The whole life carbon footprint of green infrastructure: A call for integration

L'empreinte carbone dans tout le cycle de vie des infrastructures vertes : un appel à l'intégration

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RÉSUMÉ
La législation environnementale de l’eau étant de plus en plus stricte, il est nécessaire de repenser l’infrastructure de drainage. A cet effet, la réduction des gaz à effet de serre et l’amélioration des normes environnementales de qualité de l’eau constituent le principal défi. L'infrastructure verte est une solution à faible émission de carbone pour la gestion des eaux pluviales qui peut contribuer à améliorer significativement cette situation. Cet article reprend huit études qui évaluent l'empreinte carbone dans le cycle de vie de l'infrastructure verte ; il examine également les points communs à ces huit études ainsi que les limites des solutions proposées. Il est soutenu que les évaluations actuelles doivent être remplacées par des approches qui intègrent des objectifs divergents (par exemple, le carbone et la qualité de l’eau) et considèrent le système de drainage urbain dans son intégralité ; ceci dans le but de comprendre et d'améliorer la gestion des eaux pluviales.

ABSTRACT
With increasingly stringent environmental legislation affecting the water sector, a rethink of drainage infrastructure is needed. In this context, reducing greenhouse gas emissions and aiming at higher environmental water quality standards represents a main challenge. Green infrastructure (GI) has been deemed a low-carbon solution for stormwater management that can importantly contribute to improve this situation. This paper reviews eight studies assessing the whole life carbon footprint of GI and discusses common trends and limitations. It is argued that integrated approaches that incorporate a broader variety of conflicting objectives (e.g. carbon and water quality) and consider the urban drainage system on its entirety should replace current assessments in order to ensure that opportunities to understand and improve urban stormwater management are not missed in practice.

KEYWORDS
1 INTRODUCTION

As a consequence of the commitment of EU members to cut greenhouse gas (GHG) emissions under the Kyoto protocol, legislation aiming to enforce the agreed emission targets has been implemented at European (EU Emissions Trading System) and national level. In the UK, for example, the Carbon Reduction Commitment (CRC), subject to the Climate Change Act, targets an 80% reduction of GHG emissions by 2050 and 34% by 2020 in large public and private organisations (Defra, 2008).

Another piece of European environmental legislation, the Water Framework Directive (WFD) (European Commission, 2000), takes a new viewpoint towards the management of urban water pollution, encouraging a holistic approach to deal with urban wastewaters in opposition to emission-based strategies. This integral approach stresses on the importance of the condition of receiving waters to set pollution control strategies based on increasingly stringent environmental quality standards (Rauch et al., 2005).

The water industry in the UK must comply with both legislations and thus find ways to reduce its carbon footprint while enhancing the condition of receiving water quality derived from its activities. Achieving such targets in a highly energy-dependent industry (in the UK, 70% of the water industry's carbon footprint is associated with operational carbon in the form of power from the national grid (Palmer, 2010)) would involve disproportionate costs (Environment Agency, 2009a). Indeed, the intense water-energy nexus currently constrains to a great extent the industry’s capacity to meet GHG emission targets; being of especial importance for wastewater management operations, which account for 56% of the water industry energy use (Environment Agency, 2008).

According to Ainger et al. (2009), decarbonising the water industry should follow a multi-faceted strategy, including: decarbonisation of the electricity industry, exploitation of renewables within the water sector, use of sustainable drainage systems (SuDS), encouragement of water efficiency among consumers, development of risk-based decision-making that balance water quality against carbon mitigation, reduction of levels of pollutants at source, decentralisation of water supply, development of new technologies, effective policy change, and development of financial incentives.

