ATKINS

From Regeneration to Roman Snails

A collection of published works by the Atkins ecology team
Edited by John Box and Jules Price

Plan Design Enable

"The world's future is influenced by the actions we take today. Atkins is committed to playing its part in ensuring that the world is on a development pathway which will meet the needs of future generations within the limits of the planet's resources. This means reducing the impacts of our own operations and working in a responsible, ethical and accountable way. By protecting and improving ecosystems, we will ensure we have a healthy environment for the future."

We employed our first ecologists over 40 years ago when they formed a small team of two to work on the impact assessments of projects such as the M1/M11 construction scheme and the proposed oil refinery in Cromarty. Atkins had the wisdom to invest in ecologists throughout the world – in addition to our team of over 60 in the UK, we have teams in other parts of Europe, North America and the Middle East.

We have shown time and time again that the key to successful projects is multidisciplinary working. The dynamic and influential relationship between our ecology and our engineering, planning and design teams results in the best outcomes in terms of design, consents and delivery for urban and infrastructure projects.

This collection of papers illustrates the significant breadth and depth of knowledge of our ecologists and their desire to push the boundaries and to find effective, innovative and sustainable solutions to the technical issues that we address daily. People are key to our business being successful and I am immensely proud that our ecology team is committed to the Atkins ethos of high standards and creative thought.

Uwe Krueger
Chief Executive Officer

Foreword

The days when looking after the environment was regarded as a barrier to progress, harmful to competitiveness and detrimental to growth are coming to an end. In its place we are entering an era in which it will become increasingly necessary for plans and strategies to integrate economic and ecological goals, rather than trading them off against one another.

This shift in emphasis is in part being driven by research that shows how in a multitude of different ways the economy is 100 per cent reliant on nature, and how without the services provided by natural systems there can be no growth or development. From the pollination of crops to the replenishment of freshwater, nature sustains huge economic value and this fact is increasingly well recognised by policy makers and company executives.

Rising awareness about the different ways that ecosystems underpin the economy is in turn leading to new policies and commercial strategies in which the protection of nature's services is a core component. This is not to say that environmental issues are no longer being driven by ethics or the intrinsic values of nature, but more to highlight how there is now an additional new driver that will become increasingly important in the years ahead.

It is still early days, but leadership is already being demonstrated, which is shaping future expectations as to what acceptable practice will look like. The way in which ecological design was incorporated into the Olympic Park in east London and is being similarly prioritised in a major new housing development being built on the site of a former oil refinery in South Wales provide important cases in point.

On top of changing expectations among clients, public demand for thought-through environmental solutions that improve the quality of life and good practice that reflects this emerging new narrative, legislation is evolving as well. All this will lead to a premium being placed on methods and expertise that can deliver in this new context.

From the conservation of rare species to the remediation of abandoned former industrial land, there is a range of mitigation and enhancement methods. They include carefully planned translocations, habitat restoration methods and design solutions to protect sensitive ecosystems and species from development impacts. All this is being informed by a growing ecological science literature and access to new technologies that can be used to gather better data to support stronger management decisions.

Key to success in this demanding new context is the effective establishment of cross-disciplinary teams through which the combined skills of for example engineers, architects and ecologists can be harnessed together in creative solutions-oriented professional environments. I have been delighted to see how Atkins is rising to this emerging challenge, not only through its teams of expert ecologists, but also through how their expertise is being effectively integrated.
As the idea of 'green infrastructure' gathers momentum it will be vital that their expertise and experience is shared so as to accelerate progress.
This publication, rich in ideas and information, is an excellent and important part of that process.

Dr. Tony Juniper
President, Society for the Environment

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Preface

Where do infrastructure, urban development, regeneration and mineral extraction projects begin? Dealing with the land itself is usually the first step. Groundworks, landform remodelling, re-engineering, remediation and rehabilitation often require the salvaging of habitats, the rescue of animals and plants and the establishing of new environments. The ecological aspects of such works are often undertaken to challenging deadlines and in conjunction with the creation of natural habitats and green infrastructure that is interwoven into the overall design. Vegetation depends on the soils; knowledge of plant communities can enable predictions to be made about the underlying geology and vice versa. Engineering geologists and ecologists work closely together at the start of projects, and their scientific backgrounds can provide a common understanding.

Ground engineering faced a major test during the intensive preparations for the London 2012 Olympics and Paralympics. The banks of the River Lea flowing through the Queen Elizabeth Olympic Park may look natural today but the land form has been engineered with care and planted following experimental trials of methods and species. Such excellence and innovation shows how Atkins can work with clients and others to achieve the best outcomes in terms of design, consents and delivery for urban and infrastructure projects.

People are the key to any business; future talent needs to be identified and allowed to challenge the existing ways of doing things. Business management is about change and maintaining change irrespective of whether it is Atkins as a worldwide consultancy business or the two institutions that we both lead. We hope this book will inspire people to work in collaborative and innovative ways to deliver projects that are truly inspirational.

David Shilston FGS CGeol FRSA
President of the Geological Society of London which was founded in 1807 and is the UK's national society and professional body for the geosciences.

