Ecological functions within a Sustainable Urban Drainage System

J. I. Jackson* and R. Boutle.

**School of Applied Sciences, Park Campus, Boughton Green Road, Northampton, NN2 7AL*
Corresponding author, e-mail: janet.jackson@northampton.ac.uk

ABSTACT
Sustainable Urban Drainage Systems (SUDS) are regarded as engineering solutions to urban storm water control and flood risk. Additional benefits from SUDS in the built environment include sediment entrapment and remediation of water quality from urban runoff through the use of retention/detention systems. Biodiversity value of SUDS is alluded to but few studies have evaluated conservation potential or monitored ecological processes that may occur within them.

This paper presents macro-invertebrate data from the first of the two SUDS units integrated into the design of a new urban extension at Upton, Northampton, UK. Surveys of both swales and retention ponds were performed in 2006 and 2007. Data demonstrates that aquatic fauna have colonised newly constructed sites and that monitoring biodiversity may provide a way forward towards devising a method to assess ecological status/integrity and ecological services of SUDS. These surveys have demonstrated that over a two year period the SUDS at Upton has operated as backwater habitats for aquatic and wetland species that have dispersed from the River Nene Valley. Spatial analyses of the aquatic invertebrates also indicated a previously unidentified inlet that may have had a detrimental impact. The authors discuss the need for incorporating ecological considerations into management plans for SUDS.

KEYWORDS
Ecological function, ecology, macroinvertebrates, management, monitoring, SUDS.

INTRODUCTION

Water resource management is becoming a significant planning issue as climate change scenarios, past experience of flooding, drought, modifications to river channels and development of surrounding catchments are considered. Another urgent issue in contemporary landscapes is the decline of biodiversity more recently urban areas are increasingly appreciated as important wildlife refuges (Sandström *et al.* 2006). Much practical advice has been directed at engineers and landscape architects, steering them towards best practice in design of Sustainable Urban Drainage Systems (SUDS) (HR Wallingford, 2004a, 2004b; TCPA, 2004; Balmforth, *et al*. 2006; Woods-Ballard, *et al*. 2007) and aim to address technical considerations of engineered solutions to manage storm water management associated with development. These documents have also emphasised the use of natural vegetation (native flora), as an integral functional component that facilitate reduced water velocity, energy dissipation and bank stabilisation. Vegetation within SUDS is also said to filter suspended solids, uptake nutrients and improve aesthetic quality (Woods-Ballard, *et al*. 2007). Government planning policy advise that SUDS should also be promoted for their potential for amenity value and biodiversity (ODPM, 2004; LCG, 2006). However, since recommendations by Brix (1999), there are still too few examples or case studies that illustrate how to:

- design for ecological benefit
- quantify ecological status
- assess biodiversity value
- evaluate 'SUDS ecology' and processes that may contribute to SUDS function.

River regulation, development and agriculture have all contributed to habitat loss and degradation of wetlands on floodplains and rivers, endangering many once common aquatic and wetland species (Brönmark and Hansson, 2002; Wood, *et al.* 2003; Jansson, *et al.* 2007). However, since the early 1990s the value of floodplains, wetlands and aquatic habitats has been reconsidered (Finlayson and Moser, 1991; Everard, 1997; Holmes, 1998) and now are deemed to have social, economic and environmental assets in terms of ecosystem goods and services (Turner, *et al.* 2000; Kwolek and Jackson, 2001; Schuyt and Brander, 2004; Turner and Daily, 2008). This philosophy should be applied to SUDS and other constructed wetlands as they could be valued not just as engineered solutions to mitigate development but as ecosystem services: services for storm water retention/detention (Faulkner, 1999; Mitsch, *et al.* 2005) and amelioration of urban runoff (D'Arcy and Frost, 2001; Shutes, 2001; Revitt, *et al.* 2004; Jones and Macdonald, 2007).

Newly created, or engineered wetlands can be rapidly colonised and provide conditions for specialised species (Noon, 1996). Constructed wetlands have been shown to give added ecological value by enhancing biodiversity and supporting species in decline (Biggs, *et al.* 2000; Jackson, *et al.* 2006). The biodiversity value of new aquatic or wetland habitats would be landscape dependent, in that colonisation is dependent on immigration processes and availability of species pools (Williams, *et al*. 2003) and species diversity is more likely to be higher if existing habitat are within dispersal distances (Matthiessen and Hillebrand, 2006).

There is limited information available of the ecological function of SUDS systems, apart from the function of plants. Previous research has indicated that some macroinvertebrates are intolerant of pollutants found in urban runoff (Beasley and Kneale, 2004) thus could be indicative of 'SUDS health' or 'ecological status'. As urban runoff has the potential to contaminate receiving water bodies (White and Howe, 2004) in the long term, monitoring to detect changes in community composition within SUDS could provide early signals to warn of water quality issues and inform management decisions.