The use of SuDS and similar techniques (e.g. BMPs, WSUD, LID), which nowadays tend to be aggregated under the term ‘green infrastructure’ (GI), intend to mitigate the drainage problems associated with urban development by mimicking the characteristics of the natural catchment (e.g. increasing permeable surfaces, slowing down runoff, etc.) GI has been deemed a low-energy solution for stormwater management which can therefore play a significant role in reducing the carbon footprint of urban drainage systems (and of the urban water cycle as a whole). Further, the multifunctional character of GI could potentially bring additional benefits (health and well-being, sense of place and community, amenity and recreation, biodiversity, climate change adaptation and mitigation, land management, property value, etc.) that go beyond mere stormwater objectives, such as flood mitigation (Ashley et al., 2011). This becomes particularly important as drainage solutions must increasingly satisfy a larger and more complex number of environmental, social and economic criteria that aim to deliver sustainable water management. In this sense, retrofit GI (the practice of introducing GI in already developed areas, as opposed to GI implemented as part of new developments) has been particularly encouraged in order to enhance the condition of impervious inner areas of city centres, on which drainage infrastructure was constructed to fulfil a more limited set of objectives (Brown et al., 2008; Digman et al., 2012).

Despite the potential of GI to deliver multifunctional infrastructure that serves a broader variety of purposes, practice in the UK and elsewhere has focused in assessing the performance of GI in terms of flood control, effluent quality and cost, while evidence on GHG emissions mitigation and other benefits are still in their infancy. In the UK, this is partly a consequence of the existing regulatory environment in the water sector, which currently neglects the implications that water management issues have at the system scale and beyond the periodic asset investment time frame. Ignoring all these will surely mislead decisions on the effectiveness of green and grey infrastructure, resulting in the unsustainability of solutions overall.

The aim of this paper is to identify approaches to the evaluation of the whole life carbon footprint of GI and discuss their limitations and opportunities for improvement in the context of asset investment decision making within the water industry.
2 THE WHOLE LIFE CARBON OF GREEN INFRASTRUCTURE: A REVIEW

The evaluation of energy use, carbon footprint and greenhouse gas emissions derived from activities within the water sector is not new and many examples can be found in the literature examining these for water distribution, wastewater networks and treatment plants (Environment Agency, 2008b; Corominas et al., 2009; Stokes and Horvath, 2009; Mo et al., 2010; Hall et al., 2011; Porro et al., 2011). However, there are a limited number of studies looking at the contribution of GI to carbon emissions. This section reviews eight of these studies and presents their major findings and characteristics, which are subsequently analysed and discussed within the following sections. Table 1 presents the relationship between elements and terms frequently used within the reviewed studies, particularly in those that carried out a life cycle assessment (LCA) of GI options.

One of the first LCA applied to BMPs was reported by Kirk (2006) under the Redesigning the American Neighborhood (RAN) Project. This ‘cradle-to-grave’ analysis evaluated the long-term (30-year life cycle) environmental and human health impacts of a number of BMPs (subsurface treatment and storage, wet pond, bioretention cell, and sub-surface flow gravel wetland) installed at the University of New Hampshire’s Stormwater Centre facilities (treating runoff from a parking lot). The carbon footprint of each BMP was estimated as part of the assessment, finding rather similar profiles, with the bioretention cell and wetland being the least carbon intensive. The complete assessment included a variety of indicators from the USEPA TRACI model (acidification, air pollutants, eco-toxicity, eutrophication, human health, etc.), whose values were subsequently normalised, weighted and annualised for comparison. The results of the study showed that the most advantageous system was the gravel wetland, followed by the bioretention cell, the wet pond and the subsurface system.

Table 1: Summary of the elements and activities commonly considered for the life cycle assessment (LCA) of the carbon footprint associated with green infrastructure (GI) for different LCA configurations.

<table>
<thead>
<tr>
<th>GI life cycle stages</th>
<th>MATERIALS</th>
<th>CONSTRUCTION</th>
<th>OPERATION</th>
<th>DECOMMISSION</th>
</tr>
</thead>
<tbody>
<tr>
<td>Opportunities to reduce the carbon footprint of GI</td>
<td>Sourcing of materials (local manufacturers, eco materials, etc.)</td>
<td>Low-carbon construction methods and design.</td>
<td>Low-maintenance design. Carbon sequestration. Avoided emissions from new grey infrastructure. Reduced pumping and treatment of combined sewer flows. Energy benefits from flooding or diffuse pollution mitigation. Reduced heat island effect, enhanced building insulation, etc.</td>
<td>Easy-to-remove design. Reusable/recyclable materials. Adaptable design.</td>
</tr>
<tr>
<td>Carbon category</td>
<td>Embodied Carbon</td>
<td>Operational Carbon</td>
<td>End-of-life Carbon</td>
<td></td>
</tr>
<tr>
<td>LCA boundaries</td>
<td>Cradle-to-site</td>
<td>Cradle-to-built asset</td>
<td>Whole life cycle carbon assessment (UKWIR approach)</td>
<td>Cradle-to-grave</td>
</tr>
</tbody>
</table>