John Box FCIEEM FCIWEM CEcol CEnv
President of the Chartered Institute of Ecology & Environmental Management that was established in 1991 and received its Royal Charter in 2013.

Both David and John are Directors and international specialists at Atkins.

Introduction

Atkins designs and creates habitats that are both functional and beautiful; they add to local biodiversity and are integrated into the local landscape as part of our cities and infrastructure projects. This collection of papers illustrates the breadth of experience and the depth of knowledge that our ecology team has across the world.

Biodiversity is not just beautiful and important for its intrinsic value; it provides the necessary foundations for human existence. From microbes to mountains, natural processes underpin all aspects of life: clean air, water, crops, fisheries, soil formation, climate regulation, decomposition of dead organisms – and a daily dose of nature for our physical and mental well-being.

Collectively, these critical and fundamental processes are called ecosystem services. The relationships between biodiversity and ecosystem services are important. In the UK and international markets, biodiversity inputs to our cities and infrastructure projects are driven by legislation, policy initiatives and the need to minimise risks to projects. Although natural capital and ecosystem services are in their infancy, they are having an increasing influence on future projects. Drivers include the EC Biodiversity Strategy to 2020 which refers to 'no net loss of biodiversity', the UN Convention on Biodiversity, the Intergovernmental Panel on Biodiversity & Ecosystem Services and international funding institutions.

When we have to change land use to accommodate the needs of the human population, for example for agriculture, urban development and transport, we simplify biodiversity and its ecosystems. This reduces the ability of the ecosystems to provide an effective range of services.

Understanding species and habitat processes is crucial to recognising, measuring and quantifying ecosystem services. For example, the provision of clean water depends on the links between ground water, surface water and rainfall within a river catchment and effects on any one of these can affect localised hydrological processes.

Biodiversity is an integral element of economic and social development, and the relationship between the economy and ecosystem services will increasingly engage society, government and regulators. While some businesses may struggle to relate biodiversity to the bottom line, many more are engaging with it. How else will they become market leaders, reduce environmental costs, and increase the efficient use of natural resources?

Atkins' strategy for the future

One of the world's leading infrastructure and design companies, Atkins has the expertise to plan, design and enable the most technically challenging and time-critical projects. Respect for the environment and the importance of environmental issues is integral to all our work. Ecosystems and biodiversity form part of Atkins' future sustainability and urban resilience strategy.

Ecological inputs are underpinned by sound ecological practices and a good evidence base. Ecology is a rigorous and quantitative scientific discipline. Hard data is always required on species and habitats and the effects of environmental changes such as light or noise or pollution on animals and plants. There is a constant desire by our ecologists to push the technical boundaries to develop new biodiversity survey techniques, revise ecological assessment methodologies, and explore novel mitigation measures.

Atkins and ecology

Atkins employs very talented and highly skilled ecologists who collaborate with planners, architects and engineers to design effective solutions, which can be extremely influential in developing exciting and innovative environmentally sustainable projects.

Collected ecology papers

This collection of technical ecology papers – published over the past few years by Atkins' ecologists from the UK and North America – represents Atkins' commitment to providing high quality, forward-thinking advice on biodiversity and ecosystem services.

Acknowledgments

Atkins would like to thank the publishers of the original articles for their help in producing this collected volume.

We are very grateful to our colleagues in Atkins who have written and contributed to the articles in this volume and to Janet Miller, Matt Graydon, Sarah Richards, Steve Bird, David Shilston and Oly Tuck for their expert and generous assistance with editorial, graphics and production matters.

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The Economic Value of Natural Capital and Ecosystem Services

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The Economic Value of Natural Capital and Ecosystem Services

John Box CEnv FIEEM
Atkins Ltd

The environmental services provided by ecosystems contribute to human well-being through the generation of wealth in the broadest sense and through reducing environmental impacts that result in costs to society (Millenium Ecosystem Assessment 2005). Clean air and water, soil formation, climate regulation, sinks for waste, good health and aesthetic benefits depend on ecosystem services provided by properly functioning ecosystems. Land use changes often result in a simplification of ecosystems (for example, agriculture and urban development) thus reducing the ability of the ecosystems to provide a range of services. Environmental damage (such as habitat degradation or pollution) reduces the effectiveness of ecosystem services.

Commercial and financial markets do not yet recognise the full economic value of ecosystem services, nor the biodiversity (or natural capital) from which ecosystem services are derived (see text box below for explanation of the terms). Therefore, the benefits to society of ecosystem services are not reflected in the prices set by the markets for land and for land use changes. As a result, the value of this natural capital and the ecosystem services it supports is inevitably downgraded in decisions about policies, plans and projects. In other words, biodiversity has no economic value and the market value of land with natural habitats and ecosystems is generally very low compared with other land uses. But the real economic contribution of natural capital and ecosystem services to wealth generation and to reducing damaging impacts to society (for example, pollution or flooding) can be great and needs to be fully taken into account in all situations.