METHOD

The study site is within Phase 1 of a new urban extension to the west of Northampton, UK. This 44ha development site will deliver 1,382 homes and mixes use commercial area (English Partnerships 2007). The development was divided into two hydrological units at the planning stage, each unit containing a roof to river SUDS, upstream from vulnerable flood risk areas in the town centre (Jackson, submitted). This paper presents data from the smaller of the two drainage systems on construction Site A which was completed early 2007. This system contains deep swales alongside newly built homes, which discharge into a green-field pond attenuation area before the runoff is released into the River Nene. As an ex-arable site the development at Upton has had no previous urban runoff exposure. In 2006, eight sites (Figure 1) were sampled, Sites 1, 3, 4 and 5 in the urban environment and Sites 6, 7, 8 and 10 in the green-field pond retention system. The urban swale system was installed post house construction between 2004 and 2007. The green-field retention system was excavated in 2002 before the housing development commenced. In 2007 a new swale site was opened in the urban area as construction completed (Site 2) so an additional site was allocated to the retention system (Site 9), making ten sites in total.

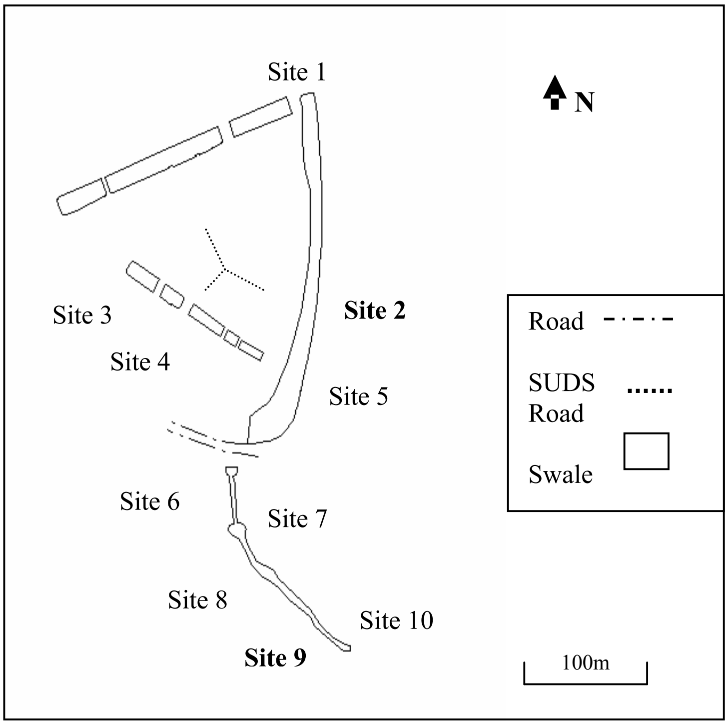

Figure 1. Sketch outline of the SUDS at Site A, Upton, Northampton, UK: indicating location of sampling sites (in **bold** Sites 2 and 9 added in 2007).

Sampling for macroinvertebrates and timed effort

Standard methods for sampling both rivers (Clarke*, et al.* 2003) and ponds (Pond Action, 1998) in the UK are based timed effort for three minutes with a kick net or pond net. Potential habitats created by SUDS systems can fall between the physical characteristics of a flowing water body during periods of rainfall, as in a stream or river, or a still water body during dryer periods, as in a pond. In addition, depending on the engineered design of the

SUDS, geology, rainfall characteristics and air temperature, some elements may be completely dry or resemble marshy wetland. At Upton, the hydrological capacity of the SUDS can determine the maximum surface area of water bodies. For this study, it was felt that the standard method of sampling would over sample the smaller water bodies and would be detrimental to the aquatic invertebrate communities within the SUDS. No previous studies were found that examined the impact of timed effort on either the sample representation or the effect of removing too many invertebrates from an aquatic community. Therefore, considering the size and structure of the water bodies within the Upton SUDS a timed pond sweep effort for each sampling site was reduced to six 10-second sweeps within a representative 10 metre section of SUDS element or the entire water body if less than 10 metres square in area. Whilst sweeping, the surveyor searched through identifiable micro-habitats with the net to increase sample representation, as recommended by Halse, *et al.* (2002). Contents of the net were emptied into one bucket after each ten second effort and individuals were identified to major taxonomic groups, family and where possible to species level.