In a similar study carried out by Flynn (2011), two GI systems representative of the Philadelphia area (rain garden and a green roof system) were compared in terms of their environmental and economic impacts throughout a 30-year life cycle (including construction, operation and decommission). Additionally, the study quantified life cycle avoided emissions arising from stormwater management benefits, carbon sequestration and building energy savings. Although the green roof scored higher in the operational phase, the lower construction and decommissioning impacts of the rain garden
ensured a far lower life cycle impact (including compensation of life cycle carbon emissions).

Kosareo and Ries (2007) evaluated the life cycle environmental impact (including global warming, in terms of CO₂ emissions) and benefit (i.e. building energy use reduction) of intensive and extensive green roofs as compared to a conventional ballasted roof in a Pittsburgh case study. A dimensionless overall impact score was calculated using LCIA Impact 2002+ (weighting the impacts derived from human health, ecosystem quality, climate change, and resources depletion). Their results reflect the better performance of intensive roof and extensive roofs, for both carbon footprint (around half of that in conventional roofing) and aggregated scores.

Taylor and Barrett (2008) quantified the ‘carbon signature’ of a number of surface drainage techniques (drain inlet insert, hydrodynamic separator, wet pond, sand filter, infiltration basin, vegetated swale and detention basin) based on experiences in southern California. Their evaluation included the carbon footprint estimation of construction (materials and fuel) and maintenance (fuel) operations for an assumed operational life of 20 years. The maintenance footprint of each technique was adjusted using a factor computed as the average of removal efficiencies for representative pollutants (suspended solids, nitrogen, phosphorus, zinc, copper and lead). Their results show that swales and infiltration basins are the least carbon intensive of the evaluated options, being drain inlet inserts and hydrodynamic separators the devices with the largest carbon footprints.

The Environment Agency (2009b) carried out a study to determine the potential of eight retrofit and new SuDS techniques in reducing costs and emissions (i.e. those associated with wastewater pumping). Whilst the study found that half of the retrofit SuDS investigated had a carbon payback period lower than 25 years, the report concluded that SuDS is not a cost-effective way of reducing emissions or costs on a life cycle basis, particularly in new developments where stormwater is directly discharged through a separate sewer infrastructure and additional centralised pumping and treatment is not required. This conclusion is based on the estimated marginal abatement cost, which reflects the cost of mitigating emissions through the use of each system within its life cycle (i.e. net present value of life cycle costs divided by net life cycle emission savings). Notwithstanding these conclusions, the authors insisted on the important role that SuDS can play in climate change adaptation and water management. It is important to note that broader benefits in emissions reduction were not considered in the study, such as reduced emissions deduced from flood mitigation, reduced CSO spills, drainage network operational savings, etc.

Additional results from the same study showed that green roofs and permeable pavements applied to greenfield developments provided a lower life cycle carbon footprint (and lower abatement cost in the case of the permeable pavement) when compared to conventional roofing and paving options. Finally, the report encouraged further research into the evaluation of more SuDS types and their impact on emissions taking into account flood mitigation and issues related to the wider water cycle, as well as exploring different financing mechanisms that share costs and responsibilities among the involved parties and might improve the economic appeal of SuDS.

Andrew and Vesely (2008) applied the ISO 14040:2006 LCA methodology to analyse the energy and carbon footprint performance of a monitored rain garden and a hypothetical sand filter for road runoff management on a site in New Zealand. The boundaries of the study included emissions from construction, maintenance and disposal (‘cradle-to-grave’) of both stormwater systems, but ignored potential operational benefits (e.g. carbon sequestration, stormwater management benefits). The evaluation found that the rain garden performed better than the sand filter, highlighting the contribution of transport and on-going maintenance to the total footprint. The authors acknowledged that site-specific conditions are crucial in such assessments and that system performance can vary significantly when considering a wider range of decision criteria.