Because biodiversity, natural capital and ecosystem services have not been assigned economic values by financial markets, the costs of the loss of ecosystem services are born by society in general whereas the benefits from their exploitation are accrued privately. Environmental costs to society can be corrected by price changes which are usually the result of a combination of resource scarcity, taxes or regulation. For example, the introduction of the landfill tax, the requirement for all new buildings to be modelled to assess their likely CO_2 emissions per unit area, or the vehicle excise duty banding that is dependent on CO_2 emissions. The way forward is for the economic value of natural capital and biodiversity to be accounted for in balance sheets and through financial markets which deal with economic services and manufactured capital. Losses and gains from policies, plans and projects can then be costed. Realistic cost-benefit analyses should inform the introduction of new planning policies and fiscal regimes which could have beneficial or adverse effects on biodiversity and ecosystem services.

Ecosystem services are starting to be costed. The initial results indicate that the economic benefits from the environmental services provided by ecosystems and habitats are very significant (Costanza *et al.* 1997, Balmford *et al.* 2002, Millennium Ecosystem Assessment 2005). The overview of the global value of ecosystem services provided by Costanza *et al.* (1997) estimated that their average value was US$33 trillion ($10^{12}$) per year (1994 prices) within a range of US$16-54 trillion which compared to global gross national product of US$18 trillion per year. This was acknowledged as being a first approximation of the relative magnitude of global ecosystem services.

Biodiversity: The 1992 United Nations Earth Summit in Rio de Janeiro defined 'biological diversity' as the variability among living organisms, and the ecological complexes of which they are part. This includes diversity within species, between species and of ecosystems.

Natural capital: Economic capital refers to the resources employed to produce goods or services that are not themselves significantly consumed (though they may depreciate) in the production process. Natural capital is generally considered to comprise three principal categories: natural resource stocks, land and ecosystems. All are considered essential to the long-term sustainability of development.

Ecosystem services: Biodiversity and ecosystems are closely related concepts. Products of biodiversity include many of the services produced by ecosystems (such as food and genetic resources). An ecosystem is a dynamic complex of plant, animal and microorganism communities as well as the non-living enviroment interacting as a functional unit. Examples of ecosystems include coral reef, rainforest, steppe and urban ecosystems. Ecosystem services are benefits people obtain from ecosystems, which are often undervalued. Examples include pollination, timber, erosion prevention, climate moderation, nutrient cycles and flood alleviation as well as aesthetic and recreational benefits.

Photo A. A surface water balancing lake used by anglers with footpaths around it which has nature conservation and landscape value, and which is likely to offer a premium price for the adjacent houses
Photo: John Box

Balmford *et al.* (2002) took this work further by looking at the economic benefits provided by natural ecosystems and by commercially exploited versions of these ecosystems (using case-studies of logging, aquaculture, drainage for agriculture and blast fishing on reefs). They found that the loss of non-marketed services accruing to society outweighed the marketed marginal benefits of conversion. Put simply, commercial exploitation yields private benefits but the social benefits from unexploited ecosystems are not fully recognised because they are not costed – and the social benefits may be greater than the private benefits. Balmford *et al.* (2002) concluded that the development of market instruments would enable the social and global values of natural ecosystems to be captured through mechanisms such as carbon or biodiversity credits or through premium pricing for ecosystems goods such as fish or timber.

An excellent example is the economic value that the city of Philadelphia (USA) derives from its 10,000 acres (4,000 ha) of park and recreation system including woodlands, rivers and streams, trails, golf courses, picnic areas and playgrounds (Trust for Public Land 2008). The direct income received annually by the city in tax receipts derived from increased property values and tourism was estimated at US$23.3 million. Cost savings of US$16.1 million were due to the open spaces providing storm water management and air pollution mitigation together with reduced anti-social problems through improved community cohesion. In addition, there are estimated savings to the citizens of US$1.1 billion annually from free use of the parks and recreation system combined with savings in medical costs. Finally, the collective wealth of the citizens was estimated to increase by US$729.1 million annually due to increased property values from proximity to parks and profits from tourism. The next step is for these economic values to be translated into the planning policies that Philadelphia will require to maintain and improve these ecosystem services.

Another example is the current open space and environmental services plans for Durban in South Africa which have been developed from pioneering work undertaken in the 1990s (eThekwini Municipality and Local Action for Biodiversity 2007). The open space system covers some 63,000 ha and the estimated value of the environmental goods and services (for example, flood control, water supply) provided by this open space system in 2003 was R3.1 billion (US$ 300 million) which could be compared to the operating budget for the Municipality in 2001/02 of R6.5 billion and the capital budget of R2.8 billion. These plans are being translated into land use policies and a climate change strategy.

The UK Government has determined that the value of services provided by the natural environment needs to be reflected in decision-making and has established a Public Service Agreement (PSA 28) for achieving a healthy natural environment through the delivery of public services (HM Government 2007). The five key indicators for measuring progress on PSA 28 are water quality, biodiversity (wild breeding bird populations), air quality, marine health and agricultural land management which are supported by a broader set of indicators. Evidence that such work on complex issues is being taken seriously is set out in the action plan to deliver PSA 28 (Defra 2007a) and in the clearly expressed arguments in the accompanying guide to valuing ecosystem services (Defra 2007b).