Macroinvertebrate analysis

Similar to querying the timed sampling method above, the analysis of macroinvertebrate data in SUDS was also questioned. In both UK river and pond survey methods both use the British Monitoring Working Party Score systems (BMWP) and Average Score per Taxon (ASPT) (Pond Action, 1998; Clarke, *et al.* 2003). These scoring systems are based upon a premise that some species are intolerant of organic pollution and prefer fast flowing waters with high levels of dissolved oxygen, while others are adapted to tolerate organically polluted and sites with high Biological Oxygen Demand (BOD) (Mason, 1996). A classic publication in 1970 by Hynes illustrated how taxonomic adaptation of aquatic invertebrates has evolved various traits that enable organisms to:

- utilise varying conditions under differing substratum characteristics
- escape from unsuitable conditions (flight)
- to obtain air for respiration
- adapted to utilise dissolved oxygen in rapid flow, slow moving water or pond/lake environments.

Essentially a sample from a SUDS element could score poorly with a biotic indices as SUDS will have different hydrodynamics to rivers (Brinson, 1993) and will not provide a habitat for the highest scoring groups (Rabeni, 2000; Whitledge and Rabeni, 2000). In addition, some SUDS elements could act as temporary ponds or wetlands due to the nature of retention, evaporation and infiltration design and it would be difficult to consistently identify ecological characteristics with an indices value. Davis, *et al.* (2006) suggested that wetlands are more complex with variable vegetation assemblages depending on topography and other physical variables and recommended that biological indices were not suitable for analysing macroinvertebrate data.

Physical and chemical parameters.

Prior to sampling for macroinvertebrates, water samples were taken for nitrate and phosphate analysis in the laboratory with the HACH Colorimetric method (HACH, 2003a; HACH, 2003b). The pH, salinity, total dissolved solids (TDS), conductivity and temperature were measured using digital electronic meters. The five water depth measurements were taken using a metre rule at each sample site.

RESULTS

Abiotic: Physical/Chemical data

In 2007 the depth of the swales varied from 26.0cm to 18.6 cm in the house development Site A. Depth within the green-field swales and ponds ranged from 10.4 to 22.8 cm with no significant variation between the two years of data. pH ranged from 6.9 and 9.74 in 2006 and 7.0 to 7.98 to in 2007. Fluctuations in pH in the first year of sampling could have been be due to the influence of the house building and the construction final phase of this SUDS; the range of pH values were more stable in 2007. Conductivity, TDS and salinity were found to have higher values in 2007 and revealed a similar spatial pattern through the SUDS indicating secondary inputs between the housing and non-housing sites. A fall in conductivity was found through the urban sections of the system in both years (Figure 2). The standard deviation for Site 5 indicated variability between measurements taken within this 10 metre section and values then increased sharply at Site 6 at the start of the green-field section of the SUDS. Conductivity levels were also variable in the housing sites and ranging from 421.0 to 696.3 $\mu S\ cm^{-1}$. There was a significant difference in conductivity levels between the urban and green field retention sites (*Kruskal Wallace p=0.002; One-way ANOVA p=<0.001*). A post hoc test showed significant differences between the non-housing site means and sites 3, 4 and 5 (*LSD p=<0.001*). TDS varied from 203.7 to 339.7 mg l^{-1}. Values increased slightly at Site 5, and rose to 391.7 mg l^{-1} in Site 6 and reduced to 370.7 mg l^{-1} by Site 10 (Figure 3). There was a significant difference in the TDS level between urban and green field sites (*Kruskal Wallace p=0.002; One-way ANOVA p=<0.001*). A post hoc test confirmed there was a significant difference between all the non-housing sites and all the housing sites (*LSD <0.05*). Salinity levels increased from 0.3 mg l^{-1} in the housing sites to 0.4 mg l^{-1} into the green-field sites. The differences in salinity between all the sites were significant (*Kruskal Wallace p=0.002; One-way ANOVA p=<0.001*).

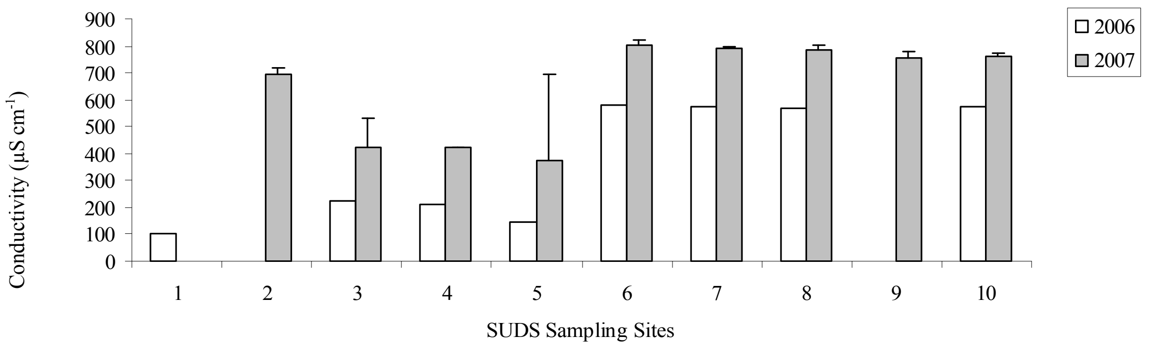

Figure 2. Mean conductivity through Site A SUDS at Upton, Northampton, UK.

