosystem quality, long term	$6.95x10^{9}$	$PDF.m^2.yr$
Climate change, ecosystem quality, short term	$2.04x10^{9}$	$PDF.m^2.yr$
Marine acidification, long term	$1.67x10^{9}$	$PDF.m^2.yr$
Land transformation, biodiversity	$1.09x10^{9}$	$PDF.m^2.yr$
Terrestrial acidification	$8.50x10^{8}$	$PDF.m^2.yr$
Land occupation, biodiversity	$4.23x10^{8}$	$PDF.m^2.yr$
Freshwater ecotoxicity, short term	$1.92x10^{8}$	$PDF.m^2.yr$
Marine acidification, short term	$1.82x10^{8}$	$PDF.m^2.yr$
Freshwater acidification	$1.18x10^{8}$	$PDF.m^2.yr$
Marine eutrophication	$2.17x10^{7}$	$PDF.m^2.yr$
Freshwater eutrophication	$3.57x10^{6}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$1.41x10^{5}$	$PDF.m^2.yr$
Climate change, human health, long term	$3.16x10^{4}$	DALY
Water availability, freshwater ecosystem	$9.97x10^{3}$	$PDF.m^2.yr$
Climate change, human health, short term	$9.45x10^{3}$	DALY
Particulate matter formation	$8.38x10^{3}$	DALY
Thermally polluted water	$5.66x10^{3}$	$PDF.m^2.yr$
Water availability, human health	$4.23x10^{3}$	DALY
Human toxicity cancer, short term	$2.06x10^{3}$	DALY
Human toxicity non-cancer, short term	$1.50x10^{3}$	DALY
Human toxicity non-cancer, long term	$1.28x10^{3}$	DALY
Human toxicity cancer, long term	36.68	DALY
Ionizing radiation, human health	14.24	DALY
Ozone layer depletion	4.03	DALY
Photochemical oxidant formation	3.77	DALY
Ionizing radiation, ecosystem quality	2.17	$PDF.m^2.yr$
Total ecological impacts	**$6.80x10^{10}$**	**$PDF.m^2.yr$**

Table A58 Sedum roofs: 7-16 tonne vehicles

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$6.03x10^{10}$	PDF.m^2.yr
Climate change, ecosystem quality, long term	$7.93x10^{9}$	PDF.m^2.yr
Climate change, ecosystem quality, short term	$2.33x10^{9}$	PDF.m^2.yr
Marine acidification, long term	$1.91x10^{9}$	PDF.m^2.yr
Land transformation, biodiversity	$1.06x10^{9}$	PDF.m^2.yr
Terrestrial acidification	$7.36x10^{8}$	PDF.m^2.yr
Land occupation, biodiversity	$3.48x10^{8}$	PDF.m^2.yr
Marine acidification, short term	$2.08x10^{8}$	PDF.m^2.yr
Freshwater ecotoxicity, short term	$1.93x10^{8}$	PDF.m^2.yr
Freshwater acidification	$1.06x10^{8}$	PDF.m^2.yr
Marine eutrophication	$1.56x10^{7}$	PDF.m^2.yr
Freshwater eutrophication	$3.84x10^{6}$	PDF.m^2.yr
Water availability, terrestrial ecosystem	$1.46x10^{5}$	PDF.m^2.yr
Climate change, human health, long term	$3.61x10^{4}$	DALY
Climate change, human health, short term	$1.08x10^{4}$	DALY
Water availability, freshwater ecosystem	$9.87x10^{3}$	PDF.m^2.yr
Particulate matter formation	$6.88x10^{3}$	DALY
Thermally polluted water	$5.85x10^{3}$	PDF.m^2.yr
Water availability, human health	$4.36x10^{3}$	DALY
Human toxicity cancer, short term	$2.05x10^{3}$	DALY
Human toxicity non-cancer, short term	$1.61x10^{3}$	DALY
Human toxicity non-cancer, long term	$1.26x10^{3}$	DALY
Human toxicity cancer, long term	38.15	DALY
Ionizing radiation, human health	15.04	DALY
Ozone layer depletion	4.35	DALY
Photochemical oxidant formation	2.76	DALY
Ionizing radiation, ecosystem quality	2.32	PDF.m^2.yr
Total ecological impacts	**$7.52x10^{10}$**	**PDF.m^2.yr**

Table A59 Sedum roofs: >32 tonne vehicles

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$1.96x10^{10}$	$PDF.m^2.yr$
Land transformation, biodiversity	$5.58x10^{9}$	$PDF.m^2.yr$
Climate change, ecosystem quality, long term	$3.87x10^{9}$	$PDF.m^2.yr$
Climate change, ecosystem quality, short term	$1.14x10^{9}$	$PDF.m^2.yr$
Marine acidification, long term	$9.30x10^{8}$	$PDF.m^2.yr$
Land occupation, biodiversity	$3.33x10^{8}$	$PDF.m^2.yr$
Terrestrial acidification	$2.73x10^{8}$	$PDF.m^2.yr$
Freshwater ecotoxicity, short term	$1.13x10^{8}$	$PDF.m^2.yr$
Marine acidification, short term	$1.01x10^{8}$	$PDF.m^2.yr$
Freshwater acidification	$3.87x10^{7}$	$PDF.m^2.yr$
Marine eutrophication	$6.33x10^{6}$	$PDF.m^2.yr$
Freshwater eutrophication	$3.14x10^{6}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$1.50x10^{5}$	$PDF.m^2.yr$
Climate change, human health, long term	$1.76x10^{4}$	DALY
Water availability, freshwater ecosystem	$7.02x10^{3}$	$PDF.m^2.yr$
Climate change, human health, short term	$5.27x10^{3}$	DALY
Particulate matter formation	$4.51x10^{3}$	DALY
Thermally polluted water	$3.77x10^{3}$	$PDF.m^2.yr$
Water availability, human health	$2.87x10^{3}$	DALY
Human toxicity cancer, short term	$1.33x10^{3}$	DALY
Human toxicity non-cancer, long term	$1.00x10^{3}$	DALY
Human toxicity non-cancer, short term	760.71	DALY
Human toxicity cancer, long term	24.82	DALY
Ionizing radiation, human health	11.09	DALY
Ionizing radiation, ecosystem quality	1.74	$PDF.m^2.yr$
Ozone layer depletion	1.56	DALY
Photochemical oxidant formation	1.36	DALY
Total ecological impacts	**$3.19x10^{10}$**	**$PDF.m^2.yr$**