Photo B. A translocated hedge provides maturity and structure to a created landscape, as well as shelter, biodiversity value and a wildlife corridor
Photo: John Box

Ecological resources and biodiversity need to be conserved for their ecosystem functions and this will require new and radical actions. Land use planning and projects from individual developments to major infrastructure projects should not compromise ecosystem services. If ecosystem services are adversely affected, local planning authorities and other regulators giving consents should be able to require developers, whether public or private, to provide for any loss of ecosystem services through appropriate mitigation and/or compensation.

Key ecological resources should always be retained on a site and losses of habitats and ecological features minimised by taking ecosystem services fully into account during project planning and design. Where there is no alternative, habitats and features should be moved to new locations in preference to their loss and subsequent compensation by habitat creation. Ecosystem services provides a powererful argument over and above biodiversity value for retaining ecological resources albeit in a different location. Habitat translocation is a method of last resort, but the technique allows the retention and use of complex and mature ecological resources which can grow and regenerate more quickly than newly planted habitats. Habitats which cannot be retained or translocated should be recreated including the use of innovative techniques such as green bridges to link habitats and green roofs and green walls on buildings.

Compensation for a loss of specific ecosystem services in a location resulting from a particular project could take the form of enhancing ecosystem services elsewhere. For example, increasing or enhancing an existing site of nature conservation value, or aggregating the compensation for a number of developments into one location which is larger and more complex than the sum of on-site mitigation for the developments involved.

In conclusion, economic systems can derive good measures of manufactured capital (machines, buildings) and human capital that are used in sustaining human populations. Economic values need to be assigned to the natural capital that underpins ecosystem services during policy-making, planning or project implementation. In practice, it is the project stage where the real economic costs are crystallised out as part of the pricing, cost-saving and value-engineering processes. Notional costs for loss of ecosystem services should be assigned during project appraisal – and indeed the costed benefits of any ecosystem enhancements that may be derived from a project. The challenge for ecologists and environmental managers is to work with economists to derive the appropriate data for the cost-benefit models.

John Box works for Atkins based in their Telford office. The views expressed in this article are his personal views. The assistance of his colleagues Veronica Lawrie MIEEM and Jules Wynn MIEEM is gratefully acknowledged.

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Building urban biodiversity through financial incentives, regulation, and targets.

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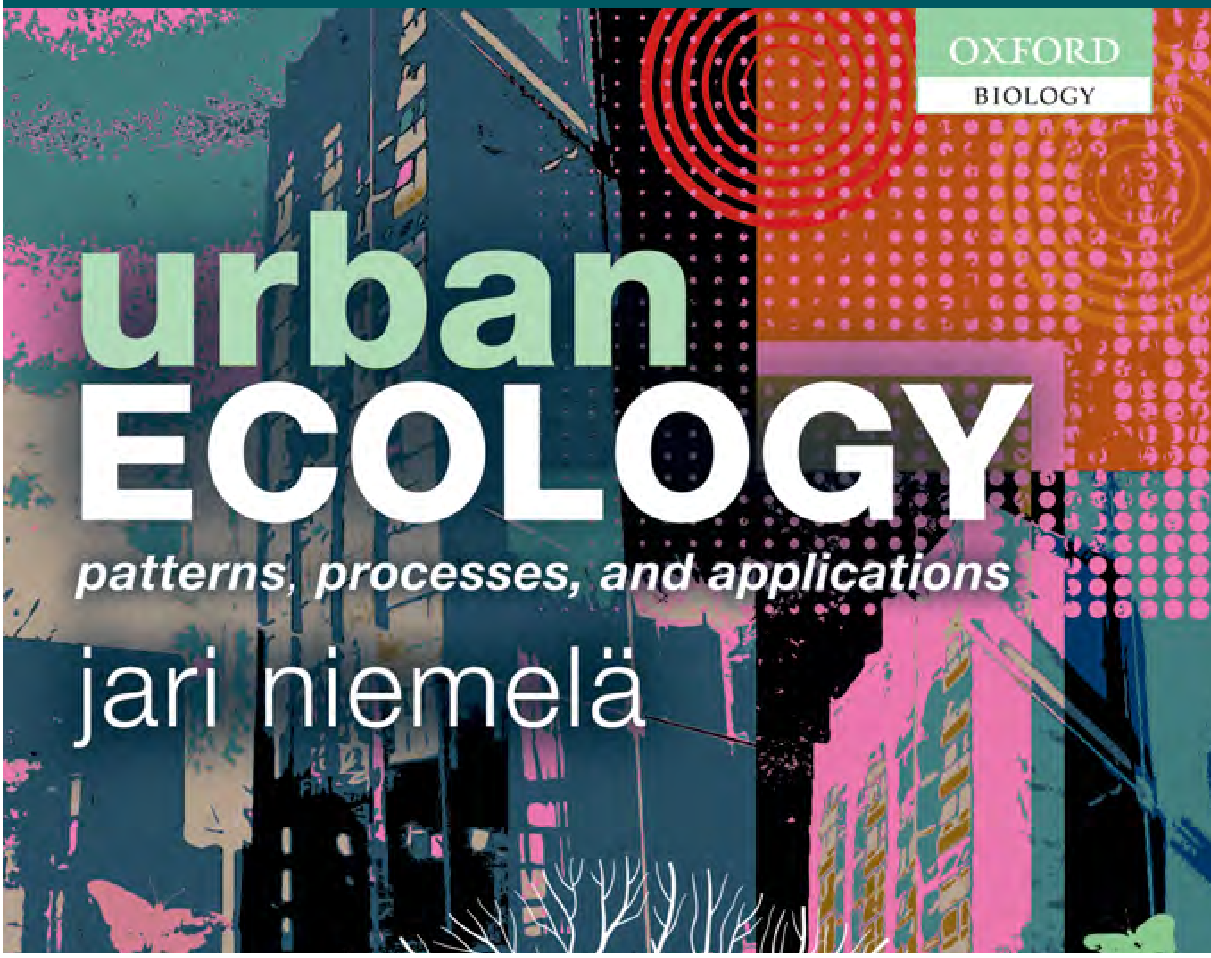

