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Review

# Optimising UK urban road verge contributions to biodiversity and ecosystem services with cost-effective management

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ABSTRACT

Urban road verges can contain significant biodiversity, contribute to structural connectivity between other urban greenspaces, and due to their proximity to road traffic are well placed to provide ecosystem services. Using the UK as a case study we review and critically evaluate a broad range of evidence to assess how this considerable potential can be enhanced despite financial, contractual and public opinion constraints. Reduced mowing frequency and other alterations would enhance biodiversity, aesthetics and pollination services, whilst delivering costs savings and potentially being publically acceptable. Retaining mature trees and planting additional ones is favourable to residents and would enhance biodiversity, pollution and climate regulation, carbon storage, and stormwater management. Optimising these services requires improved selection of tree species, and creating a more diverse tree stock. Due to establishment costs additional tree planting and maintenance could benefit from payment for ecosystem service schemes. Verges could also provide areas for cultivation of biofuels and possibly food production. Maximising the contribution of verges to urban biodiversity and ecosystem services is economical and becoming an increasingly urgent priority as the road network expands and other urban greenspace is lost, requiring enhancement of existing greenspace to facilitate sustainable urban development.



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## 1. Introduction

The greenspace associated with urban roads, called 'tree lawn', 'parking strips' or 'sidewalk buffer' in North America, 'nature strips' in Australia, or 'road verges' in the UK and Europe, often consists of narrow strips of mown grassy vegetation, typically 2–5 m wide, with trees sometimes being present. Road verges (the term used throughout the rest of this manuscript) cover large areas of land, e.g. 800 km$^2$ in the Netherlands (2% of total land area – Schaffers, 2000), 135,400 km$^2$ in Finland (0.4% of land area – Saarinen et al., 2005), and 4928 million km$^2$ in USA (Forman et al., 2003); the spatial extent of road verges will increase further due to increased urbanisation (Seto et al., 2012) and a projected increase of 60% in the global road network from 2010 to 2050 (Laurance et al., 2014).

It is thus surprising that verges are often excluded from studies assessing the value of urban greenspace, especially as they can support considerable biodiversity (Parr and Way, 1988; Hausmann et al., 2016) and provide a wide range of ecosystem services (Säumel et al., 2016). These include regulating services such as pollinator support, carbon sequestration, air quality enhancement, local climate regulation, flood risk management and noise reduction; cultural services such as aesthetic, psychological, and safety benefits; and potential for provisioning services such as food and biofuel production (reviewed in Säumel et al., 2016). Street trees are a key aspect of many road verge ecosystem services with large contributions to carbon sequestration (Rogers et al., 2011; Nowak et al., 2013a) and pollution interception (Nowak et al., 2013b), although the herbaceous layer can also contribute to these services (Bouchard et al., 2013; Weber et al., 2014a). Verge vegetation can enhance temperature regulation through evapotranspiration (Shashua-Bar et al., 2011; Armson et al., 2013), improve aesthetics (Todorova et al., 2004; Blumentrath and Tveit, 2014), reduce driver stress (Antonson et al., 2009) and contribute to reduced flood risk and erosion control (Stovin et al., 2008; Mueller and Thompson, 2009). Verges contribute markedly to maintaining structural connectivity between urban greenspaces (Davies et al., 2014), and their location along transport routes and residential areas increases their importance for regulating pollution and providing aesthetic and other cultural services (Säumel et al., 2016).

Verges experience less habitat loss and their management is determined by fewer stakeholders compared to some other forms of urban greenspace, (cf. private gardens and their multiple owners). There are thus considerable opportunities to improve the contributions of urban verges to biodiversity and ecosystem services. We focus on the UK as a case study, although our conclusions apply to regions with similar verge management such as much of temperate Europe.

Current verge management primarily focuses on safety (e.g. providing sight lines and emergency stopping locations) and aesthetics, whilst minimizing costs (Parr and Way, 1988). This typically generates regularly mown short grass verges, although trees are frequently planted to improve aesthetics, provide shade and support biodiversity (Silvera Seamans, 2013). In the UK verges are the responsibility of the highway authority or local council who have a statutory duty to consider biodiversity conservation in all their activities. There is, however, no requirement to prioritise, or even equally weight, conservation relative to other considerations. Increasingly local authorities are contracting out roadside verge management to private companies for periods of 5–25 years (AMA Research, 2015), providing an opportunity for conservation sensitive management over ecologically relevant time periods - but also a potential constraint if ecologically damaging management is locked in for such time periods. There exist substantial opportunities to alter verge management in a manner that delivers enhanced biodiversity and ecosystem services for similar, or in some cases lower, costs than current management regimes.