Table A60 Concrete roofs: end of life: manmade pasture

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$5.12x10^{10}$	PDF.m^2.yr
Climate change, ecosystem quality, long term	$6.54x10^{9}$	PDF.m^2.yr
Climate change, ecosystem quality, short term	$1.92x10^{9}$	PDF.m^2.yr
Marine acidification, long term	$1.58x10^{9}$	PDF.m^2.yr
Land transformation, biodiversity	$1.02x10^{9}$	PDF.m^2.yr
Terrestrial acidification	$7.99x10^{8}$	PDF.m^2.yr
Land occupation, biodiversity	$3.98x10^{8}$	PDF.m^2.yr
Freshwater ecotoxicity, short term	$1.80x10^{8}$	PDF.m^2.yr
Marine acidification, short term	$1.71x10^{8}$	PDF.m^2.yr
Freshwater acidification	$1.11x10^{8}$	PDF.m^2.yr
Marine eutrophication	$2.04x10^{7}$	PDF.m^2.yr
Freshwater eutrophication	$3.35x10^{6}$	PDF.m^2.yr
Water availability, terrestrial ecosystem	$1.33x10^{5}$	PDF.m^2.yr
Climate change, human health, long term	$2.98x10^{4}$	DALY
Water availability, freshwater ecosystem	$9.42x10^{3}$	PDF.m^2.yr
Climate change, human health, short term	$8.89x10^{3}$	DALY
Particulate matter formation	$7.88x10^{3}$	DALY
Thermally polluted water	$5.33x10^{3}$	PDF.m^2.yr
Water availability, human health	$3.99x10^{3}$	DALY
Human toxicity cancer, short term	$1.94x10^{3}$	DALY
Human toxicity non-cancer, short term	$1.41x10^{3}$	DALY
Human toxicity non-cancer, long term	$1.20x10^{3}$	DALY
Human toxicity cancer, long term	34.53	DALY
Ionizing radiation, human health	13.41	DALY
Ozone layer depletion	3.79	DALY
Photochemical oxidant formation	3.54	DALY
Ionizing radiation, ecosystem quality	2.04	PDF.m^2.yr
Total ecological impact	**$6.39x10^{10}$**	**PDF.m^2.yr**
Transformation as a % of case study	**-0.9**	**%**
Total ecological impact % change	**0.0**	**%**

Table A61 Concrete roofs: end of life: arable land

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$5.12x10^{10}$	PDF.m^2.yr
Climate change, ecosystem quality, long term	$6.54x10^{9}$	PDF.m^2.yr
Climate change, ecosystem quality, short term	$1.92x10^{9}$	PDF.m^2.yr
Marine acidification, long term	$1.58x10^{9}$	PDF.m^2.yr
Land transformation, biodiversity	$1.03x10^{9}$	PDF.m^2.yr
Terrestrial acidification	$7.99x10^{8}$	PDF.m^2.yr
Land occupation, biodiversity	$3.98x10^{8}$	PDF.m^2.yr
Freshwater ecotoxicity, short term	$1.80x10^{8}$	PDF.m^2.yr
Marine acidification, short term	$1.71x10^{8}$	PDF.m^2.yr
Freshwater acidification	$1.11x10^{8}$	PDF.m^2.yr
Marine eutrophication	$2.04x10^{7}$	PDF.m^2.yr
Freshwater eutrophication	$3.35x10^{6}$	PDF.m^2.yr
Water availability, terrestrial ecosystem	$1.33x10^{5}$	PDF.m^2.yr
Climate change, human health, long term	$2.98x10^{4}$	DALY
Water availability, freshwater ecosystem	$9.42x10^{3}$	PDF.m^2.yr
Climate change, human health, short term	$8.89x10^{3}$	DALY
Particulate matter formation	$7.88x10^{3}$	DALY
Thermally polluted water	$5.33x10^{3}$	PDF.m^2.yr
Water availability, human health	$3.99x10^{3}$	DALY
Human toxicity cancer, short term	$1.94x10^{3}$	DALY
Human toxicity non-cancer, short term	$1.41x10^{3}$	DALY
Human toxicity non-cancer, long term	1.20x103	DALY
Human toxicity cancer, long term	34.53	DALY
Ionizing radiation, human health	13.41	DALY
Ozone layer depletion	3.79	DALY
Photochemical oxidant formation	3.54	DALY
Ionizing radiation, ecosystem quality	2.04	PDF.m^2.yr
Total ecological impact	**$6.40x10^{10}$**	**PDF.m^2.yr**
Transformation as a % of case study	**0.6**	**%**
Total ecological impact % change	**0.0**	**%**

Table A62 Sedum roofs: end of life: manmade pasture

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$4.37x10^{10}$	PDF.m^2.yr
Climate change, ecosystem quality, long term	$5.59x10^{9}$	PDF.m^2.yr
Climate change, ecosystem quality, short term	$1.64x10^{9}$	PDF.m^2.yr
Marine acidification, long term	$1.35x10^{9}$	PDF.m^2.yr
Land transformation, biodiversity	$8.64x10^{8}$	PDF.m^2.yr
Terrestrial acidification	$6.82x10^{8}$	PDF.m^2.yr
Land occupation, biodiversity	$3.39x10^{8}$	PDF.m^2.yr
Freshwater ecotoxicity, short term	$1.54x10^{8}$	PDF.m^2.yr
Marine acidification, short term	$1.46x10^{8}$	PDF.m^2.yr
Freshwater acidification	$9.50x10^{7}$	PDF.m^2.yr
Marine eutrophication	$1.74x10^{7}$	PDF.m^2.yr
Freshwater eutrophication	$2.86x10^{6}$	PDF.m^2.yr
Water availability, terrestrial ecosystem	$1.15x10^{5}$	PDF.m^2.yr
Climate change, human health, long term	$2.54x10^{4}$	DALY
Water availability, freshwater ecosystem	$8.08x10^{3}$	PDF.m^2.yr
Climate change, human health, short term	$7.60x10^{3}$	DALY
Particulate matter formation	$6.72x10^{3}$	DALY
Thermally polluted water	$4.56x10^{3}$	PDF.m^2.yr
Water availability, human health	$3.42x10^{3}$	DALY
Human toxicity cancer, short term	$1.66x10^{3}$	DALY
Human toxicity non-cancer, short term	$1.21x10^{3}$	DALY
Human toxicity non-cancer, long term	$1.03x10^{3}$	DALY
Human toxicity cancer, long term	29.55	DALY
Ionizing radiation, human health	11.45	DALY
Ozone layer depletion	3.23	DALY
Photochemical oxidant formation	3.02	DALY
Ionizing radiation, ecosystem quality	1.74	PDF.m^2.yr
Total ecological impact	**$5.46x10^{10}$**	**PDF.m^2.yr**
Transformation as a % of case study	**-1.1**	**%**
Total ecological impact % change	**0.0**	**%**

Table A63 Sedum roofs: end of life: arable land

Name	Impact result	Unit
Freshwater ecotoxicity, long term	$4.37x10^{10}$	PDF.m^2.yr
Climate change, ecosystem quality, long term	$5.59x10^{9}$	PDF.m^2.yr
Climate change, ecosystem quality, short term	$1.64x10^{9}$	PDF.m^2.yr
Marine acidification, long term	$1.35x10^{9}$	PDF.m^2.yr
Land transformation, biodiversity	$8.79x10^{8}$	PDF.m^2.yr
Terrestrial acidification	$6.82x10^{8}$	PDF.m^2.yr
Land occupation, biodiversity	$3.39x10^{8}$	PDF.m^2.yr
Freshwater ecotoxicity, short term	$1.54x10^{8}$	PDF.m^2.yr
Marine acidification, short term	$1.46x10^{8}$	PDF.m^2.yr
Freshwater acidification	$9.50x10^{7}$	PDF.m^2.yr
Marine eutrophication	$1.74x10^{7}$	PDF.m^2.yr
Freshwater eutrophication	$2.86x10^{6}$	PDF.m^2.yr
Water availability, terrestrial ecosystem	$1.15x10^{5}$	PDF.m^2.yr
Climate change, human health, long term	$2.54x10^{4}$	DALY
Water availability, freshwater ecosystem	$8.08x10^{3}$	PDF.m^2.yr
Climate change, human health, short term	$7.60x10^{3}$	DALY
Particulate matter formation	$6.72x10^{3}$	DALY
Thermally polluted water	$4.56x10^{3}$	PDF.m^2.yr
Water availability, human health	$3.42x10^{3}$	DALY
Human toxicity cancer, short term	$1.66x10^{3}$	DALY
Human toxicity non-cancer, short term	$1.21x10^{3}$	DALY
Human toxicity non-cancer, long term	$1.03x10^{3}$	DALY
Human toxicity cancer, long term	29.55	DALY
Ionizing radiation, human health	11.45	DALY
Ozone layer depletion	3.23	DALY
Photochemical oxidant formation	3.02	DALY
Ionizing radiation, ecosystem quality	1.74	PDF.m^2.yr
Total ecological impact	**5.46E+10**	**PDF.m^2.yr**
Transformation as a % of case study	**0.7**	**%**
Total ecological impact % change	**0.0**	**%**

10.5.2 Investigation: The effect of transportation distance on the capacity to offset ex-situ biodiversity impacts at Meridian One through sedum roofs

The following data (Table A64) consider the theoretical change in transportation required to offset the life cycle ex-situ impacts on biodiversity associated with the development of Meridian One with sedum roofs (as described in Chapter 4).