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CHAPTER 5.6

Building Urban Biodiversity through Financial Incentives, Regulation, and Targets

John Box

5.6.1 Introduction

Maintaining, enhancing, and creating biodiversity is a necessity for humans living and working in urban areas. People derive considerable benefits from contact with nature in terms of physical and mental health and well-being (Rohde & Kendle 1997; Douglas 2008; Maller *et al.* 2008; Tzoulas and Greening, Chapter 5.2). However, the benefits are hard to quantify in terms of community health and are usually perceived as being part of a general 'quality of life' factor by individuals living and working in urban areas. Nevertheless, hard evidence of the benefits of contact with nature for personal health is becoming available (Fuller *et al.* 2007; Pretty *et al.* 2007; Mitchell & Popham 2008). Furthermore, there are real economic benefits, albeit usually indirect, from the services provided by ecosystems and habitats—for example, flood regulation, noise reduction, and air quality improvements (Costanza *et al.* 1997; Millenium Ecosystem Assessment 2005; Defra 2007b; Alfsen *et al.*, Chapter 4.3., Bridgewater, Chapter 4.4).

Such benefits from biodiversity are relevant at a community level but do not easily translate into direct financial benefits to individual urban residents or businesses. If they did, there would undoubtedly be a powerful incentive for actions to increase biodiversity in urban areas. Environmental taxation (such as taxes on carbon emissions or on waste to going to landfill) and financial savings (through energy efficiency or reduced water consumption) can offer financial incentives for urban residents and businesses in relation to environmental improvements. But neither of these provide an obvious model for retaining biodiversity in our towns and cities, let alone enhancing or even creating ecosystems and habitats.

Changes in the biodiversity of urban areas can only occur through changes in human behaviour whether as individuals, organizations, or businesses. Drivers for change include 1) economic benefits directly affecting businesses and people; 2) legislation, regulation and official guidance; and 3) targets to encourage individuals and organizations to pioneer change and overcome cultural norms by providing evidence of tangible achievements. The application of such a package of measures to retaining biodiversity in urban areas, let alone increasing biodiversity in our towns and cities, will require a real change in human behaviour that must be driven by economics and financial incentives in combination with legislation, regulation, and targets. Voluntary changes in the behaviour of individuals or organizations will also occur—there will always be pioneers who will make a reality of new ideas. But where there are likely to be economic costs which cannot be set against quantified financial benefits, substantial and lasting changes will not be achieved by voluntary changes in behaviour alone. Individuals and organizations are not generally altruistic and require an environment where all are subject to a comparable financial and legal framework so that an extra economic cost does not result in a financial disadvantage.

5.6.2 Economic drivers to increase urban biodiversity

Economic systems can derive good measures of manufactured capital (machines, buildings) and human capital that are used in sustaining human populations. Biodiversity, natural capital, and the associated ecosystem services (such as climate regulation, water regulation, soil formation, nutrient cycling, and waste treatment) are not fully accounted for in economic systems and commercial markets. Natural capital in this context is the ecosystems themselves together with the atmosphere, water, and minerals required to sustain them.

Ecosystem services can range from simple services such as crop pollination, to more complex services such as soil formation, sinks for waste, and climate regulation (see McDonald and Marcotullio, Chapter 4.1). Humans depend on these services for food, clean water, and clean air. Land-use changes often result in a simplification of the ecosystem, for example to create agriculture, thus reducing the ability of the ecosystem to provide a range of services. Urban areas contain very greatly modified ecosystems and curtailed ecosystem services.

The economic benefits from the environmental services provided by ecosystems and habitats are very significant (Costanza *et al*. 1997; Balmford 2002; Millenium Ecosystem Assessment 2005a). The overview of the global value of ecosystem services provided by Costanza *et al*. (1997) estimated that their average value was US$33 trillion ($10^{12}$) per year (1994 prices) within a range of US$16–54 trillion, which compared to global gross national product of US$18 trillion per year. This was acknowledged as being a first approximation of the relative magnitude of global ecosystem services. A detailed account of characterizing, measuring, and valuing ecosystem services is given by Daily (1997).