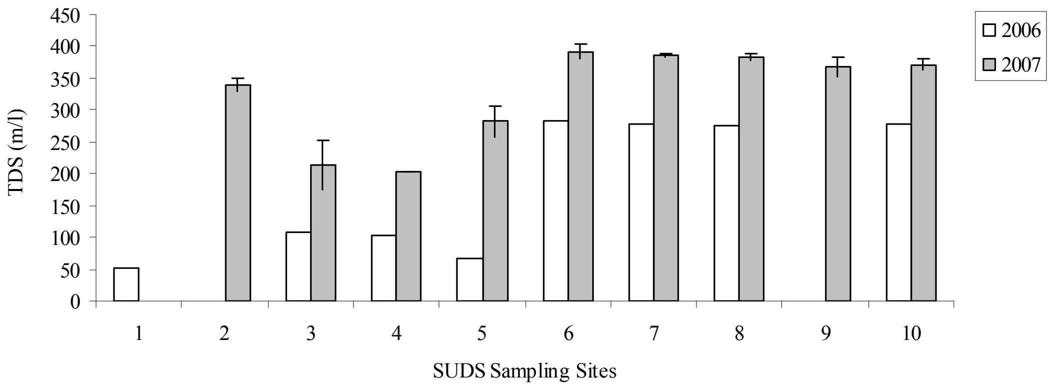

Figure 3. Mean Total Dissolved Solids through Site A SUDS at Upton, Northampton, UK.

Biotic: Macroinvertebrate data

A total of 21 taxonomic groups were identified in 2006, with Site 8 having a maximum of 13 families. By 2007, a total of 34 families were found: 20 families were in Sites 3, 8 and 10 (Table 1). A total of 14 families were new in 2007 (Table 2). Two species of Gastrapoda (snails) (Families: Lymnaeidae and Planorbidae) were identified in 2007. The aquatic pond snail (*Lymnea peregra*) had colonised all sites; with substantially higher numbers of juveniles in Sites 8 and 10 (Table 2). Figure 4 displays the total abundance of individuals in family groups with the extraction of the *Lymnea peregra*; to remove the impact of those extreme outlier data. These abundance values strongly suggest depressed populations at both Site 5 and Site 6 and may indicate an ecological response to an undetermined polluting inlet indicated by the abiotic data above.

Table 1. Macroinvertebrate Families in SUDS site locations.

