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Life Cycle Assessment modelling of stormwater treatment systems

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ABSTRACT

Stormwater treatment technologies to manage runoff during rain events are primarily designed to reduce flood risks, settle suspended solids and concurrently immobilise metals and nutrients. Life Cycle Assessment (LCA) is scarcely documented for stormwater systems despite their ubiquitous implementation. LCA modelling quantified the environmental impacts associated with the materials, construction, transport, operation and maintenance of different stormwater treatment systems. A prefabricated concrete vortex unit, a sub-surface sandfilter and a raingarden, all sized to treat a functional unit of 35 m^3 of stormwater runoff per event, were evaluated. Eighteen environmental mid-point metrics and three end-point 'damage assessment' metrics were quantified for each system's lifecycle. Climate change (kg CO_2 eq.) dominated net environmental impacts, with smaller contributions from human toxicity (kg 1,4-DB eq.), particulate matter formation (kg PM10 eq.) and fossil depletion (kg oil eq.). The concrete unit had the highest environmental impact of which 45% was attributed to its maintenance while impacts from the sandfilters and raingardens were dominated by their bulky materials (57%) and transport (57%), respectively. On-site infiltrative raingardens, a component of green infrastructure (GI), had the lowest environmental impacts because they incurred lower maintenance and did not have any concrete which is high in embodied CO_2. Smaller sized raingardens affording the same level of stormwater treatment had the lowest overall impacts reinforcing the principle that using fewer resources reduces environmental impacts. LCA modelling can serve as a guiding tool for practitioners making environmentally sustainable solutions for stormwater treatment.

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1. Introduction

Urbanization exacerbates stormwater runoff, resulting in increased flooding risk, decreased local infiltration (Walsh, 2000) and increased metal, sediment (Sansalone and Buchberger, 1997; Zanders, 2005) and sometimes nutrient [e.g. (Walker, 1995)] pollutants. Heavy metal contaminants typically originate from roof weathering (Egodawatta et al., 2009; Pennington and Webster-Brown, 2008) and wear-and-tear of vehicle parts including brake linings (e.g. Cu) and tyre fillers (e.g. Zn) as well as additives in oil and petrol (Davis et al., 2001; Ward, 1990). These contaminants accumulate on impermeable surfaces (Egodawatta et al., 2009; Wicke et al., 2012) and are transported in runoff via stormwater networks to either sewer systems or directly to downstream aquatic ecosystems (Davis et al., 2001). Different approaches have been employed globally to mitigate stormwater. More recently, there has been increased focus on preventing stormwater runoff (i.e. source control) through green infrastructure (GI). However, legacies of older urban development require stormwater to be treated where GI preventative measures do not exist. Treatment approaches include combined sewer systems (De Sousa et al., 2012; Spatari et al., 2011), older drainage networks retrofitted with prefabricated devises (e.g. vortex separators and filters) and detention systems facilitating infiltration (e.g. raingardens, swales) primarily designed to reduce flooding and concurrently remove suspended solids (Hatt et al., 2008; Palhegyi, 2010).

Stormwater treatment technologies are typically implemented without understanding the environmental impacts associated with their materials, construction, transport, operation and maintenance quantifiable through a life cycle assessment (LCA) (ISO, 2006). Financial aspects are not usually incorporated into a LCA, but in practice a cost benefit analysis (CBA) is also typically conducted (Fisher-Jeffes and Armitage, 2011; Montalto et al., 2007) or a coupled economic input–output (EIO) analysis (De Sousa et al., 2012). Different life cycle inventory analysis (LCIA) methods,

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including Eco Indicator 99, Ecopoints, CML 2000 and EcoInvent, can influence LCA outcomes [e.g. (Zhou et al., 2009)] so caution must be exercised in interpreting and comparing results from different LCA models. LCA modelling of urban systems has been applied to building materials (Ligthart et al., 2010; Mithraratne and Vale, 2004) and municipal wastewater treatment (Foley et al., 2010; Ortiz et al., 2007; Remy and Jekel, 2008). However, there is a dearth of LCA studies reported for stormwater treatment systems despite their ubiquitous implementation (De Sousa et al., 2012; Spatari et al., 2011). It is commonly assumed that with CBA dollar savings (Davis et al., 2009; Spatari et al., 2011), GI stormwater components will incur lower environmental costs compared with manufactured units, however systematic analysis of their discrete environmental impacts has not yet been reported.

Spatari et al. (2011) examined the 'avoided' energy and green house gas (GHG) emissions of coupled Low Impact Design (LID) strategies using LCA and a stochastic urban watershed model in reducing runoff volumes otherwise treated in municipal facilities. They also employed LCA to ascertain the reduction of environmental impacts of different multi-system strategies in reducing combined sewer outflows (CSO) in New York City US (De Sousa et al., 2012). Spatari et al. (2011) concluded that it is important to investigate a variety of LID approaches applied to different urban catchments using site-specific data before a good understanding of LCA can be used in GI decision making globally. Clauson-Kaas et al. (2012) conducted LCA modelling scenarios to reduce CSO by different stormwater management approaches. Their study highlighted that focussing on global warming alone (from energy conservation and GHG emissions), as most urban LCA studies do, is important for mitigating climate change but neglected many other significant environmental impacts including eutrophication, acidification etc. There is limited data in LCA inventories for GI stormwater systems and especially for systems not connected to sewer networks. Environmental impacts from individual stormwater treatment units are also lacking, although a recent analysis by {Moore, 2012 #26} derived carbon footprints for individual stormwater control measures.