Commercial markets do not recognize biodiversity, natural capital, and ecosystem services because they cannot be easily accounted for. The benefits to society of the ecosystem services are not reflected in the prices set by the markets for land and land-use changes. Therefore, the value of this natural capital and the ecosystem services it supports is inevitably downgraded in decisions about policy, plans, and projects. But markets and economic models can price in risk and do respond to scarcity in resources or to 'natural sinks' starting to fill up with pollution. Ecosystem services contribute to human welfare through contributing to the generation of wealth in the broadest sense and through preventing damage that inflicts costs on society. Clean air and water, soil formation, climate regulation, waste treatment, good health, and aesthetic benefits are all dependent on these ecosystem services. Crucially, the contribution of natural capital and ecosystem services to both wealth generation and damage prevention needs to be taken into account in policies, plans, and projects.

Because biodiversity, natural capital, and ecosystem services have not been assigned economic values, they are outside the market and are ignored and treated as externalities, which means their costs are born by society whereas their benefits are accrued privately. Market failures occur through prices not giving the real cost to society of land-use changes or producing goods and services. Markets with failures lead to inefficiency and waste which can be corrected by price changes, taxes, or regulation.

Assigning economic values to ecosystem services is a new concept for both environmentalists and economists alike. Awareness of the environment as one of the three fundamental parts of sustainable development (the other two being economic and social) has required proper engagement with environmental issues. There is an argument that we cannot, or should not, put a value on human lives and biodiversity and ecosystems. But society does place value on human lives through mechanisms such as compensation payments for death and injury; through paying for health and safety measures and construction standards (e.g. for buildings, roads, transport systems); and through cost/benefit models, for example for new roads where notional costs are placed on reducing accidents and saving individual lives.

The effective way forward can only be to assign real economic value in decision-making, whether at the policy, plan, or project stage, to the biodiversity and natural capital that produces these ecosystem services. In practice, it is the project stage where the real economic costs of a project are crystallized as part of the pricing, cost-saving, and value-engineering processes. Notional costs for loss of ecosystem services must be able to be assigned to an individual project during project appraisal—and indeed the costed benefits of any ecosystem enhancements that may be derived from a project. Such an approach requires appropriate data to be added into the cost/benefit models.

Balmford *et al*. (2002) took the work by Costanza *et al*. (1997) further by looking at the economic benefits provided by natural ecosystems and by commercially exploited versions (using case-studies of logging, aquaculture, drainage for agriculture, and blast fishing on reefs). They found that the loss of non-marketed services accruing to society outweighed the marketed marginal benefits of conversion. Put simply, commercial exploitation yields private benefits because social benefits from unexploited ecosystems are not costed. Balmford *et al*. (2002) concluded that the development of market instruments would enable the social and global values of natural ecosystems to be captured privately through mechanisms such as carbon or biodiversity credits, or through premium pricing for ecosystems goods such as fish or timber. Whilst these authors were looking at natural ecosystems in a global context and trying to find market mechanisms to conserve natural ecosystems, their fundamental argument is relevant to urban areas—how are community benefits factored into private cost/benefit models?

The UK Government has determined that the value of services provided by the natural environment needs to be reflected in decision-making. It has established a strategic agreement for the delivery of public services known as a Public Service Agreement (PSA) across the whole of the UK Government (PSA 28: *Secure a Healthy Natural Environment for Today and the Future*) (UK Government 2007). The five key indicators for measuring progress on PSA 28 are water quality, biodiversity (wild breeding bird populations), air quality, marine health, and agricultural land management which are supported by a broader set of indicators. Evidence that such work on complex issues is being taken seriously is set out by the UK Department for Environment, Food and Rural Affairs in the action plan to deliver PSA 28 (Defra 2007a) and in the clearly expressed arguments in the accompanying guide to valuing ecosystem services (Defra 2007b). The environmental cost of incremental development of large natural resources is well illustrated in Defra (2007b) by examples based on the residential development of urban parks.

An excellent example of the results of costing the environmental goods and services provided by open spaces in an urban area has been set out in the pioneering open space and environmental services plans for Durban, in South Africa, which is managed and administered by the eThekwini Municipality (Durban Metropolitan Open Space System (1999); eThekwini Municipality (2001, 2003)). The open space system covers some 63,000 ha and the estimated value of the environmental goods and services supplied by this open space system in 2003 was R3.1 billion (US$300 million), which could be compared to the operating budget for the Municipality in 2001/02 of R6.5 billion) and the capital budget of R2.8 billion. These plans are being translated into land-use policies and a climate change strategy. The process of establishing the open space system and refining it, together with the costing exercise which was undertaken as a contribution to the global exercise of valuing ecosystem services (Costanza *et al*. 1997), is a case study which merits very careful examination (an overview is given by eThekwini Municipality & Local Action for Biodiversity (2007)).