## 2. Objectives and approach

Our primary objective is to review and critically evaluate the potential for enhancing road verge management for biodiversity and ecosystem service provision, whilst considering pragmatic constraints, trade-offs, and economic viability. To facilitate this we ascertained the relative impacts of alternative management options on the costs of verge management through discussions with the private company contracted to manage road verges in Sheffield, the 5th largest city in the UK. These discussions obtained initial information on the economic implications of alternative management regimes, but it is beyond the scope of this manuscript to attempt an analysis of full economic costings. This information was supplemented with data from the literature in the few cases where such information was available. Despite much interest in road ecology (e.g. Forman et al., 2003; Coffin, 2007), there has been insufficient evaluation of alternative management options for enhancing verges' biodiversity and ecosystem services whilst recognising the constraints associated with this habitat type. Our study thus addresses a crucial gap concerning improvement of road verge management in the UK and elsewhere.

## 3. Management options to enhance biodiversity and ecosystem service provision

Enhancement of ecosystem services may not protect biodiversity per se, but enhanced biodiversity can strengthen the long-term resilience of ecosystem function and service provision to environmental change (Oliver et al., 2015). Road verge management that enhances both biodiversity and ecosystem services is thus preferable to management that focuses exclusively on ecosystem services.

### *3.1. Pollinator support and biodiversity*

Urban verges in the UK are typically mown every 2–4 weeks with occasional use of glyphosate herbicide for spot-weeding. They thus rarely provide the floral and other resources needed by pollinators, although some verges can support diverse and abundant populations of bees and other pollinators (Noordijk et al., 2009; Skórka et al., 2013).

Biodiversity improvements could be delivered by reduced mowing frequencies, as demonstrated for insects on urban roundabouts (Helden and Leather, 2004). A two cut regime (early and late summer) was shown to be optimal for plant (Parr and Way, 1988) and animal (Noordijk et al., 2009) biodiversity in non-urban highway verges, but remains to be tested in an explicitly urban context. Partial cutting of verges, e.g. only mowing areas closest to the road, can maintain sight lines for motorists whilst still enabling a taller more biodiverse sward to develop (Valtonen et al., 2006). Rotational or mosaic cutting, i.e. dividing verges into two longitudinal strips that have a staggered cutting regime, can deliver similar biodiversity benefits and ensures continuous provision of floral resources (Noordijk et al., 2009). Additional enhancement for herbaceous plant biodiversity may be achieved by removal of cuttings (Parr and Way, 1988; Manninen et al., 2010). Maintenance and selection of appropriate tree species in verges will help to support pollinators, and selection of native tree species is likely to maximise resource provision for other invertebrates and their predators (Burghardt and Tallamy, 2013, Table 1). Whilst glyphosate is rapidly degraded by soil microorganisms (Busse et al., 2001), reducing its use may have environmental benefits (Vereecken, 2005; Helander et al., 2012).

Botanical biodiversity gains from altered mowing regimes rely on recovery from a persistent seed bank which can be enhanced by direct sowing or planting. Recommended seeding densities for wildflower seed mixes are lower than those for the typical UK Department of Transport approved grass seed mix so per unit area seed mix costs for these two options are similar (price data from www.phoenixamenity.co.uk [accessed August 2016], a UK seed supplier). However, overall costs are higher for wildflower seed mixes, due to the need for more regular reseeding compared to grass mixes, but verges dominated by short perennial wildflowers require less regular cutting than grass verges helping to off-set these establishment costs. Seed mixes should be carefully selected to maximise the provision of floral resources over the entire flowering season, and for as wide a range of pollinators as possible. Inclusion of the grass parasite *Rhinanthus minor* can reduce competition between grasses and forbs, enhancing biodiversity (Bullock and Pywell, 2005), whilst potentially further reducing required cutting frequencies, and thus management costs, due to reduced verge biomass growth (Ameloot et al., 2006, Table 2).

### 3.2. Carbon sequestration

Tress, particularly larger ones, store significant amounts of carbon, but this may not offset emissions arising from management during the tree's lifecycle, i.e. from nursery growth to ultimate removal (Nowak and Crane, 2002; McHale et al., 2007; Timilsina et al., 2014). Reducing management of urban street trees and allowing them to reach maximal growth potential will thus increase their contribution to carbon sequestration (Díaz-Porras et al., 2014). Large, fast-growing tree species that require lower maintenance and sequester more carbon (Nowak et al., 2002) should be prioritised, and planted in patches to reduce travel for maintenance and thus associated emissions and costs (McHale et al., 2007; Escobedo et al., 2011).

Carbon uptake is largely determined by growth rates and wood density, which vary considerably between species (Table 1). Selecting those with high wood densities rather than growth rates may maximise total life cycle carbon sequestration may be maximised by reducing management requirements associated with larger trees that can create a safety risk or cause infrastructure damage (Mullaney et al., 2015). Long-term carbon sequestration requires maintenance of a healthy tree stock making it necessary to consider species' resilience to future disease outbreaks and climate change. For example, Midland Hawthorn *Crataegus laevigata* has a high wood density but a low score for drought tolerance suggesting that this species may suffer during drought conditions that may become more severe with climate change. Increasing the diversity of the urban tree-stock will increase overall resistance to environmental change (Oliver et al., 2015) but the number of approved tree species for urban planting is often limited (Table 1). This is especially true for native tree species which typically have higher biodiversity value (64% of exotic tree species are in the lowest category for biodiversity value, cf. 13% of native species; Table 1).