Spatari et al. (2011) studied the avoided GHG emissions (including construction materials, transportation, and annual reduction in stormwater collection and treatment from the site) in a street regeneration scheme using permeable pavements and street trees, as compared to conventional street reconstruction. A LCA model was used to estimate the GHG emissions associated with the permeable paving scheme, resulting in an overall reduction over time; however, these were considered too small when compared to the emissions of the materials required to implement the scheme, translating into long payback periods (between 70-100 years). Nevertheless, the authors suggested that the implementation of such systems throughout the urban catchment may have important beneficial impacts that outweigh these results.

A broader study carried out by Moore and Hunt (2012) investigated the carbon footprint of stormwater control measures (wet ponds, constructed wetlands, bioretention systems, sand filters, level spreader-
grassed filter strips, permeable pavement, green roofs, and rainwater harvesting systems) and stormwater conveyance systems (reinforced concrete pipe, concrete swales and grassed swales). Their study accounted for: embodied carbon of construction materials and construction; and maintenance and carbon sequestration by vegetation in landscape over a 30-year period; however, end-of-life emissions or indirect emission reductions affected by the stormwater control systems (e.g. embodied carbon of irrigation water when using rainwater harvesting systems) were not considered. Their conclusions confirmed that the embodied carbon of construction materials represents a prominent part of the lifetime carbon footprint of green roofs, permeable pavements, sand filters, rainwater harvesting and reinforced concrete pipes; whereas material transport and construction dominated that of bioretention systems, ponds, wetlands, filter strips and swales. Thus, decisions regarding material sourcing (or assumptions made for life cycle assessments) may importantly impact the resulting footprint of these systems. Overall, construction and maintenance regimes are regarded as an important source of carbon, and therefore an obstacle to achieve carbon neutrality over operational lifetime (only attained by wetlands and swales in the assessment).

3 DISCUSSION

3.1 Preliminary considerations

A comparison of the results obtained from the reviewed studies is not a straightforward task for a number of reasons presented below (refer to Table 2).

Firstly, the units on which the results are reported vary across studies, including: $kg\ CO_2e/m^2$ (i.e. emissions per unit area of device/technique), $tCO_2e/ha\ DA$ or $tCO_2e/ac\ DA$ (i.e. emissions per unit area of drainage area serviced by the device/technique), $tCO_2e$ (i.e. as total or net emissions of a specific device design or intervention), $tCO_2e/year$ (i.e. annualised total or partial emissions of a specific design or intervention), $GJ$ (i.e. energy required for the total or part of the life cycle of a device/technique). Even when transforming these units it is possible in some studies (and some authors do so for clarity) this does not mean that the calculated values are correct or accurate with regards to the original ones. This effect can be particularly troubling when results expressed in $tCO_2e$ are transformed into $tCO_2e/ha\ DA$ using the drainage area of the serviced catchment. For example, a GI servicing a small drainage area can introduce large errors through such generalisations, misleading the obtained results, which will not account for scale effects.