Table A64 Calculating the transport demand required to offset ex-situ ecological impacts in the sedum roofs scenario. Three variations are presented: (**a**) 100% transport requirement (case study), (**b**) 75% transport requirement, and (**c**) 1% transport requirement. The data are plotted (right) and an answer inferred

(a) 100% transport requirement (case study)		
Name	**Impact result**	**Unit**
Terrestrial acidification	$2.73x10^{8}$	$PDF.m^2.yr$
Marine eutrophication	$6.33x10^{6}$	$PDF.m^2.yr$
Human toxicity cancer, short term	$1.33x10^{3}$	DALY
Freshwater acidification	$3.87x10^{7}$	$PDF.m^2.yr$
Particulate matter formation	4.51x103	DALY
Ionizing radiation, ecosystem quality	1.74	$PDF.m^2.yr$
Land transformation, biodiversity	$5.57x10^{9}$	$PDF.m^2.yr$
Freshwater ecotoxicity, long term	$1.96x10^{10}$	$PDF.m^2.yr$
Ionizing radiation, human health	11.09	DALY
Freshwater ecotoxicity, short term	$1.13x10^{8}$	$PDF.m^2.yr$
Human toxicity cancer, long term	24.82	DALY
Climate change, ecosystem quality, long term	$3.87x10^{9}$	$PDF.m^2.yr$
Thermally polluted water	$3.76x10^{3}$	$PDF.m^2.yr$
Marine acidification, short term	$1.01x10^{8}$	$PDF.m^2.yr$
Freshwater eutrophication	$3.14x10^{6}$	$PDF.m^2.yr$
Human toxicity non-cancer, long term	$1.00x10^{3}$	DALY
Marine acidification, long term	$9.30x10^{8}$	$PDF.m^2.yr$
Photochemical oxidant formation	1.36	DALY
Human toxicity non-cancer, short term	760.71	DALY
Ozone layer depletion	1.56	DALY
Climate change, ecosystem quality, short term	$1.14x10^{9}$	$PDF.m^2.yr$
Climate change, human health, short term	$5.27x10^{3}$	DALY
Climate change, human health, long term	$1.76x10^{4}$	DALY
Land occupation, biodiversity	$3.33x10^{8}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$1.50x10^{5}$	$PDF.m^2.yr$
Water availability, human health	$2.87x10^{3}$	DALY
Water availability, freshwater ecosystem	$7.02x10^{3}$	$PDF.m^2.yr$
Total ecological impacts	**$3.19x10^{10}$**	**$PDF.m^2.yr$**
Total ecological impacts (including modelled result)	**$2.84x10^{10}$**	**$PDF.m^2.yr$**

Table A64 (continued)

(b) 75% transport requirement		
Name	**Impact result**	**Unit**
Terrestrial acidification	$2.08x10^{8}$	$PDF.m^2.yr$
Marine eutrophication	$4.80x10^{6}$	$PDF.m^2.yr$
Human toxicity cancer, short term	$1.00x10^{3}$	DALY
Freshwater acidification	$2.95x10^{7}$	$PDF.m^2.yr$
Particulate matter formation	$3.41x10^{3}$	DALY
Ionizing radiation, ecosystem quality	1.31	$PDF.m^2.yr$
Land transformation, biodiversity	$4.18x10^{9}$	$PDF.m^2.yr$
Freshwater ecotoxicity, long term	$1.49x10^{10}$	$PDF.m^2.yr$
Ionizing radiation, human health	8.39	DALY
Freshwater ecotoxicity, short term	$8.54x10^{7}$	$PDF.m^2.yr$
Human toxicity cancer, long term	18.87	DALY
Climate change, ecosystem quality, long term	$2.94x10^{9}$	$PDF.m^2.yr$
Thermally polluted water	$2.87x10^{3}$	$PDF.m^2.yr$
Marine acidification, short term	$7.66x10^{7}$	$PDF.m^2.yr$
Freshwater eutrophication	$2.36x10^{6}$	$PDF.m^2.yr$
Human toxicity non-cancer, long term	757.62	DALY
Marine acidification, long term	$7.06x10^{8}$	$PDF.m^2.yr$
Photochemical oxidant formation	1.03	DALY
Human toxicity non-cancer, short term	575.39	DALY
Ozone layer depletion	1.18	DALY
Climate change, ecosystem quality, short term	$8.67x10^{8}$	$PDF.m^2.yr$
Climate change, human health, short term	$4.01x10^{3}$	DALY
Climate change, human health, long term	$1.34x10^{4}$	DALY
Land occupation, biodiversity	$2.51x10^{8}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$1.15x10^{5}$	$PDF.m^2.yr$
Water availability, human health	$2.19x10^{3}$	DALY
Water availability, freshwater ecosystem	$5.39x10^{3}$	$PDF.m^2.yr$
Total ecological impact	**$2.43x10^{10}$**	**$PDF.m^2.yr$**
Total ecological impact (including modelled result)	**$2.07x10^{10}$**	**$PDF.m^2.yr$**

Table A64 (continued)

(c) 1% transport requirement		
Name	**Impact result**	**Unit**
Terrestrial acidification	$1.48x10^{7}$	$PDF.m^2.yr$
Marine eutrophication	$2.56x10^{5}$	$PDF.m^2.yr$
Human toxicity cancer, short term	40.55	DALY
Freshwater acidification	$2.17x10^{6}$	$PDF.m^2.yr$
Particulate matter formation	138.13	DALY
Ionizing radiation, ecosystem quality	0.04	$PDF.m^2.yr$
Land transformation, biodiversity	$5.49x10^{7}$	$PDF.m^2.yr$
Freshwater ecotoxicity, long term	$1.15x10^{9}$	$PDF.m^2.yr$
Ionizing radiation, human health	0.37	DALY
Freshwater ecotoxicity, short term	$4.48x10^{6}$	$PDF.m^2.yr$
Human toxicity cancer, long term	1.23	DALY
Climate change, ecosystem quality, long term	$1.70x10^{8}$	$PDF.m^2.yr$
Thermally polluted water	216.86	$PDF.m^2.yr$
Marine acidification, short term	$4.44x10^{6}$	$PDF.m^2.yr$
Freshwater eutrophication	$5.21x10^{4}$	$PDF.m^2.yr$
Human toxicity non-cancer, long term	38.56	DALY
Marine acidification, long term	$4.09x10^{7}$	$PDF.m^2.yr$
Photochemical oxidant formation	0.04	DALY
Human toxicity non-cancer, short term	25.74	DALY
Ozone layer depletion	0.04	DALY
Climate change, ecosystem quality, short term	$5.39x10^{7}$	$PDF.m^2.yr$
Climate change, human health, short term	249.07	DALY
Climate change, human health, long term	774.91	DALY
Land occupation, biodiversity	$6.99x10^{6}$	$PDF.m^2.yr$
Water availability, terrestrial ecosystem	$1.28x10^{4}$	$PDF.m^2.yr$
Water availability, human health	182.53	DALY
Water availability, freshwater ecosystem	545.53	$PDF.m^2.yr$
Total ecological impact	**$1.50x10^{9}$**	**$PDF.m^2.yr$**
Total ecological impact (including modelled result)	**$-2.04x10^{9}$**	**$PDF.m^2.yr$**