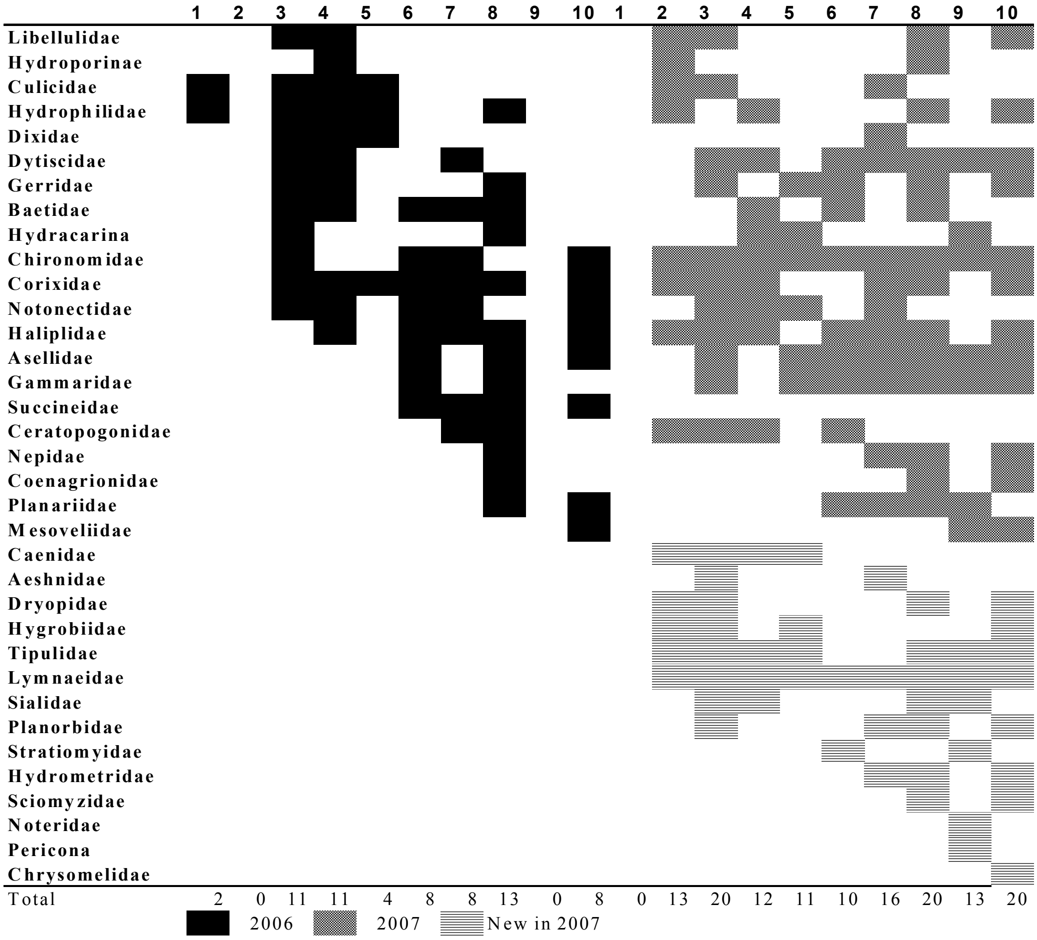

Table 2. Family number and abundance of Gastropoda (snails).

Site	1	2	3	4	5	6	7	8	9	10
Lymnea peregra	-	109	9	229	61	59	49	**1788**	41	**1273**
Planorbis albus	-	-	3	-	-	-	1	5	-	1

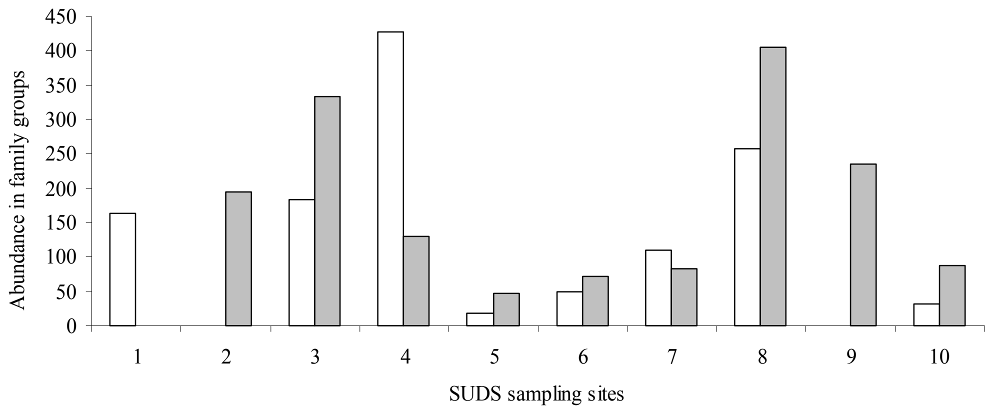

Figure 4. Abundance of individuals in family groups (excluding *Lymnea peregra).*

Discussion

The landscape context of habitat is important when considering dispersal and colonization processes (Matthiessen and Hillebrand, 2006; Van de Meutter, *et al.* 2006), particularly when assessing newly created freshwater habitats (Williams *et al*. 2007) or constructed SUDS. The SUDS at the Upton site have been monitored since 2003 and observations have demonstrated additional benefit to local biodiversity (Jackson, *et al*. 2006). The results presented in this paper illustrate that freshwater 'SUDS habitats' were created and colonised rapidly by macroinvertebrates. Newts were found in Site 10 and both frogs and toads have been present within these SUDS habitats during the monitoring period. Results also indicated that some family groups utilised these new aquatic and wetland habitats as an aquatic backwaters that are physically connected to the river.

Site 5 was found to have fewer macroinvertebrate families and less abundance. Suppressed populations or absentees could be due to point sources of urban runoff and the effected SUDS sites would be then considered as 'sink' habitats. The bio-monitoring of freshwater habitats has progressed and particularly as researchers now have to consider the implementation of the European Water Framework Directive. Fresh water ecologists are focusing at relationships between environmental parameters, invertebrate life histories and taxonomic traits (Statzner, *et al.* 1994; Rabeni, 2000, Dolédec and Statzner, 2008) to determine ecological status of aquatic and wetland habitats. This study also demonstrated how macroinvertebrates could be used as a 'biotic alarm system' that checks the 'SUDS ecosystem health'.In terms of bio-monitoring, some taxonomic groups (e.g. beetles) can either fly or swim rapidly from unfavourable sites (Davy-Bowker, 2002; van de Meutter, *et al*. 2006; Williams, *et al*. 2007). Other groups, e.g. dragonflies and damselflies, are attracted to vegetation but if conditions are not suitable once eggs have been deposited, aquatic nymphs will not reach maturity and emerge as adults (Gibbons, *et al.* 2000). For future research at Upton, we will need to understand 'SUDS ecology' and to determine which water quality parameters or habitat conditions limit population abundance and species diversity.