Another and different example is provided by the economic value that the city of Philadelphia (USA) derives from its 10,000 acres (4,000 ha) of park and recreation system including woodlands, rivers and streams, trails, golf courses, picnic areas, and playgrounds (The Trust for Public Land & Philadelphia Parks Alliance 2008). Seven major aspects were included in the valuation – clean air, clean water, tourism, direct use, health, property value, and community cohesion (through being involved in neighbourhood parks). The direct income received annually by the city in tax receipts derived from increased property values and tourism was estimated at US$23.3 million. Cost savings of US$16.1 million were due to the open spaces providing stormwater management and air pollution mitigation, together with reduced anti-social problems through improved community cohesion. In addition, there are estimated savings to the citizens of Philadelphia of US$1.1 billion annually from free use of the parks and recreation system, combined with savings in medical costs. Finally, the collective wealth of the citizens was estimated to increase by US$729.1 million annually due to increased property values from proximity to parks and profits from tourism. These economic values need to be translated into the planning policies that Philadelphia will require to maintain these ecosystem services.

Much can be achieved for biodiversity and people through the promotion of multi-functional urban greenspace where multiple land-uses are recognized (e.g. for the UK, see Barker 1997; CABE 2004; see also Fig. 5.6.1).

The value of such urban greenspaces can therefore be costed in terms of ecosystem services (e.g. flood regulation, air quality amelioration), thus increasing the notional land value of a given urban greenspace. Planning of urban areas with multi-functional urban greenspace or green infrastructure (e.g. TCPA 2004) is fundamental to making cities work to the benefit of those living in them.

Fiscal incentives are required for the inclusion and, crucially, the maintenance of measures such as accessible natural greenspace, biodiversity-friendly sustainable urban drainage systems (SUDS) (e.g. Woods-Ballard *et al*. 2007), green roofs (e.g. Dunnett & Kingsbury 2004), and new habitats—both in new housing and development projects and retro-fitted into existing developments. The review of housing supply undertaken for the UK Treasury (Barker 2004) included a proposal for a planning-gain supplement imposed on development gains accruing to a landowner who receives planning permission. Such a tax would extract some of the windfall gains and recycle them back to local communities—a concept which is consistent with the potential for transference between economic, social, and environmental assets required by sustainable development. Natural England (the statutory nature conservation agency in England) and the Royal Society for the Protection of Birds (RSPB) assessed ways in which such a tax might benefit nature conservation, including discounted tax rates for developments incorporating biodiversity measures (English Nature & RSPB 2006).

5.6.3 Legislation, regulation, and targets to increase biodiversity

Legislation and regulation is usually a last resort when change is required by government but, in relation to biodiversity, official guidance and standards have been found to be inadequate, with individuals

Figure 5.6.1 An example of multifunctional urban greenspace—a surface water balancing lake used by anglers and overlooked by houses whose residents can appreciate wildlife both on the water and in the surrounding wetlands and woodlands. Copyright: John Box (author)

and businesses insufficiently motivated to undertake the changes required to achieve the desired outcome. Whereas legislation legitimizes the changes for all those affected by a new law or set of regulations, guidance will only work if everyone does what is required.

Biodiversity legislation is generally introduced to give protection to rare, threatened, or notable habitats and species—or those species with economic benefits (e.g. fish and fisheries, deer, wildfowl, and game birds). The breadth of biodiversity legislation in terms of the habitats—and particularly the species involved—is increasing in the UK, as is its depth in relation to the severity of the penalties. This is also the situation in a global context as countries industrialize, and competition for land and resources results in the inevitable decline in ecosystems and species to the point where their rarity requires protection through legislation as a result of public pressure. Wildlife legislation in the UK focuses on the protection of rare habitats and species. This legislation and the planning laws in the UK do not yet set limits for permitted changes in the local or even national populations of particular species, or in the extent of certain habitat types in relation to new developments—due either to the initial land-take or to subsequent disturbance from the operation of the permitted development. The use of the word 'yet' is deliberate, because the introduction of such planning legislation will undoubtedly come as a justified response to increasing scarcity in notable species and habitats.

Sustainability has been proposed as a more appropriate mechanism to achieve net gains in biodiversity for individual projects rather than further losses. It is true that any project—industrial, residential, commercial, agricultural, mineral extraction, fisheries—can be assessed in terms of biodiversity and sustainability. The real tests are set by transferring admirable but theoretical concepts, such as sustainability, from the drawing board to the economic realities of the commercial boardroom or the practical realities of the construction site. If there are neither significant economic benefits, nor legislative constraints, admirable concepts rarely survive initial cost/benefit appraisal, profit/loss financial scrutiny, or construction deadlines.

The regulatory framework for urban development needs to move away from *mitigating* biodiversity losses. Instead, it should demand demonstrable biodiversity *gains* over and above requirements for mitigation or compensation, that are formally agreed by an informed regulator whose standards are based on real evidence and good science. Such an approach is consistent with the need to maintain ecosystem services that are dependent on a stock of natural capital.