10.5.3 Reference species richness data for ecoregion 'Western European Broadleaf Forest' used in the IMPACT World+ LCIA method

Table A65 Species richness data used by de Baan et al. (2013b)[a,b,c] and subsequently IMPACT World+ for the ecoregion 'Western European Broadleaf Forest' (PA0445). Data for mammals, birds, reptiles, and amphibians are presented, sourced from WWF[b]

Ecoregion code	Species ID	Endemic (Y/N)	Class
PA0445	158	N	Amphibian
PA0445	161	N	Amphibian
PA0445	836	N	Amphibian
PA0445	839	N	Amphibian
PA0445	844	N	Amphibian
PA0445	972	N	Amphibian
PA0445	1119	N	Amphibian
PA0445	3326	N	Amphibian
PA0445	3364	N	Amphibian
PA0445	4148	N	Amphibian
PA0445	4183	N	Amphibian
PA0445	4195	N	Amphibian
PA0445	4247	N	Amphibian
PA0445	4294	N	Amphibian
PA0445	4307	N	Amphibian
PA0445	4343	N	Amphibian
PA0445	4661	N	Amphibian
PA0445	4666	N	Amphibian
PA0445	4669	N	Amphibian
PA0445	4671	N	Amphibian
PA0445	4674	N	Amphibian
PA0445	4677	N	Amphibian
PA0445	5990	N	Reptile
PA0445	6011	N	Reptile
PA0445	6012	N	Reptile
PA0445	6197	N	Reptile
PA0445	6706	N	Reptile
PA0445	6709	N	Reptile
PA0445	6710	N	Reptile
PA0445	7618	N	Reptile
PA0445	10128	N	Reptile
PA0445	10136	N	Reptile
PA0445	10155	N	Reptile
PA0445	10156	N	Reptile
PA0445	10184	N	Reptile
PA0445	10185	N	Reptile
PA0445	10256	N	Reptile
PA0445	10533	N	Reptile
PA0445	11996	N	Reptile
PA0445	12726	N	Reptile

Table A65 (continued)

Ecoregion code	Species ID	Endemic (Y/N)	Class
PA0445	13146	N	Mammal
PA0445	13225	N	Mammal
PA0445	13281	N	Mammal
PA0445	13290	N	Mammal
PA0445	13335	N	Mammal
PA0445	13389	N	Mammal
PA0445	13390	N	Mammal
PA0445	13394	N	Mammal
PA0445	13395	N	Mammal
PA0445	13407	N	Mammal
PA0445	13431	N	Mammal
PA0445	13474	N	Mammal
PA0445	13503	N	Mammal
PA0445	13791	N	Mammal
PA0445	13793	N	Mammal
PA0445	13797	N	Mammal
PA0445	13809	N	Mammal
PA0445	14086	N	Mammal
PA0445	14123	N	Mammal
PA0445	14132	N	Mammal
PA0445	14176	N	Mammal
PA0445	14180	N	Mammal
PA0445	14182	N	Mammal
PA0445	14186	N	Mammal
PA0445	14187	N	Mammal
PA0445	14190	N	Mammal
PA0445	14217	N	Mammal
PA0445	14218	N	Mammal
PA0445	14219	N	Mammal
PA0445	14252	N	Mammal
PA0445	14253	N	Mammal
PA0445	14255	N	Mammal
PA0445	14300	N	Mammal
PA0445	14310	N	Mammal
PA0445	14315	N	Mammal
PA0445	14319	N	Mammal
PA0445	14328	N	Mammal
PA0445	14329	N	Mammal
PA0445	14359	N	Mammal
PA0445	14385	N	Mammal
PA0445	14469	N	Mammal
PA0445	14736	N	Mammal
PA0445	14746	N	Mammal
PA0445	14823	N	Mammal
PA0445	14829	N	Mammal

Table A65 (continued)

Ecoregion code	Species ID	Endemic (Y/N)	Class
PA0445	14855	N	Mammal
PA0445	14857	N	Mammal
PA0445	14863	N	Mammal
PA0445	14864	N	Mammal
PA0445	14868	N	Mammal
PA0445	14871	N	Mammal
PA0445	14873	N	Mammal
PA0445	14959	N	Mammal
PA0445	15099	N	Mammal
PA0445	15132	N	Mammal
PA0445	15139	N	Mammal
PA0445	15152	N	Mammal
PA0445	15245	N	Mammal
PA0445	15270	N	Mammal
PA0445	15385	N	Mammal
PA0445	15451	N	Mammal
PA0445	15463	N	Mammal
PA0445	15588	N	Mammal
PA0445	15717	N	Mammal
PA0445	15749	N	Mammal
PA0445	15750	N	Mammal
PA0445	15754	N	Mammal
PA0445	15759	N	Mammal
PA0445	15802	N	Mammal
PA0445	15803	N	Mammal
PA0445	15815	N	Mammal
PA0445	15855	N	Mammal
PA0445	15892	N	Mammal
PA0445	16070	N	Mammal
PA0445	16071	Y	Mammal
PA0445	16076	N	Mammal
PA0445	16088	N	Mammal
PA0445	16089	N	Mammal
PA0445	16314	N	Mammal
PA0445	17094	N	Mammal
PA0445	17097	N	Mammal
PA0445	17103	N	Mammal
PA0445	17104	N	Mammal
PA0445	17375	N	Mammal
PA0445	17389	N	Mammal
PA0445	17395	N	Mammal
PA0445	17490	N	Bird
PA0445	17491	N	Bird
PA0445	17497	N	Bird
PA0445	17507	N	Bird

Table A65 (continued)

Ecoregion code	Species ID	Endemic (Y/N)	Class
PA0445	17509	N	Bird
PA0445	17510	N	Bird
PA0445	17688	N	Bird
PA0445	17746	N	Bird
PA0445	17749	N	Bird
PA0445	17751	N	Bird
PA0445	17755	N	Bird
PA0445	17756	N	Bird
PA0445	17758	N	Bird
PA0445	17764	N	Bird
PA0445	17783	N	Bird
PA0445	17805	N	Bird
PA0445	17810	N	Bird
PA0445	17812	N	Bird
PA0445	17821	N	Bird
PA0445	17834	N	Bird
PA0445	17842	N	Bird
PA0445	17848	N	Bird
PA0445	17852	N	Bird
PA0445	17855	N	Bird
PA0445	17859	N	Bird
PA0445	17863	N	Bird
PA0445	17865	N	Bird
PA0445	17867	N	Bird
PA0445	17872	N	Bird
PA0445	17873	N	Bird
PA0445	17876	N	Bird
PA0445	17877	N	Bird
PA0445	17880	N	Bird
PA0445	17883	N	Bird
PA0445	17885	N	Bird
PA0445	17900	N	Bird
PA0445	17908	N	Bird
PA0445	17917	N	Bird
PA0445	17920	N	Bird
PA0445	17921	N	Bird
PA0445	17922	N	Bird
PA0445	17933	N	Bird
PA0445	17947	N	Bird
PA0445	17957	N	Bird
PA0445	17981	N	Bird
PA0445	17989	N	Bird
PA0445	18001	N	Bird
PA0445	18006	N	Bird
PA0445	18032	N	Bird