Noon (1996) observed that for newly created wetland systems, ecological processes called primary succession were steering biodiversity and that man made wetlands develop over time similar to natural wetland habitats. Accumulated sediments provide substrata for burrowing macroinvertebrates and a medium for plants to establish. Vegetation within the Upton SUDS has been observed as having an important ecological role in providing shelter, perches, and food, nesting and breeding sites for a wide variety of fauna. A significant number of bird species observed at Upton are associated with wetlands and river floodplain habitats. Many of these bird species are of conservation concern and have been classified as red or amber status: for example Reed Bunting (UKBAP and Red Status species) have been frequently observed in a block of *Typha latifolia* (Reedmace) in one of the larger retention ponds and may be breeding there. Macroinvertebrates presented in this paper may contribute to an ecological food web that extends beyond the SUDS. During the monitoring period birds and bats have been recorded feeding around the retention ponds and along the swales within the housing development.

HR Wallingford (2003, 2004) recommended that SUDS should have the capacity to work with minimal intervention. Ellis, *et al.* (2006) warned that accumulated sediment could contain chemical pollutants, nutrients, heavy metals, hydrocarbons, and other elements from an urban catchment. If disturbed, they may have a significant impact on other SUDS elements, the receiving river and downstream ecosystems. SUDS performance and ecological status over the long term could be at risk without a long term management plan. The frequency of sediment removal needs to be considered. As well as providing structure and an aesthetic quality, aquatic vegetation can trap pollutants within the sediments. *Typha latifolia*, for example, has been shown to tolerate and bioaccumulate heavy metals within plant tissues (Ye, *et al*. 1997) and will physically assist in sediment detention. Removing sediments and plant material may require controlled disposal.

Further research will be directed towards understanding the SUDS catchment, sediment accumulation and biodiversity. An education programme is also planned to help new residents to be aware of local biodiversity within 'their SUDS'. However, external sources of contaminants from road runoff are beyond to influence of the residential development will need to be examined in more detail. However, if water quality of inlets and outlets are identified as having 'ecologically limiting factors', then improved mitigation measures could be installed at multiple inlets sites and reducing the frequency and cost of SUDS management.

CONCLUSION

Ecological processes will not only inform biodiversity potential, but enable the assessment of 'SUDS ecosystem health', function and capacity. Monitoring organisms will also assist in long term management planning. Adaptive management solutions can be designed to take into account dispersal and colonization processes and the landscape context. A deeper understanding of the ecological function and biodiversity assets of Sustainable Urban Drainage Systems can potentially lead to the development of landscape-scaled strategic tools, to help plan and design not only storm water management systems within a river catchment (de Groot, 2006), but can also influence the mobility and habitat availability of rare and declining species in urban and rural landscapes.

REFERENCES

Balmforth D., Digman C., Kellagher R. and Butler D. (2006). Designing for excedance in urban drainage - good practice. CIRIA, London.

Beasley G. and Kneale P. E. (2004). Assessment of heavy metal and PAH contamination of urban streambed sediments on macroinvertebrates. *Water, Air and Soil Pollution*, 4, 563-578.

Biggs J., Williams P., Whitfield M., Fox G. and Nicolet P. (2000). Ponds, Pools and Lochans: Guidance on good practice in the management and creation of small water bodies in Scotland. Scottish Environmental Protection Agency, UK.

Brinson M. M. (1993). A Hydrogeomorphic classification for Wetlands. Wetlands Research Program Technical Report, WRP-DE-4, US Army Corps of Engineers, Washington DC, USA.

Brix H. (1999). How 'Green' Are Aquaculture, Constructed Wetlands and Conventional Wastewater Treatment Systems? *Water Science and Technology*, **40**(3), 45-50.

Brönmark C. and Hansson, L. (2002). Environmental issues in lakes and ponds: current state and perspectives. *Environmental Conservation*, **29**(3), 290-306.

Clarke R. T., Wright, J. F. and Furse M. T. (2003). RIVPACS models for predicting the expected macroinvertebrate fauna and assessing the ecological quality of rivers. *Ecological Monitoring*, **160**, 219-233.