<table>
<thead>
<tr>
<th>Reviewed study</th>
<th>LCA boundaries</th>
<th>Life cycle</th>
<th>Carbon benefits</th>
<th>Reporting units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Kirk (2006)</td>
<td>Cradle-to-grave</td>
<td>30 years</td>
<td>None</td>
<td>$tCO_2e/annum$, Total $tCO_2e$</td>
</tr>
<tr>
<td>Kosareo and Ries (2007)</td>
<td>Cradle-to-grave</td>
<td>45 years</td>
<td>Energy use in buildings</td>
<td>Impact points of LCA software</td>
</tr>
<tr>
<td>Taylor and Barrett (2008)</td>
<td>Materials, construction and operation</td>
<td>20 years</td>
<td>None</td>
<td>$tCO_2e$ (total, per stage and per annum)</td>
</tr>
<tr>
<td>Andrew and Vesely (2008)</td>
<td>Cradle-to-grave</td>
<td>50 years</td>
<td>None</td>
<td>$tCO_2e$ and $GJ$ (total and per stage)</td>
</tr>
<tr>
<td>EA (2009b)</td>
<td>Materials, construction and operation</td>
<td>25 years</td>
<td>Energy from stormwater collection</td>
<td>$tCO_2e/ha\ DA$</td>
</tr>
<tr>
<td>Spatari et al. (2011)</td>
<td>Materials, construction and operation</td>
<td>Not specified (based on payback period)</td>
<td>Energy from stormwater collection and treatment</td>
<td>Total $GJ$ and $tCO_2e$</td>
</tr>
<tr>
<td>Flynn (2011)</td>
<td>Cradle-to-grave</td>
<td>30 years</td>
<td>Carbon sequestration, energy from stormwater/buildings</td>
<td>$tCO_2e/acre\ DA$, Total $tCO_2e$</td>
</tr>
<tr>
<td>Moore and Hunt (2012)</td>
<td>Materials, construction and operation</td>
<td>30 years</td>
<td>Carbon sequestration</td>
<td>$tCO_2e/ha\ DA$, $tCO_2e/m^2$ device</td>
</tr>
</tbody>
</table>

Secondly, even when an LCA method (frequently ISO 14040:2006) is commonly used across the studies, the assessment boundaries used and the assumptions made are rather different. All the reviewed studies accounted, to a greater or lesser degree, for carbon emissions associated with the
construction (materials and transportation) and operation of GI techniques; however, only half of them (Andrew and Vesely, 2008; Flynn, 2011; Kirk, 2006; Kosareo and Ries, 2007) included emissions from decommissioning the infrastructure (end-of-life emissions). The assumed operational life of GI also differs between studies, ranging from 20 to 50 years, and avoided emissions due to carbon sequestration, stormwater management benefits or building energy savings are only accounted for in five of the reviewed studies (Environment Agency, 2009a; Flynn, 2011; Kirk, 2006; Kosareo and Ries, 2007; Spatari et al., 2011). Avoided emissions were in turn accounted in different ways or with different levels of detail (e.g. carbon sequestration was calculated using average values found in the literature in Moore and Hunt, 2012, as opposed to the iTree Eco model used by Flynn, 2011).

Finally, data used in the studies (e.g. emission inventories and other life cycle inventory data) vary according to geographical location and other specific conditions of the sites where the assessed GI systems lie (construction methods, cost of materials and fuel, catchment type, sourcing of materials and labour, etc.); accordingly, the extrapolation of any results and comparison between studies under these conditions becomes even more challenging.

### 3.2 Trends and analysis

Despite the many differences described above, it is still possible to report on trends found across the reviewed studies. The following trends intend to be a summary of collated findings in the referenced studies and should be interpreted with caution.

- Infiltration basins and swales present lower carbon footprints than other GI techniques in those studies considering a larger range of solutions (Environment Agency, 2009b; Moore and Hunt, 2012; Taylor and Barrett, 2008).

- Wetlands are generally associated with low life cycle carbon emissions when considered in the assessment (Kirk, 2006; Environment Agency, 2009b; Moore and Hunt, 2012).

- Wetlands and ponds are within the same range of life cycle emissions according to three studies assessing four or more options (Environment Agency, 2009b; Kirk, 2006; Taylor and Barrett, 2008).

- Permeable pavement and green roofs present higher initial carbon footprints (derived from the construction stage) than other GI options (Environment Agency, 2009b; Flynn, 2011; Spatari et al., 2011; Moore and Hunt, 2012).

- Filter strips present the lowest carbon footprint associated with the construction stage in two studies (Environment Agency, 2009b; Moore and Hunt, 2012).

- Sand filters showed moderate carbon emissions with relatively high associated maintenance emissions (Moore and Hunt, 2012; Taylor and Barrett, 2008).

In addition to this, it is important to note that the magnitude of the results reviewed varies significantly from study to study and their numerical comparison would not help clarifying them. This is partly due to the many factors described in the previous subsection (assumptions made, accounted variables, life cycle span, etc.) All the aforementioned factors introduced important discrepancies that are more or less evident depending on the GI type assessed. The clearest example of this is materialised in rain gardens, whose life cycle carbon footprint varied hugely across studies, from 39.8 tCO₂/ha (Andrew and Vesely, 2008), to 121.5 tCO₂/ha (Kirk, 2006), to -291 tCO₂/ha (Flynn, 2011).