Table A65 (continued)

Ecoregion code	Species ID	Endemic (Y/N)	Class
PA0445	18051	N	Bird
PA0445	18053	N	Bird
PA0445	18063	N	Bird
PA0445	18083	N	Bird
PA0445	18103	N	Bird
PA0445	18112	N	Bird
PA0445	18116	N	Bird
PA0445	18160	N	Bird
PA0445	18172	N	Bird
PA0445	18220	N	Bird
PA0445	18228	N	Bird
PA0445	18240	N	Bird
PA0445	18247	N	Bird
PA0445	18252	N	Bird
PA0445	18299	N	Bird
PA0445	18319	N	Bird
PA0445	18321	N	Bird
PA0445	18331	N	Bird
PA0445	18333	N	Bird
PA0445	18426	N	Bird
PA0445	18431	N	Bird
PA0445	18475	N	Bird
PA0445	18481	N	Bird
PA0445	18560	N	Bird
PA0445	18561	N	Bird
PA0445	18582	N	Bird
PA0445	18584	N	Bird
PA0445	18586	N	Bird
PA0445	18588	N	Bird
PA0445	18708	N	Bird
PA0445	18712	N	Bird
PA0445	18735	N	Bird
PA0445	18736	N	Bird
PA0445	18738	N	Bird
PA0445	18771	N	Bird
PA0445	18778	N	Bird
PA0445	18817	N	Bird
PA0445	18818	N	Bird
PA0445	18854	N	Bird
PA0445	18866	N	Bird
PA0445	18876	N	Bird
PA0445	18888	N	Bird
PA0445	18893	N	Bird
PA0445	18900	N	Bird
PA0445	18903	N	Bird

Table A65 (continued)

Ecoregion code	Species ID	Endemic (Y/N)	Class
PA0445	18951	N	Bird
PA0445	18960	N	Bird
PA0445	18973	N	Bird
PA0445	18979	N	Bird
PA0445	19001	N	Bird
PA0445	19009	N	Bird
PA0445	19043	N	Bird
PA0445	19076	N	Bird
PA0445	19088	N	Bird
PA0445	19089	N	Bird
PA0445	19092	N	Bird
PA0445	19101	N	Bird
PA0445	19106	N	Bird
PA0445	19119	N	Bird
PA0445	19121	N	Bird
PA0445	19135	N	Bird
PA0445	19149	N	Bird
PA0445	19202	N	Bird
PA0445	19216	N	Bird
PA0445	19221	N	Bird
PA0445	19224	N	Bird
PA0445	19270	N	Bird
PA0445	19282	N	Bird
PA0445	19650	N	Bird
PA0445	19936	N	Bird
PA0445	20079	N	Bird
PA0445	20134	N	Bird
PA0445	20159	N	Bird
PA0445	20180	N	Bird
PA0445	20209	N	Bird
PA0445	20213	N	Bird
PA0445	20240	N	Bird
PA0445	20244	N	Bird
PA0445	20258	N	Bird
PA0445	20343	N	Bird
PA0445	20469	N	Bird
PA0445	20472	N	Bird
PA0445	20867	N	Bird
PA0445	20997	N	Bird
PA0445	21020	N	Bird
PA0445	21280	N	Bird
PA0445	21371	N	Bird
PA0445	21380	N	Bird
PA0445	21381	N	Bird
PA0445	21382	N	Bird

Table A65 (continued)

Ecoregion code	Species ID	Endemic (Y/N)	Class
PA0445	21396	N	Bird
PA0445	21448	N	Bird
PA0445	21470	N	Bird
PA0445	21474	N	Bird
PA0445	23353	N	Bird
PA0445	23378	N	Bird
PA0445	23387	N	Bird
PA0445	23388	N	Bird
PA0445	23389	N	Bird
PA0445	23391	N	Bird
PA0445	23406	N	Bird
PA0445	23416	N	Bird
PA0445	23429	N	Bird
PA0445	23520	N	Bird
PA0445	23688	N	Bird
PA0445	23700	N	Bird
PA0445	23711	N	Bird
PA0445	23824	N	Bird
PA0445	23828	N	Bird
PA0445	23946	N	Bird
PA0445	23948	N	Bird
PA0445	23959	N	Bird
PA0445	23960	N	Bird
PA0445	23961	N	Bird
PA0445	23963	N	Bird
PA0445	24131	N	Bird
PA0445	24187	N	Bird
PA0445	24215	N	Bird
PA0445	24216	N	Bird
PA0445	24224	N	Bird
PA0445	24292	N	Bird
PA0445	24297	N	Bird
PA0445	24300	N	Bird
PA0445	24355	N	Bird
PA0445	24356	N	Bird
PA0445	24382	N	Bird
PA0445	24386	N	Bird
PA0445	24403	N	Bird
PA0445	24439	N	Bird
PA0445	24463	N	Bird
PA0445	24464	N	Bird
PA0445	24466	N	Bird
PA0445	24530	N	Bird
PA0445	24569	N	Bird
PA0445	24609	N	Bird

Table A65 (continued)

Ecoregion code	Species ID	Endemic (Y/N)	Class
PA0445	24616	N	Bird
PA0445	24620	N	Bird
PA0445	24653	N	Bird
PA0445	24669	N	Bird
PA0445	24672	N	Bird
PA0445	25095	N	Bird
PA0445	25099	N	Bird
PA0445	25100	N	Bird
PA0445	25102	N	Bird
PA0445	25104	N	Bird
PA0445	25110	N	Bird
PA0445	25113	N	Bird
PA0445	25114	N	Bird
PA0445	25143	N	Bird
PA0445	25144	N	Bird
PA0445	25209	N	Bird
PA0445	25210	N	Bird
PA0445	25216	N	Bird
PA0445	25218	N	Bird
PA0445	25236	N	Bird
PA0445	25300	N	Bird
PA0445	25302	N	Bird
PA0445	25303	N	Bird
PA0445	25307	N	Bird
PA0445	25308	N	Bird
PA0445	25311	N	Bird
PA0445	25313	N	Bird
PA0445	25315	N	Bird
PA0445	25317	N	Bird
PA0445	25588	N	Bird
PA0445	25657	N	Bird
PA0445	25675	N	Bird
PA0445	25765	N	Bird
PA0445	25772	N	Bird
PA0445	25773	N	Bird
PA0445	25780	N	Bird
PA0445	25799	N	Bird
PA0445	25807	N	Bird
PA0445	25809	N	Bird
PA0445	25975	N	Bird
PA0445	25983	N	Bird
PA0445	25984	N	Bird
PA0445	26019	N	Bird
PA0445	26022	N	Bird
PA0445	26026	N	Bird

Table A65 (continued)

Ecoregion code	Species ID	Endemic (Y/N)	Class
PA0445	26040	N	Bird
PA0445	26050	N	Bird
PA0445	26355	N	Bird
PA0445	26357	N	Bird
PA0445	26359	N	Bird
PA0445	26362	N	Bird
PA0445	26406	N	Bird
PA0445	26411	N	Bird
PA0445	26430	N	Bird
PA0445	26432	N	Bird
PA0445	26433	N	Bird
PA0445	26434	N	Bird
PA0445	26452	N	Bird
PA0445	26477	N	Bird
PA0445	26652	N	Bird
PA0445	26653	N	Bird
PA0445	26654	N	Bird
PA0445	26656	N	Bird
PA0445	26662	N	Bird
PA0445	26689	N	Bird
PA0445	26690	N	Bird
PA0445	26695	N	Bird
PA0445	95645	N	Mammal
PA0445	95695	N	Mammal