D'Arcy B. and Frost A. (2001). The role of best management practices in alleviating water quality problems associated with diffuse pollution. *The Science of the Total Environment*, **265**, 359-367.

Davis J., Horwitz P., Norris R., Chessman B., McGuire M. and Sommer B. (2006). Are river bioassessment methods using macroinvertebrates applicable to wetlands? *Hydrobiologia*, **572**, 115 -128.

Davy-Bowker J. (2002). A mark and recapture study of water beetles (Coleoptera: Dytiscidae) in a group of semi-permanent and temporary ponds. *Aquatic Ecology*, **36**, 435-446.

de Groot R. (2006). Function-analysis and valuation as a tool to assess land use conflicts in planning for sustainable, multifunctional landscapes. *Landscape and Urban Planning*, **75**, 175-186.

Dolédec S. and Statzner B. (2008). Invertebrate traits for the biomonitoring of large European rivers: an assessment of specific types of human impact. *Freshwater Biology*, **53**, 617-634.

EC (2000). Directive 2000/60/EC of the European Parliament and of the Council: establishing a framework for Community action in the field of water policy. *Official Journal of the European Communities*, **L327**, 1-72.

Ellis B. J. and Mitchell G. (2006). Urban diffuse pollution: key data information approaches for the Water Framework Directive. *Water and Environment Journal*, **20**, 19-26.

English Partnerships (2007) Upton / South-West District, Northampton. www.englishpartnerships.co.uk/upton.htm. Date accessed: May 2008.

Everard M. (1997). Development of a British wetland strategy. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **7**(3), 223-238.

Faulkner B. (1999). The control of surface water runoff from new development: UK national 'policy 'in need of review? *Urban Water*, **1**, 207-215.

Finlayson M. and Moser M. (eds.) (1991). Wetlands. International Waterfowl and Wetlands Research Bureau: Oxford, UK.

Gibbons L. K., Reed J. M. and Chew, F. S. (2002) Habitat requirements and local persistence of three damselfly species (Odonata: Coenagrionidae). *Journal of Insect Conservation*, **6**, 47-55.

HACH (2003a). DR/4000 Procedure: Nitrate, Cadmium Reduction Method. HACH Company, available from: www.hach.com, visited January 2008.

HACH (2003b). DR/4000 Procedure: Phosphorous, Reactive (Orthophosphate), Amino Acid Method. HACH Company, available from: www.hach.com, visited January 2008.

Halse S. A., Cale D. J., Jasinska E. J. and Shiel R. J. (2002). Monitoring change in aquatic invertebrate biodiversity: sample size, faunal elements and analytical methods. *Aquatic Ecology*, **36**, 395-410.

Holmes N. T. H. (1998). Floodplain Restoration. In: Bailey R.G., Jose, P.V. and Sherwood B.R. (eds.): United Kingdom Floodplains. Westbury, Otley, UK, pp. 331-348.

HR. Wallingford (2004a). Operation and Maintenance of Sustainable Drainage Infrastructure. Information Sheet SR 626, HR Wallingford, Oxford, UK.

HR. Wallingford (2004b). Maximising the ecological benefits of sustainable drainage schemes. Information Sheet SR 625, HR Wallingford, Oxford, UK.

Hynes H. B. N. (1970). The Ecology of Running Waters. Liverpool University Press, Liverpool, UK.

Jackson J. I., Holt C. P., Newman D. R. and Clarke D. J. (2006). Planning for environmental and ecological gain with Sustainable Urban Drainage Systems (SUDS): monitoring and evaluation of a storm water drainage system within a new housing development at Upton, Northampton, UK. In: Davies B. and Thompson S. (eds.): Water and the landscape: The landscape ecology of freshwater ecosystems. Proc. 14th An. Conf. IALE(UK), September, Oxford Brookes University, Colin Cross Printers Ltd., Garstang, UK, pp47-53.

Jackson J. I. (Submitted). Partnership development and collaboration in sustainable housing developments. *Proceedings of the Institute for Civil Engineers: Management Procurement and Law*, submitted January 2008.

Jansson R., Nilsson, C. and Malmqvist, B. (2007). Restoring freshwater ecosystems in riverine landscapes: the roles of connectivity and recovery processes. *Freshwater Biology*, **52**, 589-596.