The contribution of the different stages of the life cycle of GI (i.e. materials, construction, operation, decommission) to total emissions was also examined by the majority of the reviewed studies. In this sense, the carbon emissions associated with the procurement of materials and construction stage (i.e. embodied carbon) represented a large share (between 40 and 80%) of the total carbon footprint for green roofs, permeable pavement, wetlands, ponds, rain gardens and infiltration systems (Andrew and Vesely, 2008; Taylor and Barrett, 2008; Environment Agency, 2009b; Flynn, 2011; Spatari et al., 2011; Moore and Hunt, 2012). Operational emissions resulted more important in sand filters (Andrew and Vesely, 2008; Moore and Hunt, 2012) and in those GI techniques with low embodied carbon (e.g. swales and filter strips).

As a consequence of this, the ability of GI to achieve carbon neutrality within its life cycle strongly depended on two main factors: the magnitude of the embodied carbon emissions, and the accountability for avoided emissions incorporated as part of the operational carbon assessment (e.g. carbon sequestration). Thus, GI with high embodied carbon, such as green roofs and permeable pavements, presented long payback periods and did not achieve neutrality within their life cycle, even
when a number of ‘carbon benefits’ (e.g. carbon sequestration, avoidance from reduced combined sewer treatment volumes, carbon savings from enhanced building insulation) were accounted for in the assessment (Kosareo and Ries, 2007; Environment Agency, 2009b; Flynn, 2011; Spatari et al., 2011; Moore and Hunt, 2012). Similarly, GI with low embodied carbon (swales, filter strips, wetlands and ponds) were more likely to achieve carbon neutrality in those studies accounting for carbon benefits throughout their life cycle. In this sense, it is important noting that end-of-life carbon (from the decommission stage) can also represent an important part of the total emissions and may therefore importantly hinder carbon neutrality in some GI types, particularly in those whose construction and end-of-life footprints are comparable.

As pointed out before, embodied carbon emissions depend on many factors (see Table 1), including: construction methods, transportation distances, materials sourcing, etc. Thus, it is reasonable to assume that these emissions can significantly vary from case to case and that local conditions can importantly affect the ability of GI to achieve carbon neutrality.

Four of the considered studies (Kirk, 2006; Kosareo and Ries, 2007; Moore and Hunt, 2012; Spatari et al., 2011) assessed one or more than one conventional systems along with the GI options. All these studies revealed a better carbon footprint performance of GI solutions as compared to their conventional counterparts over time; however, Spatari et al. (2011) insisted that such benefits ‘can be small and slow to accrue compared to the materials needed to implement those strategies’, which supports the idea that long payback periods may be still expected for some GI solutions.

The majority of the studies analysed in the review included other criteria in their assessment, in addition to carbon emissions; however, these tended to be limited to environmental factors, and few included socio-economic impacts. In general, there was a correlation between reduced carbon footprint and overall good environmental performance. In those studies including cost impacts (Kirk, 2006; Environment Agency, 2009b; Flynn, 2011) there was also a clear correlation between higher carbon footprint and higher costs, mostly due to the impact of materials and fuel consumption at the construction stage.

3.3 An integrated framework for decision making in the water industry

The life cycle assessment of the carbon footprint of GI interventions entails a detailed and frequently difficult process that would benefit from a standardised approach if comparisons and analysis across different experiences are to be done. Further, the need to report and manage carbon in the water sector will require such an approach, against which current and future results can be assessed consistently and progress audited (Frijns, 2012).

In the UK, the water industry (through the UK Water Industry Research, UKWIR) has pioneered guidelines for the inclusion of whole life carbon accounting in asset investment decisions (UKWIR, 2008, 2012). Their approach gives recommendations on the application of LCA (e.g. boundaries of the analysis, datasets regarding carbon emission sources and emission factors commonly used across the industry) as well as assisting in the development of strategies for carbon footprint reduction. According to these guidelines, the inclusion of end-of-life emissions (i.e. those derived from decommissioning and disposal of a given asset) introduces, in practice, very large uncertainties in the assessment and may be well ignored (UKWIR, 2012) (see UKWIR whole life carbon boundaries in Table 1).