[a] de baan et al. (2013b): https://doi.org/10.1021/es400592q
[b] species richness data for mammals, birds, reptiles, and amphibians from the WWF WildFinder database: https://www.worldwildlife.org/publications/wildfinder-database
[c] Species richness for plants (2900 species, average) is from SI file No.2 of G. Keir et al. (2005): https://doi.org/10.1111/j.1365-2699.2005.01272.x

10.6 Appendix 6: Supplementary data for Chapter 5, 'Quantifying the land use, energy requirements, and rates of water consumption of wild species in terrestrial ecosystems: A case study of the American kestrel (*Falco sparverius*) on St Martin Island, Caribbean'

In this appendix, the data supplementary to Chapter 5, 'Using 'ecological needs' to quantify the natural resource requirements of the American kestrel (Falco sparverius) on St Martin Island, Caribbean', is presented across. This appendix contains some but not all the supplementary data as most are too large to include as physical appendices. As with the previous appendices, these and all other supplementary data for Chapter 5 are available in the form of an online repository (see 'SuppData_Chapter5'):

- Mason, A. R. (2023). Appendix 6: Supplementary data for Chapter 5 in the thesis: 'Quantifying the natural resource requirements of terrestrial ecosystems for managing biodiversity alongside human development' [Data set]. Zenodo. https://doi.org/10.5281/zenodo.11217972

10.6.1 Estimated ecological needs of species comprising the kestrel food web on St Martin Island

Local species composition is taken from Goldwasser and Roughgarden (1993) [87]. The same species categories used by Goldwasser and Roughgarden are used. Estimates of adult body mass are made using species-specific data from the ecological literature, or where such data is unavailable, inferred from comparable species. The land use, and daily water consumption and energy requirement of each species is then calculated using the ecological needs equations proposed in Table 3.1, Chapter 3. These results, which underpin subsequent calculations in Chapter 5, are presented in Table A66, below.

Table A66 Land use, L_i (km^2 $individual^{-1}$), energy requirements, Ei (kJ $individual^{-1}$ day^{-1}), and rates of water consumption, Wi (m^3 $individual^{-1}$ day^{-1}), of individuals of each species comprising the kestrel food web on St Martin Island, Caribbean

Species	Class	Adult body mass (kg $individual^{-1}$)	Land use, L_i (km^2 $individual^{-1}$)	Water consumption, W_i (m^3 $individual^{-1}$ day^{-1})	Energy requirement, E_i (kJ $individual^{-1}$ day^{-1})
Anolis gingivinus	Reptile	$1x10^2$	0.00	No data	1.48
Anolis pogus	Reptile	$3x10^3$	0.00	No data	0.52
Adult spider	Invertebrate	$1x10^3$	0.00	No data	13.43
Ants	Invertebrate	$3x10^6$	0.00	No data	0.11
Centipede	Invertebrate	$2x10^4$	0.00	No data	3.52
Coleoptera (adult)	Invertebrate	$3x10^4$	0.00	No data	4.24
Coleoptera (larvae)	Invertebrate	$3x10^4$	0.00	No data	4.24
Diptera (adult)	Invertebrate	$1x10^5$	0.00	No data	0.34
Diptera (larvae)	Invertebrate	$1x10^5$	0.00	No data	0.39
Hemiptera	Invertebrate	$1x10^4$	0.00	No data	2.25
Homoptera	Invertebrate	$1x10^4$	0.00	No data	2.25
Isopoda	Invertebrate	$2x10^5$	0.00	No data	0.52
Isoptera	Invertebrate	$1x10^6$	0.00	No data	0.04
Juvenile spider	Invertebrate	$5x10^5$	0.00	No data	1.11
Lepidoptera (adult)	Invertebrate	$5x10^4$	0.00	No data	7.73
Lepidoptera (larvae)	Invertebrate	$1x10^3$	0.00	No data	13.43
Millipede	Invertebrate	$2x10^4$	0.00	No data	3.52
Orthoptera	Invertebrate	$3x10^4$	0.00	No data	4.93
Thysanoptera	Invertebrate	$1x10^6$	0.00	No data	0.04
Other hymenoptera	Invertebrate	$1x10^4$	0.00	No data	2.46
Bananquit	Bird	$1x10^2$	0.03	$3.41x10^6$	49.54
Bullfinch	Bird	$1x10^2$	0.03	$3.35x10^6$	48.76
Elaenia	Bird	$2x10^2$	0.03	$4.09x10^6$	58.69
Grassquit	Bird	$1x10^2$	0.03	$2.98x10^6$	43.63
Hummingbirds	Bird	$7x10^2$	0.02	$1.46x10^6$	22.35
Kestrel	Bird	$4x10^3$	0.319	$1.75x10^5$	351.84

10.6.2 Sankey diagram construction and the underlying data

The Sankey diagrams for daily energy use (Fig. 5.2, Chapter 5) and daily water consumption (Fig. 5.3, Chapter 5) are produced using 'Plotly' (v.5.17.0) in the 'Jupyter Notebook' software (v.7.0.4). The programming language 'Python' was used. The codes used to produce both Sankey diagrams are presented below: 'A.4.2.1. Code to produce 'daily energy' Sankey diagram (Fig. 5.2, Chapter 5)' and 'A.4.2.2. Code used to produce 'daily water consumption' Sankey diagram (Fig. 5.3, Chapter 5)'. These codes require the input of quantitative data on the energy requirements and water consumption of species within the trophic system. These input data are presented in 'A.4.2.2. Data underlying the Sankey diagrams of daily energy and water flows', within Tables A76 – A79.

10.6.2.1 Code to produce 'daily energy' Sankey diagram (Fig. 5.2, Chapter 5)

```
import plotly.graph_objects as go
import pandas as pd

# Load your CSV data
k1 = pd.read_csv("webdata.csv")

# Extract data from the CSV
label_node = k1["nodes"].dropna().tolist()
node_x = k1["node_x_h"].dropna().tolist()
node_y = k1["node_y_v"].dropna().tolist()
source = k1["from_sankey_index"].dropna().tolist()
target = k1["to_sankey_index"].dropna().tolist()
value = k1["quantity"].dropna().tolist()

# Increase the nodepad to adjust the height of the plot area
nodepad = 500  # Adjust this value to control the plot area height

# Create the Sankey diagram with increased nodepad
fig1 = go.Figure(go.Sankey(
    node=dict(
        thickness=20,
        label=label_node,
        x=node_x,
        y=node_y,
        pad=nodepad  # Adjust the nodepad as needed
    ),
    link=dict(
        source=source,
        target=target,
        value=value,
    )
))

# Set the layout properties
fig1.update_layout(
    title_text="Sankey Diagram",
    font_size=10,
)

# Show the diagram with the increased plot area height
fig1.show()
# Show the diagram in a larger window
fig1.show(renderer='browser')

#import plotly.io as pio

# Assuming you have a figure object named fig1
#file_path = r"C:\Users\arm319\OneDrive - Imperial College London\Kestrelweb.png"
# You can specify the full path if needed

# Save the plot as a high-quality PNG image
#pio.write_image(fig1, file_path, format="png")

import plotly.offline as pyo

# Assuming you have a figure object named fig1
file_path_html = r"C:\Users\arm319\OneDrive - Imperial College
London\Kestrelweb.html"

pyo.plot(fig1, filename=file_path_html, auto_open=False)
```