In New Zealand (NZ), especially in Christchurch, CSO is rarely of concern because stormwater is historically conveyed (independent of sanitary sewer networks) untreated to local waterways. On-site infiltrative stormwater detention is the current preferred approach for mitigation. As legislation progressed, coupled with increased frequency of extreme weather patterns due to climate change, catchment managers are required to retrofit older and design new urban developments with local stormwater treatment systems. A previous NZ study (Andrew and Vesely, 2008) conducted a LCA of two (one hypothetical) stormwater treatment systems but did not employ a dynamic modelling tool, which limited their outputs to estimations of embodied energy (joules) consumption and CO_2 emissions. In this paper, LCA modelling is presented for different on-site stormwater treatment systems by employing a robust tool affording detailed insight into the environmental sustainability of these systems. The environmental impact of 18 mid- and 3 end-point environmental metrics with five different scenarios including recycled materials and a smaller system footprint, are quantified. The resulting analysis provides information beneficial to regulators and practitioners charged with making sustainable stormwater treatment engineering decisions. In Christchurch, the second largest city in NZ, more than two thirds of the water infrastructure (including stormwater network) is currently being rebuilt following the devastating 2010–2011 earthquakes. An understanding of its full environmental impacts, especially of smaller decentralised systems that afford lower flood potential and greater energy independence, is important for ensuring sustainable re-development of a resilient city.

2. Methods

2.1. Treatment systems

Stormwater treatment designs investigated included operational systems with a 30 year design life comprising a prefabricated (manufactured) concrete vortex unit and a sub-surface sandfilter. A raingarden designed according to current NZ practice (ARC, 2003) and similarly documented (Andrew and Vesely, 2008) was also evaluated. The vortex unit consisted of a cylindrical concrete chamber (1.2 m diameter) with internal plastic (HDPE), stainless steel and cast iron components (Fig. 1A). The subsurface sand filtration system (Fig. 1B) consisted of a containment wall (100 cm D X 14 cm W X 34 cm L to protect road sub-base from failure), a cylindrical sediment chamber with a cast iron dome lid, six in-series perforated plastic detention chambers and a polypropylene (PP)-lined sandbed chamber. These components were made from: concrete for the sediment trap and containment wall; recycled plastic PP for the detention chambers; new PP for the geotextile weed mat and sand and gravel as filtration media (Table S1), mainly modelled using data provided in the Australasian LCI (Grant, 2011) database. The raingarden (Fig. 1C) consisted of an upper bark-chip layer for inflow velocity control, local 'topsoil' media (excavated from the ground) overlying sand and gravel separated by a perforated geotextile liner and PVC under drain pipes.

Each modelled system operated by gravity-fed stormwater runoff and infiltration through the system, requiring no pumping and was sized to treat stormwater from the same 5300 m^2 (75% impervious) older local catchment receiving combined roof and road runoff. All system designs were targeted to efficiently address common urban stormwater pollution problems of elevated total suspended solids (TSS) and metals. The modelled systems had the same water quality treatment (WQT) requirements but different footprints: 69 m^2 for the raingarden, 60 m^2 for the sandfilter system, and <3 m^2 for the concrete unit. The water quality volume (WQV) resulting from one third (15.33 mm) of the site-specific two-year 24-h design storm depth (using standard NZ hydrological data) was calculated providing a functional unit of 35 m^3 WQV (Figure S1) to be treated in each system during each storm event. A WQV functional unit was similarly reported in NZ (Andrew and Vesely, 2008) but is different to that reported in US studies where drainage area is used based on regulatory sizing guidelines adopted in Pennsylvania (US) (De Sousa et al., 2012; Spatari et al., 2011). The difference in functional unit adopted in NZ and US studies might relate to the fact that stormwater networks are typically independent of sewer systems in NZ by contrast to many US situations. Two additional scenarios were included within the analysis to investigate design criteria that could alter their environmental impacts: (1) employing new instead of recycled plastic for the hydraulic detention chambers in the sandfilter system and; (2) a 40% smaller raingarden footprint. These scenarios were realistic given the detention components of the sandfilter are typically recycled plastic but similar products might constitute new plastic. Reducing (−40%) the raingarden to 60% of classic design size to treat the same WQV to the same level is reported elsewhere (Andrew and Vesely, 2008; Smythe et al., 2007); results can be explained by the typically higher hydraulic conductivity (~10 times) occurring in NZ raingardens compared to the conservative current design guidelines of >0.3 m/day (ARC, 2003).

2.2. LCA model

Life Cycle Assessment modelling was conducted using SimaPro 7.3 software (Pré, 2008), widely adopted in other LCA applications

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