### 3.3. Air quality

Roadside vegetation is particularly beneficial for enhancing air quality due to its proximity to traffic (Kardan et al., 2015; Weber et al., 2014a), which is the primary cause of air pollution in de-industrialised urban areas such as the UK (Tiwary and Colls, 2010). Urban trees, particularly street trees, deliver significant human health benefits by reducing pollutants, including ozone, nitrogen dioxide, sulphur dioxide and small particulates (Tallis et al., 2011; Nowak et al., 2013b). Species with rough, hairy or waxy leaves are particularly effective at trapping pollutants (Sæbø et al., 2012; Weber et al., 2014a). Evergreen conifers could provide year round pollution regulation, enhancing their value, but they are often sensitive to pollution (Sæbø et al., 2012) making them unsuitable for highly polluted sites. A structurally diverse and species rich herbaceous layer would enhance air quality benefits (Weber et al., 2014a) alongside careful tree species selection as there are trade-offs to consider concerning release of allergens or volatile organic compounds (VOCs) that can negatively affect air quality (Table 1).

### 3.4. Local climate regulation

Verges' contribution to mitigating large-scale urban heat islands is limited by their small size (Kong et al., 2014). They can regulate local climate, however, e.g. street trees in Manchester (UK) reduce building surface temperatures by 12 °C (Armson et al., 2013). Verges have low tree densities and the cooling benefits of an additional tree are greatest at such sites (Streiling and Matzarakis, 2003), highlighting the potential to improve ecosystem service delivery from increased tree planting in verges. Trees with broad and dense canopies provide the largest cooling effects, with Midland Hawthorn *C. laevigata* and Callery pear *Pyrus calleryana* found to be most effective of the five most common street tree species in Manchester (Armson et al., 2013). Low winter temperatures are a human health risk in temperate Europe and thus deciduous trees should be selected as shading effects from coniferous trees can limit temperatures during the winter (McPherson et al., 1988).

### 3.5. Flood risk and erosion control

Vegetation, especially trees, contributes to reduced flood risk and erosion control by absorbing run-off (or reducing flow rates in the case of herbaceous vegetation) and intercepting rainfall. Roots create additional soil pores and can break up compacted sub-soils increasing drainage and water holding capacity (Bartens et al., 2008; Stovin et al., 2008; Mueller and Thompson, 2009). The relative contribution of roadside verges to urban stormwater management can thus be enhanced by planting trees. Species with high water use requirements will be most effective at reducing flood risk, but are less drought resistant (Table 1), a potentially problematic scenario in many parts of Europe for future-proofing urban forestry against climate change (Roloff et al., 2009). There is much uncertainty surrounding rainfall predictions as although total precipitation may decrease, isolated heavy rainfall events may become more frequent – which highlights the importance of maximising road verges contribution to mitigating flood risk (Kendon et al., 2014).

### 3.6. Noise reduction and safety

Strips of densely planted roadside vegetation can reduce noise

**Table 1**
Relative value of tree species commonly planted in urban areas of Britain and Europe for key ecosystem services including biodiversity value. Scores are assigned from previously published datasets and for each performance measure (except drought tolerance and winter hardiness) are allocated into three approximately equal sized groups, albeit with some adjustments to the size of each group to take tied ranks into account, with +, ++ and +++ respectively indicating low, medium and high performance. For drought tolerance and winter hardiness +, ++ and +++ respectively indicate problematic or not very suitable species, suitable and very suitable species. Air quality regulation is assessed by tree species' net contribution to volatile organic compound (VOC) emissions (data from Donovan et al., 2005) and effectiveness in capturing particulate matter (PM) (data from Sæbø et al., 2012). Drought tolerance and winter hardiness are linked to climate change resilience, but note that high performance in drought tolerance trades-off against water uptake rates and thus flood alleviation (data from Roloff et al., 2009). Biodiversity value incorporates data from Alexander et al. (2006) on value for mycorrhizal fungi, foliage invertebrates (richness and biomass), leaf litter communities, pollinators, provision of fruits and seeds and epiphyte communities (data on value for rotten wood communities are excluded as rotten trees are removed from road verges). Performance in sequestering carbon is a function of growth rate (McHugh et al., 2015) and wood density (Tree Functional Attributes and Ecological Database, 2016) whereby faster growth rates and high wood densities are advantageous. Only a few species are currently used for planting in urban verges in the UK, and these include many that score poorly for biodiversity or ecosystem service values - those approved for use in Sheffield (UK) are marked with a * for use in narrow verges and tree pits and ** for use only in wider grass verges – the majority (60%) of which are not native to the UK.