These considerations and the realisation of a robust framework for the assessment of the carbon footprint for GI have become even more important as decentralised solutions are deemed the future of integrated stormwater management. Thus, the effect that large GI interventions may have in urban drainage systems overall, and not just as isolated units servicing small catchments, must also be considered in such an assessment. As it has been shown in this review, whole life carbon footprint assessments of GI frequently overlook the vast majority of the operational side of urban drainage systems.

In other words, the interdependencies that exist between GI and the larger urban drainage system (including, the sewer network, treatment plant and receiving waters), or even the whole urban water cycle, cannot be ignored. Carbon savings due to GI do not guarantee by themselves a lower carbon footprint of the drainage system overall. Thus, carbon footprint evaluations of GI interventions at the catchment scale should ideally account for embodied carbon, and operational carbon derived from affected wastewater activities, including: direct emissions (operational fugitive emissions, fuel combustion and owned transport emissions), and indirect emissions (grid electricity from purchased energy use, operational use of chemicals, bought-in transport for future operation and maintenance).
In addition to this, decisions regarding asset planning and investment, like those affecting GI, must satisfy an increasing number of criteria that pursue sustainable solutions. Carbon footprint assessments must therefore be part of a broader decision framework that integrates a variety of evaluation tools throughout the urban drainage system. In this sense, the consideration of additional social, economic and environmental issues is commonly ignored or limited to criteria with a positive correlation with carbon footprint. Instead, assessments that analyse conflicting objectives (i.e. carbon and water quality) are far more required in order to address European and national legislation and find better compromises within a larger variety of sustainable criteria, in line with the future demands of the water sector and society.

Integrated approaches that aim to meet a number of conflicting objectives for drainage infrastructure planning are not new (Ashley et al., 2008; Makropoulos et al., 2008; Fu et al., 2010); however, their application has been limited to the academic realm and rarely found within water utilities, which in the UK tend to focus on flood mitigation and costs. Thus, carbon footprint needs to be assessed alongside other environmental and socio-economic objectives in order to better inform decisions and advance in our understanding of drainage systems as we become increasingly aware of their intrinsic complexities.

4 CONCLUSIONS

The review presented above has shown that comparing results between studies assessing the carbon footprint of GI is currently very challenging. The reason behind this is generally found in the lack of a common standardised approach that establishes clear boundaries, scope and assumptions which would permit an easier extrapolation and sharing of results.

Despite these limitations, some common trends can still be identified within the reviewed studies, pointing out to three main common ideas; 1) embodied carbon can importantly contribute to the whole life cycle carbon footprint of some GI techniques (e.g. permeable pavement), which in turn may constrain their ability to achieve carbon neutrality; 2) GI techniques result in lower carbon footprints than their conventional counterparts; 3) future work should involve broader assessments of GHG emissions and benefits to fully realise the actual role of GI in reducing the carbon footprint of the system.

Current legislation increasingly requires water utilities to manage the wider variety of social, environmental and economic impacts of their activities. In this context, decisions regarding asset planning must account for a broader number of criteria in order to find fairer compromises that pursue more sustainable solutions. In this sense, current carbon assessments fall short in contributing to inform this decision making process regarding GI in three important aspects: 1) they ignore the existing interdependencies and complexity of urban drainage systems and evaluate GI as isolated independent units; 2) they overlook the beneficial impact of introducing conflicting objectives in their assessment; 3) they do not account for the effect of GI at the large scale, ignoring the practical implications of GI for water utilities.

In order to move forward and fill these gaps, assessments of green (and grey) infrastructure looking individually at carbon, costs, etc. should be replaced by integrated assessments that consider all these objectives holistically and extend the boundaries of the analysis to embrace the urban drainage system as a whole. This way it is ensured that opportunities to understand and improve urban stormwater management are not missed in practice.

LIST OF REFERENCES