Jones P. and Macdonald N. (2007). Making space for unruly water: Sustainable drainage systems and the disciplining of surface runoff. *Geoforum*, **38**, 534-544.
Kwolek A. K. and Jackson J. I. (2001). Floodplains and Agenda 21: The Upper Nene Valley Project. *Sustainable Development*, **9**, 165-174.
LCG (2006). Planning Policy Statement 25: Development and Flood Risk. Communities and Local Government, The Stationary Office, London, UK.
Mason C. F. (1996). Biology of Freshwater Pollution. Longman Group Ltd., Harlow, UK.
Matthiessen B. and Hillebrand H. (2006). Dispersal frequency affects local biomass production by controlling biodiversity. *Ecology Letters*, **9**, 652-662.
Mitsch W. J., Zhang L., Anderson C. J., Altor A. E. and Hernández M. E. (2005). Creating riverine wetlands: Ecological succession, nutrient retention and pulsing effects. *Ecological Engineering*, **25**, 510-527.
Morris R. K. A., Alonso I., Jefferson R. G. and Kirby K. J. (2006). The creation of compensatory habitat - Can is secure sustainable development? *Journal for Nature Conservation*, **14**, 106-116.
Noon K. F. (1996). A model of created wetland primary succession. *Landscape and Urban Planning*, **34**,97-123.
ODPM (2004). Planning Policy Statement 23: Planning and Pollution Control. Office of the Deputy Prime Minister, The Stationary Office, London, UK .
Pond Action (1998). A guide to the methods of the National Pond Survey. Pond Action, Oxford, UK.
Rabeni C. F. (2000). Evaluating physical habitat integrity in relation to the biological potential of streams. *Hydrobologia*, **422/423**, 245-256.
Revitt D. M., Shutes R. B. E., Jones R. H., Forshaw M. and Winter B. (2004). The performances of vegetative treatment systems for highway runoff during dry and wet conditions. *Science of the Total Environment*, **334-335**, 261-270.
Sandström U. G., Angelstram P. and Khakee A. (2006) Urban comprehensive planning: identifying barriers for the maintenance of functional habitat networks. *Landscape and Urban Planning*, **75**, 43-57.
Santoul F., Figuerola J. and Green A. J. (2004). Importance of gravel pits for the conservation of water birds in the Garonne river floodplain. *Biodiversity and Conservation*, **13**, 1231-1243.
Schuyt K. and Brander L. (2004). Living Waters, Conserving the source of life: The Economic values of the World's Wetlands. World Wide Fund for Nature, Zeist, The Netherlands.
Shutes R. B. E. (2001). Artificial wetlands and water quality improvement. *Environmental International*, **26**, 441-447.
Statzner B., Resh V. H. and Roux, A. L. (1994). The synthesis of long-term ecological research in the context of concurrently developed ecological theory: design of a research strategy for the Upper Rhône River and its floodplain. *Freshwater Biology*, **31**, 253-63.
TCPA (2004). Biodiversity by design: A guide for Sustainable Communities. Town and Country Planning Association, London, UK.
Turner R. K. and Daily G. C. (2008). The Ecosystem Services Framework and Natural Capital Conservation. *Environ. Resource Econ.* **39**, 25-35.
Turner R. K., van den Bergh J.C.J.M., Söderquist T., Barendregt A., van der Straaten J., Maltby E. and van Ierland E. C. (2000). Ecological-economic analysis of wetlands: scientific integration for management and policy. *Ecological Economics*, **35**, 7-23.
Van de Meutter F., Stoks R. and De Meester L. (2006). Lotic dispersal of lentic macroinvertebrates. *Ecography*, **28**, 223-230.
White I. and Howe A. (2004). The mismanagement of surface water. *Applied Geography*, **24**, 261-280.
Whitledge G. W. and Rabeni C. F. (2000). Benthic community metabolism in three habitats in an Ozark Stream. *Hydrobologia*, **437**, 165-170.
Williams D. D., Heeg N. and Magnusson A. K. (2007) Habitat background selection by colonizing intermittent pond invertebrates. *Hydrobologia*, **592**, 487-498.
Williams P., Whitfield M., Bigg, J., Bray S., Fox G., Nicolet P., and Sear D. (2003) Comparative biodiversity of rivers, streams, ditches and ponds in an agricultural landscape in Southern England. *Biological Conservation*, **115**(2), 329-341.
Williams P., Whitfield M. and Biggs J. (2007) How can we make new ponds biodiverse? A case study monitored over 7 years. *Hydrobiogia*, **597**, 137-148.
Woods-Ballard B., Kellagher R., Martin P., Jefferies C., Bray R. and Shaffer, P. (2007). The SUDS Manual. CIRIA: London.
Woods P. J., Greenwood M. T. and Agnew M. D. (2003). Pond biodiversity and habitat loss in the UK. *Area*, **35**(2), 206-216.
Ye Z. H., Baker A. J. M., Wong M. H. and Willis A. J. (1997). Zinc, Lead and Cadmium tolerance, uptake and accumulation by *Typha latifolia*. *New Phytologist*. **136**, 469-480.