10.6.2.2 Code to produce ‘daily water consumption’ Sankey diagram (Fig. 5.3, Chapter 5)

```
import plotly.graph_objects as go
import pandas as pd

# Load your CSV data
k1 = pd.read_csv("sankey_water.csv")

# Extract data from the CSV
label_node = k1["nodes"].dropna().tolist()
node_x = k1["node_x_h"].dropna().tolist()
node_y = k1["node_y_v"].dropna().tolist()
source = k1["from_sankey_index"].dropna().tolist()
target = k1["to_sankey_index"].dropna().tolist()
value = k1["quantity"].dropna().tolist()

# Increase the nodepad to adjust the height of the plot area
nodepad = 500  # Adjust this value to control the plot area height

# Create the Sankey diagram with increased nodepad
fig1 = go.Figure(go.Sankey(
    node=dict(
        thickness=20,
        label=label_node,
        x=node_x,
        y=node_y,
        pad=nodepad  # Adjust the nodepad as needed
    ),
    link=dict(
        source=source,
        target=target,
        value=value,
    )
))

# Set the layout properties
fig1.update_layout(
    title_text="Sankey Diagram",
    font_size=10,
)

# Show the diagram with the increased plot area height
fig1.show()
# Show the diagram in a larger window
fig1.show(renderer='browser')

#import plotly.io as pio

# Assuming you have a figure object named fig1
#file_path = r"C:\Users\arm319\OneDrive - Imperial College London\Kestrelweb.png"
# You can specify the full path if needed

# Save the plot as a high-quality PNG image
#pio.write_image(fig1, file_path, format="png")

import plotly.offline as pyo

# Assuming you have a figure object named fig1
file_path_html = r"C:\Users\arm319\OneDrive - Imperial College
London\Kestrelwater.html"

pyo.plot(fig1, filename=file_path_html, auto_open=False)
```

10.6.2.3 Data underlying the Sankey diagrams of daily energy and water flows

Table A76 Node data input alongside the 'daily energy requirement' Sankey diagram code

nodes_index	nodes [a]
0	Vegetation (E_s = 1,574)
1	Isopoda (N = 226.5, E = 117.4)
2	Isoptera (N = 7,105, E = 516.2)
3	Homoptera (N = 15.3, E = 58.3)
4	Thysanoptera (N = 1,159, E = 84.2)
5	Coleoptera, larvae (N = 1.4, E = 10.1)
6	Diptera, adult (N = 215.5, E = 123.8)
7	Diptera, larvae (N = 223.7, E =146.0)
8	Lepidoptera, adult (N = 0.45, E = 5.90)
9	Lepidoptera, larvae (N = 6.3, E = 142.5)
10	Millipede (N = 0.48, E = 2.89)
11	Ants (N = 503.7, E = 91.3)
12	Other hymenoptera (N = 16.7, E = 69.6)
13	Orthoptera (N = 12.8, E = 84.0)
14	Juvenile spider (N = 8.6, E = 12.8)
15	Coleoptera, adult (N = 11.2, E = 63.2)
16	Hemiptera (N = 8.6, E = 25.8)
17	Centipede (N = 0.36, E = 1.71)
18	Adult spider (N = 2.6, E = 46.8)
19	Elaenia (N = 0.13, E = 12.2, W = 5.5E-7)
20	Hummingbirds (N = 0.63, E = 21.6, W = 9.1E-7)
21	Bullfinch (N = 0.18, E = 13.6, W = 6.1E-7)
22	Grassquit (N = 0.22, E = 14.5, W = 6.4E-7)
23	Bananquit (N = 0.18, E = 13.5, W = 6.0E-7)
24	Anolis pogus (N = 5.6, E = 3.8)
25	Anolis gingivinus (N = 6.8, E = 13.5)
26	Kestrel (N = 1.0, E = 645.3, W = 1.8E-5)

Table A77 Flows data used alongside the 'daily energy requirement' Sankey diagram code

process	from_sankey	from_sankey_index	to_sankey	to_sankey_index	quantity
primary_consumption	vegetation	0	isopoda	1	0.5
primary_consumption	vegetation	0	isoptera	2	0.1
primary_consumption	vegetation	0	homoptera	3	3.8
primary_consumption	vegetation	0	thysanoptera	4	0.1
primary_consumption	vegetation	0	coleoptera_larvae	5	7.2
primary_consumption	vegetation	0	diptera_adult	6	0.6
primary_consumption	vegetation	0	diptera_larvae	7	0.7
primary_consumption	vegetation	0	lepidoptera_adult	8	13.1
primary_consumption	vegetation	0	lepidoptera_larvae	9	22.8
primary_consumption	vegetation	0	millipede	10	6.0
primary_consumption	vegetation	0	ants	11	0.2
primary_consumption	vegetation	0	other_hymenoptera	12	4.2
primary_consumption	vegetation	0	orthoptera	13	6.3
primary_consumption	vegetation	0	coleoptera_adult	15	4.1
primary_consumption	vegetation	0	hemiptera	16	1.5
primary_consumption	vegetation	0	elaenia	19	74.0
primary_consumption	vegetation	0	hummingbirds	20	29.4
primary_consumption	vegetation	0	bullfinch	21	64.4
primary_consumption	vegetation	0	grassquit	22	55.9
primary_consumption	vegetation	0	bananquit	23	65.4
isopoda_consumption	isopoda	1	orthoptera	13	1.9
isopoda_consumption	isopoda	1	coleoptera_adult	15	1.7
isopoda_consumption	isopoda	1	centipede	17	1.3
isopoda_consumption	isopoda	1	bullfinch	21	20.9
isopoda_consumption	isopoda	1	anolis_pogus	24	0.0
isopoda_consumption	isopoda	1	anolis_gingivinus	25	0.0
isoptera_consumption	isoptera	2	orthoptera	13	3.0

Table A77 (continued)

process	from_sankey	from_sankey_index	to_sankey	to_sankey_index	quantity
isoptera_consumption	isoptera	2	coleoptera_adult	15	2.7
isoptera_consumption	isoptera	2	centipede	17	2.1
isoptera_consumption	isoptera	2	bullfinch	21	32.7
isoptera_consumption	isoptera	2	anolis_pogus	24	0.0
isoptera_consumption	isoptera	2	anolis_gingivinus	25	0.0
homoptera_consumption	homoptera	3	orthoptera	13	0.1
homoptera_consumption	homoptera	3	hemiptera	16	0.1
homoptera_consumption	homoptera	3	adult_spider	18	0.5
homoptera_consumption	homoptera	3	hummingbirds	20	4.7
homoptera_consumption	homoptera	3	grassquit	22	15.2
homoptera_consumption	homoptera	3	bananquit	23	2.1
homoptera_consumption	homoptera	3	anolis_pogus	24	0.1
homoptera_consumption	homoptera	3	anolis_gingivinus	25	0.1
thysanoptera_consumption	thysanoptera	4	hemiptera	16	0.5
thysanoptera_consumption	thysanoptera	4	adult_spider	18	2.6
thysanoptera_consumption	thysanoptera	4	bananquit	23	10.7
thysanoptera_consumption	thysanoptera	4	anolis_pogus	24	0.0
thysanoptera_consumption	thysanoptera	4	anolis_gingivinus	25	0.0
coleoptera_larvae_consumption	coleoptera_larvae	5	orthoptera	13	0.2
coleoptera_larvae_consumption	coleoptera_larvae	5	coleoptera_adult	15	0.1
coleoptera_larvae_consumption	coleoptera_larvae	5	centipede	17	0.1
coleoptera_larvae_consumption	coleoptera_larvae	5	bullfinch	21	1.6
coleoptera_larvae_consumption	coleoptera_larvae	5	anolis_pogus	24	0.0
coleoptera_larvae_consumption	coleoptera_larvae	5	anolis_gingivinus	25	0.0
diptera_adult_consumption	diptera_adult	6	hemiptera	16	0.9
diptera_adult_consumption	diptera_adult	6	adult_spider	18	5.3
diptera_adult_consumption	diptera_adult	6	hummingbirds	20	2.3