| Species name | | Native Distribution | Air quality | | Drought tolerance | Winter Hardiness | Biodiversity value | Growth rates | Wood density |
|---|---|---|---|---|---|---|---|---|---|
| | | | PM | VOCs | | | | | |
| *Acer campestre*** | Field maple | Europe, N. Africa and W. Asia | ++ | + | +++ | +++ | ++ | + | +++ |
| *Acer platanoides* | Norway maple | Europe and W. Asia (not UK) | + | + | ++ | +++ | | +++ | ++ |
| *Acer pseudoplatanus* | Sycamore | Europe and W. Asia (not UK) | + | +++ | + | +++ | +++ | + | +++ |
| *Aesculus hippocastanum* | Horse chestnut | Europe (not UK) | ++ | | + | ++ | + | + | ++ |
| *Alnus cordata#* | Italian alder | Europe (not UK) | | ++ | ++ | ++ | | +++ | + |
| *Alnus glutinosa* | Alder | Europe, N. Africa and W. Asia | | + | + | ++ | ++ | ++ | + |
| *Alnus incana* | Grey alder | Northern temperate (not UK) | | ++ | +++ | +++ | | | + |
| *Betula ermanii** | Erman's birch | E. Asia | | | | | | | ++ |
| *Betula pendula*** | Silver birch | Europe and W. Asia | +++ | + | ++ | +++ | +++ | +++ | ++ |
| *Carpinus betulus*** | Common hornbeam | Europe and W. Asia | ++ | | ++ | +++ | + | + | +++ |
| *Castanea sativa* | Sweet Chestnut | Europe & Asia Minor (not UK) | | | ++ | ++ | + | | ++ |
| *Catalpa bignonioides*** | Indian Bean Tree | N. America | | | + | + | | | + |
| *Cedrus atlantica*** | Atlas Cedar | N. Africa | | | +++ | + | | | + |
| *Chamaecyparis lawsoniana* | Lawson cypress | N. America | | + | | | | | + |
| *Corylus colurna** | Hazel | Europe and W. Asia | | ++ | ++ | ++ | ++ | ++ | ++ |
| *Crataegus laevigata** | Midland Hawthorn | Europe | | | + | +++ | +++ | +++ | +++ |
| *Crataegus monogyna*** | Common hawthorn | Europe, N. Africa and W. Asia | | + | ++ | +++ | +++ | + | +++ |
| *Cupressocyparis leylandii* | Leyland cypress | N. America | | ++ | ++ | ++ | | +++ | |
| *Fagus sylvatica*** | Beech | Europe | + | | + | ++ | +++ | + | +++ |
| *Fraxinus excelsior* | Ash | Europe and W. Asia | + | + | ++ | ++ | ++ | ++ | +++ |
| *Ginkgo biloba** | Ginko | E. Asia | | | +++ | ++ | | | |
| *Gleditsia triacanthos** | Honey locust | Central and N. America | | | +++ | ++ | | | +++ |
| *Gymnocladus dioica*** | Kentucky coffeetree | N. America | | | ++ | ++ | | | ++ |
| *Ilex aquifolium* | Holly | Europe, N. Africa and W. Asia | | ++ | | | ++ | + | +++ |
| *Juglans nigra* | Walnut | N. America | ++ | | | | + | | ++ |
| *Larix decidua* | Larch | Europe (not UK) | | + | | | ++ | | ++ |
| *Liquidambar styraciflua** | Sweet gum | Central and N. America | | | ++ | + | | | ++ |
| *Liriodendron tulipifera*** | Tulip tree | N. America | | | + | ++ | | | + |
| *Malus** | Apple | Northern temperate | | ++ | + | ++ | +++ | + | +++ |
| *Parrotia persica*** | Persian ironwood | Central Asia | | | | | | | |
| *Picea abies* | Norway spruce | Europe (not UK) | | | | | ++ | | + |
| *Pinus nigra* | Black pine | Europe, N. Africa (not UK) | | + | +++ | +++ | | +++ | |
| *Pinus sylvestris*** | Scots pine | Europe and W. Asia | +++ | | +++ | +++ | ++ | +++ | + |
| *Platanus x hispanica** | London plane | Hybrid - N. America/E. Asia | | | +++ | ++ | + | ++ | +++ |
| *Populus tremula* | Aspen | Europe and Asia | ++ | +++ | ++ | +++ | + | +++ | + |
| *Prunus cerasifera* | Plum | Europe & Asia Minor (not UK) | | | +++ | ++ | +++ | | ++ |
| *Prunus x hillier** | Cherry hybrid | Hybrid - E. Asia/E. Asia | + | ++ | | | | | |
| *Prunus laurocerasus* | Laurel | N. America | | + | + | +++ | | | |
| *Prunus* spp. | Cherry tree spp. | Northern temperate | | | | | +++ | +++ | ++ |
| *Pterocarya fraxinifolia*** | Caucasian walnut | W. Asia | | | + | + | | | + |
| *Pyrus calleryana** | Callery pear | E. Asia | ++ | | +++ | ++ | | +++ | +++ |
| *Quercus cerris*** | Turkey Oak | Europe & Asia Minor (not UK) | | | +++ | ++ | + | | +++ |
| *Quercus ilex* | Holm Oak | Mediterranean Basin | | | | | + | | |
| *Quercus petraea* | Sessile oak | Europe and W. Asia | | +++ | ++ | ++ | +++ | | +++ |
| *Quercus robur*** | English Oak | Europe and W. Asia | ++ | +++ | + | +++ | +++ | ++ | +++ |
| *Quercus rubra* | Red oak | N. America | | +++ | ++ | ++ | | | ++ |
| *Robinia pseudoacacia* var. *frisia* | False acacia | N. America | + | | + | +++ | + | | +++ |
| *Salix alba* | White willow | Europe and Asia | | +++ | + | +++ | +++ | ++ | + |
| *Salix caprea* | Goat willow | Europe and Asia | | +++ | ++ | +++ | +++ | ++ | + |
| *Salix fragilis* | Crack willow | Europe and W. Asia | +++ | +++ | | | +++ | ++ | + |

(*continued on next page*)

**Table 1** (*continued*)

| Species name | | Native Distribution | Air quality | | Drought tolerance | Winter Hardiness | Biodiversity value | Growth rates | Wood density |
|---|---|---|---|---|---|---|---|---|---|
| | | | PM | VOCs | | | | | |
| *Sambucus nigra* | Elder | Europe and W. Asia | | ++ | | | | + | ++ |
| *Sequoiadendron giganteum*** | Giant redwood | N. America | | | | | | | + |
| *Sorbus aucuparia* | Mountain ash | Europe, N. Africa and W. Asia | | ++ | + | +++ | ++ | ++ | ++ |
| *Sorbus intermedia 'Brouwers'* | Swedish whitebeam | Northern Europe (not UK) | ++ | | ++ | +++ | | ++ | ++ |
| *Sorbus x arnoldiana** | Rowan hybrid | Hybrid - East Asia/ Europe, N Africa & W. Asia | | | + | +++ | | ++ | ++ |
| *Syringa vulgaris* | Lilac | Europe (not UK) | | +++ | ++ | +++ | | + | |
| *Taxus baccata*** | Yew | Europe, N. Africa and W. Asia | ++ | | | | + | + | ++ |
| *Tilia* | Lime spp. | Northern temperate | | ++ | | | ++ | ++ | + |
| *Tilia cordata 'Winter Orange'** | Small leaved lime variety | Europe and W. Asia | + | | ++ | +++ | ++ | | + |
| *Tilia cordata x mongolica** | Lime hybrid | Hybrid - East Asia/ Europe and W. Asia | | | | | | ++ | + |
| *Ulmus 'New Horizon'*** | Elm Hybrid | Hybrid - E. Asia/E. Asia | ++ | ++ | | | | + | +++ |

pollution, but typically only by a modest 2–3 dB (Heisler, 1975; Harris and Cohn, 1985), although the mere presence of vegetation that blocks views of traffic can enhance perceived noise reduction (Harris and Cohn, 1985). Dense shrubs also improve road safety by absorbing collision impacts more slowly than metal barriers (Huang, 1987), and reducing headlight glare and thus accident rates (van der Heijden and Martens, 1982).

### 3.7. Provisioning services

Urban agriculture can contribute to improving food security, reducing greenhouse gas emissions (by reducing food miles) and providing social benefits, with verges contributing through community programs or unofficial "guerrilla" gardening (Hunter and Brown, 2012; Adams and Hardman, 2013). These contributions are currently negligible in the UK, partly due to food safety concerns regarding contamination by pollutants (Clarke et al., 2015). This may be less of a risk than typically perceived (Brown et al., 2016). Fruit and nuts harvested from street trees are likely to be safe for consumption (von Hoffen and Säumel, 2014) and, although food crops grown in road verge soils may exceed critical pollutant safety values (Säumel et al., 2012), cultivated urban soils show lower levels of contamination than uncultivated soils (Clarke et al., 2015) and the erection of barriers, either artificial or planted, can greatly reduce pollutant loads (Säumel et al., 2012). Local authorities have been encouraged to support local community "guerrilla" gardening (Adams and Hardman, 2013) due to their strong social benefits (Crane et al., 2012; Anderson, 2014) allowing people to engage with their local community, enhancing cultural ecosystem service provision. In addition, cuttings from grass verges could be used for compost although their use may be limited due to high concentrations of heavy metals in the vegetation (Kalavrouziotis and Koukoulakis, 2009).