Table A77 (continued)

process	from_sankey	from_sankey_index	to_sankey	to_sankey_index	quantity
diptera_adult_consumption	diptera_adult	6	bananquit	23	21.7
diptera_adult_consumption	diptera_adult	6	anolis_pogus	24	0.1
diptera_adult_consumption	diptera_adult	6	anolis_gingivinus	25	0.0
diptera_larvae_consumption	diptera_larvae	7	orthoptera	13	1.3
diptera_larvae_consumption	diptera_larvae	7	coleoptera_adult	15	1.2
diptera_larvae_consumption	diptera_larvae	7	centipede	17	0.9
diptera_larvae_consumption	diptera_larvae	7	bullfinch	21	14.5
diptera_larvae_consumption	diptera_larvae	7	anolis_pogus	24	0.0
diptera_larvae_consumption	diptera_larvae	7	anolis_gingivinus	25	0.0
lepidoptera_adult_consumption	lepidoptera_adult	8	hemiptera	16	0.0
lepidoptera_adult_consumption	lepidoptera_adult	8	adult_spider	18	0.1
lepidoptera_adult_consumption	lepidoptera_adult	8	elaenia	19	3.1
lepidoptera_adult_consumption	lepidoptera_adult	8	bananquit	23	0.3
lepidoptera_adult_consumption	lepidoptera_adult	8	anolis_pogus	24	0.0
lepidoptera_adult_consumption	lepidoptera_adult	8	anolis_gingivinus	25	0.0
lepidoptera_larvae_consumption	lepidoptera_larvae	9	hemiptera	16	1.4
lepidoptera_larvae_consumption	lepidoptera_larvae	9	adult_spider	18	8.2
lepidoptera_larvae_consumption	lepidoptera_larvae	9	grassquit	22	36.6
lepidoptera_larvae_consumption	lepidoptera_larvae	9	bananquit	23	33.4
lepidoptera_larvae_consumption	lepidoptera_larvae	9	anolis_pogus	24	0.0
lepidoptera_larvae_consumption	lepidoptera_larvae	9	anolis_gingivinus	25	0.8
millipede_consumption	millipede	10	anolis_gingivinus	25	0.0
millipede_consumption	millipede	10	kestrel	26	1.8
ants_consumption	ants	11	orthoptera	13	0.2
ants_consumption	ants	11	juvenile_spider	14	1.5
ants_consumption	ants	11	centipede	17	0.1
ants_consumption	ants	11	hummingbirds	20	1.2
ants_consumption	ants	11	bullfinch	21	1.9

Table A77 (continued)

process	from_sankey	from_sankey_index	to_sankey	to_sankey_index	quantity
ants_consumption	ants	11	anolis_pogus	24	0.4
ants_consumption	ants	11	anolis_gingivinus	25	0.7
other_hymenoptera_consumption	other_hymenoptera	12	hemiptera	16	0.1
other_hymenoptera_consumption	other_hymenoptera	12	adult_spider	18	0.8
other_hymenoptera_consumption	other_hymenoptera	12	hummingbirds	20	20.7
other_hymenoptera_consumption	other_hymenoptera	12	bananquit	23	3.1
other_hymenoptera_consumption	other_hymenoptera	12	anolis_pogus	24	0.0
other_hymenoptera_consumption	other_hymenoptera	12	anolis_gingivinus	25	0.2
orthoptera_consumption	orthoptera	13	grassquit	22	6.1
orthoptera_consumption	orthoptera	13	anolis_pogus	24	0.0
orthoptera_consumption	orthoptera	13	anolis_gingivinus	25	0.0
orthoptera_consumption	orthoptera	13	kestrel	26	65.6
juvenile_spider_consumption	juvenile_spider	14	centipede	17	0.2
juvenile_spider_consumption	juvenile_spider	14	grassquit	22	6.1
juvenile_spider_consumption	juvenile_spider	14	anolis_pogus	24	0.0
juvenile_spider_consumption	juvenile_spider	14	anolis_gingivinus	25	0.0
coleoptera_adult_consumption	coleoptera_adult	15	elaenia	19	29.5
coleoptera_adult_consumption	coleoptera_adult	15	hummingbirds	20	0.3
coleoptera_adult_consumption	coleoptera_adult	15	anolis_pogus	24	0.0
coleoptera_adult_consumption	coleoptera_adult	15	anolis_gingivinus	25	0.1
coleoptera_adult_consumption	coleoptera_adult	15	kestrel	26	18.3
hemiptera_consumption	hemiptera	16	adult_spider	18	0.3
hemiptera_consumption	hemiptera	16	elaenia	19	14.8
hemiptera_consumption	hemiptera	16	bananquit	23	1.4
hemiptera_consumption	hemiptera	16	anolis_pogus	24	0.0
hemiptera_consumption	hemiptera	16	anolis_gingivinus	25	0.0
centipede_consumption	centipede	17	anolis_pogus	24	0.0
centipede_consumption	centipede	17	anolis_gingivinus	25	0.0

Table A77 (continued)

process	from_sankey	from_sankey_index	to_sankey	to_sankey_index	quantity
centipede_consumption	Centipede	17	kestrel	26	1.4
adult_spider_consumption	adult_spider	18	elaenia	19	38.7
adult_spider_consumption	adult_spider	18	hummingbirds	20	3.6
elaenia_consumption	elaenia	19	kestrel	26	13.8
hummingbirds_consumption	hummingbirds	20	kestrel	26	13.8
bullfinch_consumption	bullfinch	21	kestrel	26	13.8
grassquit_consumption	grassquit	22	kestrel	26	13.8
bananquit_consumption	bananquit	23	kestrel	26	13.8
anolis_pogus_consumption	anolis_pogus	24	anolis_gingivinus	25	0.0
anolis_pogus_consumption	anolis_pogus	24	kestrel	26	91.7
anolis_gingivinus_consumption	anolis_gingivinus	25	kestrel	26	366.9

Table A78 Nodes data input alongside the 'daily water consumption' Sankey diagram code

nodes_index	nodes
0	Water (Ws = 2.1E-5)
1	Isopoda
2	Isoptera
3	Homoptera
4	Thysanoptera
5	Coleoptera, larvae
6	Diptera, adult
7	Diptera, larvae
8	Lepidoptera, adult
9	Lepidoptera, larvae
10	Millipede
11	Ants
12	Other hymenoptera
13	Orthoptera
14	Juvenile spider
15	Coleoptera, adult
16	Hemiptera
17	Centipede
18	Adult spider
19	Elaenia
20	Hummingbirds
21	Bullfinch
22	Grassquit
23	Bananquit
24	*Anolis pogus*
25	*Anolis gingivinus*
26	Kestrel

Table A79 Flows data input alongside the 'daily water consumption' Sankey diagram code

process	from_sankey	from_sankey_index	to_sankey	to_sankey_index	quantity
Water consumption	Water	0	Elaenia	19	$5.50x10^7$
Water consumption	Water	0	Hummingbirds	20	$9.14x10^7$
Water consumption	Water	0	Bullfinch	21	$6.07x10^7$
Water consumption	Water	0	Grassquit	22	$6.43x10^7$
Water consumption	Water	0	Bananquit	23	$6.02x10^7$
Water consumption	Water	0	Kestrel	26	$1.75x10^5$