Some studies have examined the potential for road verges in biofuel production from grass cuttings (Meyer et al., 2014) or short rotation *Salix* or *Populus* coppice (SRC) (Voinov et al., 2015). Energy Return on Energy Invested (EROEI) values vary from 2 to 3 for biogas from grass (Meyer et al., 2014), to 17–33 and 18–42 respectively for electricity and heat cogeneration from grass and *Salix* (Voinov et al., 2015). These estimates compare favourably with those from other biofuels, and alternative energy sources including wind (15–18) and nuclear (5–15) (Hall et al., 2014; Voinov et al., 2015). Roadside vegetation may be contaminated with pollutants but safety risks from burning are generally low and within EU legislative limits (Meyer et al., 2014). Additional benefits of SRC plantations may include positive impacts for biodiversity support, carbon sequestration, and stormwater management (Rowe et al., 2009). Notably, SRC often focuses on species of willow *Salix*, which have high performance scores for regulating air pollution and supporting biodiversity; whilst *Salix* has relatively low drought tolerance thus increasing the potential to mitigate flood risk (Table 1). Not all urban verges are suitable for SRC, as drivers' sight lines must be maintained, and their fragmented nature increases transportation costs, but efficiencies arise due to easy access for harvesting, and because production occurs where products are used (Esteban et al., 2014). The viability of using verges for SRC is further enhanced as their large edge to interior ratio is ideal for SRC (Rowe et al., 2009), although field trials in an urban context have yet to be published.

### 3.8. Cultural ecosystem services

Verges do not provide large recreational areas, but urban residents typically have daily exposure to them, suggesting that they contribute to the numerous social and well-being benefits derived from green-space and additional indirect benefits such as enhanced property values (Säumel et al., 2016). Verges probably contribute to these cultural ecosystem services, but the magnitude of this contribution is uncertain. Good maintenance (e.g. removal of litter) and naturalness are two key factors of a roadside verge that contribute to its aesthetic qualities (Chon and Shafer, 2009; Mell et al., 2013; Blumentrath and Tveit, 2014). The presence of trees and a diverse and colourful herbaceous layer that supports additional biodiversity is likely to further enhance verges' contributions to cultural ecosystem services (Todorova et al., 2004; Säumel et al., 2016; Southon et al., 2017).

## 4. Discussion

Enhancing the biodiversity and ecosystem service provision of urban verges is subject to three key management considerations: 1) cost, 2) constraints imposed by contracts between private service providers and local authorities, and 3) residents' perceptions.

### 4.1. Cost constraints

Reduced mowing frequencies would deliver immediate savings; establishing wildflower populations or SRC for biofuel may provide downstream savings despite substantial upfront investment costs (Table 2). Increased tree planting is constrained by establishment

**Table 2**
The impact of different roadside verge management options on biodiversity and ecosystem service provision together with expected change in costs compared with common practices. *Key*: ↘: decreased management cost; ↗: increased management cost; –: no major change in management cost.

| Management option | Cost upfront | Cost maintenance | Effect on ecosystem services and biodiversity | Comments |
|---|---|---|---|---|
| Mowing – reduced frequencies | – | ↘ | Enhanced biodiversity<br>Increased pollinator numbers<br>Enhanced aesthetics from increased floristic diversity<br>Potential for reduced aesthetics if taller vegetation gathers litter or appears untidy | Verges that are cut twice per year show the highest biodiversity for plants and invertebrates (see section 3.1). |
| Mowing – altered timings for verges currently receiving two cuts per year | – | – | Enhanced biodiversity<br>Increased pollinator numbers<br>Enhanced aesthetics from increased floristic diversity | Cuts in early and late summer (i.e. May/June and August) show enhanced plant and invertebrate biodiversity (see section 3.1). Maintenance costs likely to remain static if the proposed timings of cut do not require extra vehicles. |
| Mowing – mosaic cutting | – | ↗ | Enhanced invertebrate biodiversity<br>Increased pollinator numbers<br>Potential for reduced aesthetics if vegetation strips lead to uneven appearance | Plant diversity would not be affected but the continued presence of flowers in the different strips would increase availability of nectar sources and benefit insect diversity and abundance (see section 3.1). A staggered cutting regime will likely increase maintenance costs since sites would need to be visited twice as often. |
| Mowing – partial cutting | – | ↘ | Enhanced biodiversity<br>Increased pollinator numbers<br>Enhanced aesthetics from increased floristic diversity and enhanced signs of care | Frequent mowing of first metre nearest road would maintain sight lines. Less frequent mowing of area further back would enhance plant diversity and insect diversity (see section 3.1). Reduced maintenance costs due to reduced cutting area, and magnitude of savings will increase with verge width. |
| Removal of vegetation cuttings from grass verges | ↗ | ↗ | Enhanced biodiversity<br>Potential compost source for brownfield sites | Removal of cuttings will enhance roadside verge biodiversity of plants and animals (section 3.1). Upfront and maintenance costs expected to increase due to the need for collection and disposal. This may be offset to some extent by composting cuttings although use may be limited due to contamination (section 3.7). |
| Decreased herbicide use | ↘ | ↗ | Reduced possibility of contamination of nearby ecosystems | Glyphosate should not be used if heavy rain forecast due to increased mobility leading to contamination of aquatic ecosystems (section 3.1). Decreased herbicide use will lead to some savings although maintenance costs may increase if extra labour required to control vegetation at points where mowers cannot reach. |
| Wildflower seed mix | ↗ | ↘? | Enhanced biodiversity<br>Enhanced aesthetics | Presence of wildflower species can support insect biodiversity. Upfront costs would involve purchase of seed mix and site preparation. Maintenance costs may decrease if savings from reduced mowing completely offset reseeding costs (section 3.1). |
| Establishment of Yellow Rattle (*Rhinanthus minor*) | ↗ | ↘? | Enhanced biodiversity<br>Enhanced aesthetics arising from floristic changes | High density sowing required (at least 1000 seeds per $m^2$) for successful establishment (Ameloot et al., 2006). As well as enhancing biodiversity, reduced biomass production may lead to reduced mowing frequencies and reduced maintenance costs (section 3.1). |
| Tree planting | ↗ | ↗ | Carbon sequestration<br>Pollution interception<br>Local climate regulation<br>Noise interception<br>Stormwater management<br>Improved aesthetic quality<br>Enhanced biodiversity | An urban tree planting program would involve large upfront and some increased maintenance costs but potentially contribute to savings in other areas due to provision of a wide range of ecosystem services. Urban trees have been demonstrated to contribute to support for local biodiversity (section 3.1); enhanced carbon sequestration (section 3.2); enhanced local air quality and removal of PM, particularly if planting is targeted in high traffic pollution areas such as roundabouts (section 3.3); mitigation against urban heat island effect due to increased evaporative cooling form leaf transpiration (section 3.4); enhanced stormwater management through interception of rainfall and water uptake by roots, thereby increasing the capacity of the soils to store water (section 3.5); noise interception when planted in dense strips (section 3.6); and |

(*continued on next page*)

**Table 2** (*continued* )

| Management option | Cost upfront | Cost maintenance | Effect on ecosystem services and biodiversity | Comments |
|---|---|---|---|---|
| | | | | aesthetic improvements and enhanced psychological well-being (section 3.8). The magnitude of benefits varies between species and there are trade-offs between different services (Table 1). |
| Urban Agriculture – vegetable crops | – | ↘ | Food production<br>Improved community engagement | Care must be taken to select suitable sites for health and safety reasons with respect to potential risks from passing traffic and any buried cables. In addition, sites in high pollution areas (i.e. high traffic densities) should be avoided so that any food produced is suitable for consumption (section 3.7). Root crops such as carrot or potato are not suitable due to high uptake of lead in contaminated soils by these plants (Clarke et al., 2015). Fruit trees are less likely to produce contaminated food alongside additional ecosystem service benefits associated with urban trees generally (von Hoffen and Säumel, 2014).<br>Urban agriculture has been identified as a successful way to increase community engagement (section 3.7).<br>Costs expected to decrease if local community groups take over management of verges for food production. Planting and maintenance of fruit trees, however, would involve additional upfront and maintenance costs. |
| Urban Agriculture – fruit trees | ↗ | ↗ | Food production<br>*Note that fruit trees will also provide the additional benefits associated with tree planting* | |
| Coppice planting | ↗ | ↘ | Fossil fuel replacement<br>Additional benefits associated with tree planting will also arise, but will be influenced by the duration of the rotation cycle. | Some roadside verges may be suitable sites for short rotation coppice biofuel production, with significant carbon benefits through fossil fuel replacement (section 3.7). SRC establishment would involve large upfront costs and costs associated with harvesting every 3–5 years (Rowe et al., 2009). Net maintenance costs expected to fall due to no requirement for regular mowing and revenue from selling the final product.<br>SRC usually uses *Salix* species which have high biodiversity value and perform well at regulating air pollution. |

and maintenance costs yet over long time periods can be cost effective when considering carbon sequestration, reduced air pollution and cooling benefits (McPherson et al., 1998; Sunderland et al., 2012; Kroeger et al., 2014), with annual net benefits estimated to be US$18–38 per tree for five cities across the US (McPherson et al., 2005). With the exceptions of reduced mowing frequencies and SRC these financial benefits are, however, unlikely to be received by organizations conducting the management, which is a particular problem if aspects of urban management are privatized. Payment for Ecosystem Services (PES) schemes could help incentivise maintenance, improved selection, diversification and expansion of the roadside tree stock (Farley and Costanza, 2010).

### 4.2. Contractual constraints

The contracting of verge management to private companies, combined with the current format of such contracts, creates additional financial constraints as companies risk large fines if strict performance criteria (e.g. litter free verges) are not met. This promotes low quality short grassy verges over structurally diverse vegetation types, such as shrubs, that are more likely to trap litter (Nasar and Fisher, 1993). Conversely, private companies may be more likely to experiment with alternative management regimes in the interests of long-term cost savings, and often have greater potential to invest in additional equipment (e.g. for SRC) than public authorities. There is also a need to re-assess the performance criteria embedded within these contracts. In the UK verge management contracts with private companies often refer to tree numbers, without any reference to size distributions or total biomass. The magnitude of many ecosystem services provided by trees (including carbon storage, regulation of air quality, and cultural ecosystem services) scales with tree size or biomass (McPherson et al., 1997; Roy et al., 2012). The typical current contractual performance criteria thus do not promote maintenance of current levels of ecosystem service provision as they enable large trees to be felled (for safety or maintenance cost reasons) and replaced with an equal number of small trees that provide a fraction of the ecosystem services formerly provided. This is counter to the off-setting principle of ensuring no net loss of biodiversity or ecosystem services which explicitly establishes off-setting ratios that recognise the smaller contributions of recently created habitats and young trees (Escobedo et al., 2011; Bull et al., 2013).

### 4.3. Public perception constraints

Public perception is crucial as altered management may challenge people's 'sense of place', i.e. attachment to a neighbourhood's current state (Shamai, 1991). Social norms are a key factor influencing residents' preference for different street designs i.e. ecologically beneficial management is preferred if a neighbour has already demonstrated approval (Nassauer et al., 2009). Consequently, engaging the entire neighbourhood, in particular key community activists, may be needed to maximise the probability of widespread approval for any proposed changes to verge management. Explaining the ecological value of altered management is likely to further increase public approval (Garbuzov et al., 2014; Southon et al., 2017), but there may still be a limit to the vegetation types that people are willing to tolerate on verges close to their homes. That said, urban dwellers are often willing to pay higher taxes for investment in green infrastructure (Mell et al., 2013), and surveys of people using motorways and A-roads suggest that people frequently favour less intensively managed verges (Akbar et al., 2003). However, there are still likely to be limits to what local people will be willing to tolerate, especially in residential settings. Whilst wild urban vegetation attracts strongly negative reactions for some urban residents (Weber et al., 2014b) typically people prefer meadow vegetation to mown-amenity grassland, especially when it is species rich (Southon et al., 2017).

### 4.4. Prioritisation and trade-offs

There are numerous alternative solutions for improving urban verge management for enhancing ecosystem services and biodiversity (Table 2), but it is not possible to maximise all ecosystem services and biodiversity in any one location because of trade-offs. Choosing between these alternatives can be informed by prioritising particular benefits in particular locations, whilst considering the net benefits across the entire road verge network. Alternative management options will often consist of choices regarding which tree species to plant, as no single species maximises all ecosystem services whilst also having high biodiversity value (Table 1). This variation in contributions to individual ecosystem services provides a rationale for diverse tree plantings in locations where multiple services need to be prioritised. In other locations, there may be a clear need to focus on maximising the provision of one ecosystem service, even at the cost of reduced provision of alternative services. Such scenarios would include selecting species that maximise air quality benefits on verges where traffic causes significant air pollution, or selecting species that maximise aesthetic or climate regulation services in heavily paved city centre locations that currently have low aesthetic value or high summer temperatures.

Despite the disproportionate contribution of street trees to many ecosystem services not all verges are able to support street trees due to their small size, safety considerations, or proximity of buildings or other infrastructure. Thus maximising the contribution of the herbaceous layer to ecosystem service provision should be considered in these situations. There may be some situations, such as along major gateways and transport corridors, where aesthetic values need to be prioritised, although significant changes in verge appearance close to peoples' homes may be resisted by local residents. In these situations the most feasible options for alternative management of the herbaceous layer will be those that do not compromise aesthetic value (see Table 2). Despite these ecological and contextual constraints our overall assessment of the benefits and associated costs of alternative verge management options (Table 2) indicate that for most urban verges there are feasible alternative management options that could be used to increase urban verges contributions to biodiversity and provision of ecosystem services.

## 5. Conclusions

Roadside verges are frequently overlooked and given insufficient attention in discussions of urban biodiversity and ecosystem services. Their spatial extent and configuration throughout the urban matrix, limited alternative land uses, and proximity to traffic (the dominant source of pollution in many cities) means that urban road-verges could make a significant contribution to enhancing urban biodiversity and ecosystem service provision. Currently these contributions are limited by management that sustains the dominance of a low diversity grass sward. Alternative management options are available that would maximise these contributions e.g. altered mowing regimes, use of biodiverse and forb rich mixes, planting tree species that are selected to maximise priority ecosystem services for a particular location, retention of larger trees, and increased tree species diversity. These changes can be cost-effective, although the time-scale over which this occurs varies considerably from almost immediately (e.g. reduced mowing frequencies) and the medium term (e.g. use of verges for growing bio-fuel), to much longer time scales that may also require the implementation of payment for ecosystem service schemes. Enhancing the biodiversity and ecosystem service provision of urban road verges will be facilitated by changing contractual arrangements with private companies that manage urban infrastructure so that they explicitly recognise and prioritise biodiversity and provision of ecosystem services. Verge management also needs to consider the heterogeneity of the network to ensure overall performance (and not that of just a few focal sites) is optimised. Despite these constraints, cost-effective management strategies are available that enhance the ability of urban verges to support biodiversity and ecosystem services. As other forms of urban greenspace continue to be lost from towns and cities it is increasingly important to optimise benefits from road verges.

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