Loss of ecosystem services needs to be quantified, costed, and regulated. A comparison with climate change and atmospheric CO_2 levels is instructive. The need for global action has been agreed through the Kyoto Protocol and in the UK the economic case has been comprehensively made by the (then) Head of the UK Government Economic Service (Stern 2007). The UK now has the Climate Change Act 2008 which introduces legally-binding commitments for cutting CO_2 emissions, the 2007 Planning Policy Statement on climate change for England, and the radical overhaul in 2006 of Part L of the Building Regulations for England & Wales, which deals with energy conservation and which now requires all new buildings to be modelled to assess their likely CO_2 emissions per unit area (it is envisaged that future revisions of Part L will seek to achieve zero carbon emissions by 2016). The additional costs of such legislation and regulatory mechanisms is accepted as necessary by the business community because the new carbon regime is added onto existing mechanisms, such as building regulations. The quantification of carbon outputs to the atmosphere and its implications provide a sound justification for such new costs to businesses, which can respond through initiatives such as carbon calculators for planning and designing projects (Atkins 2008). The effects of losses and gains in relation to biodiversity, natural capital, and ecosystem services now requires just such a technical and quantified approach if it is to win the hearts and minds of both individuals and businesses.

Voluntary approaches using standards and targets are usually introduced before a government resorts to legislation and regulation. Standards and targets for urban greenspace are essential for the initial stages of planning large-scale development at a regional or subregional scale in order to incorporate green networks and green infrastructure (TCPA 2004), for example, the Green Network of

Telford in the West Midlands of England (Box, Cossons, & McKelvey 2001), the East London Green Grid Framework (Greater London Authority 2008), and the Green Space Plan 2000 for Tokyo (CABE 2004, p. 16). The concept of green networks and green infrastructure needs to be updated to accommodate the ecosystem infrastructure required to support ecosystem services and natural capital.

Urban greenspace provision is usually seen in terms of quantitative standards (unit area of greenspace per resident or household), or accessibility standards (defined areas of greenspace within defined distances from every resident). An example of a quantitative standard is the 31 largest towns and cities in the Netherlands that have agreed on a guideline of 75 m^2 green space per dwelling (van Egmond & Vonk 2007). An example of an accessibility standard is Aarhus, the second largest city in Denmark, where there are standards defined in the Green Structure Plan that no dwelling should be more than 500 m from a green area of at least 6,000 m^2 (reported in CABE 2004, p. 25).

Standards for accessible natural greenspaces—such as no person living more than 300 m from a natural greenspace of at least two hectares in size—have been adopted by the statutory nature conservation agency for England (English Nature 1996; Natural England 2010). Technical and institutional barriers for the implementation of such an urban greenspace model have been identified and a toolkit produced for local authorities, who are envisaged as being the key agencies for applying the targets at a local level through local planning policies and local development frameworks (Handley *et al*. 2003). A broadly similar process is being undertaken in Wales where comparable accessible natural greenspace standards have been established by the Countryside Council for Wales (Countryside Council for Wales 2006) and are being promoted by the Welsh Assembly Government through the environment strategy for Wales (Welsh Assembly Government 2006, p. 42 & 43) and through planning advice for open spaces (Welsh Assembly Government 2009, p. 9).

Some may argue that there is no room for more urban greenspace in crowded urban areas. But why not create these areas? One very public example of this attitude is shown in Fig. 5.6.2, which is the 'green wall' at the CaixaForum culture forum on the Paseo del Prado in Madrid.

Figure 5.6.2 The 'green wall' at the CaixaForum culture forum on the Paseo del Prado in Madrid, which covers some 460 m^2 and supports 15,000 plants from 250 different species. Copyright: John Box (author)

The nature conservation strategy for the metropolitan county of the West Midlands, which included Wolverhampton and the Black Country, Birmingham, and Coventry, pioneered the aim that all residents should have reasonable access to wildlife habitats (West Midlands County Council 1984). The strategy identified 'urban deserts' based on areas where residents were more than 1 km away from accessible wildlife habitats, and a policy of habitat creation was proposed for these Wildlife Action Areas. Such a methodology has been subsequently used in other nature conservation strategies for major urban areas, for example London (Greater London Authority 2002) and Birmingham (Birmingham City Council 1997).

The challenge is for local authorities and public bodies to turn areas that they own, such as mown amenity grassland, into more interesting and stimulating natural greenspace and to ensure that accessible natural greenspace is incorporated into new developments through planning policies and through working with those involved in the new developments. Creative management of biodiversity at the local level is demonstrated on an international scale by the ICLEI (Local Governments for Sustainability) initiative known as Local Action for Biodiversity, which is supported by the UNEP Urban Environment Unit and IUCN.

High urban land values mean that a significant commitment by the landowner is required for land to be designated such that the primary function is nature conservation (in such cases, the landowner is usually a public body, such as a local authority or local council). However, recognized legal mechanisms are available in the UK to link the granting of planning permissions to the provision by the developer of monies for the provision of new roads, new surface water sewers, new schools, and new open spaces. Such provisions should be extended to the creation of new wildlife habitats and ecosystems, and also to the funding of such habitat creation programmes, from the full range of built developments that require planning consent.

Targets and standards can drive such a process along if there is an overall strategy to increase open spaces. For example, some 16,000 ha of public greenspace are proposed to be created by 2013 in the Randstad, the major urban area in the west of the Netherlands that includes Amsterdam, Rotterdam, The Hague, and Utrecht (van Egmond & Vonk 2007). Tokyo has significantly less green open space per person (6.1 m^2/person) than London (26.9 m^2/person) and the Green Space Plan 2000 for Tokyo aims to develop 400 ha of green space by 2015 (CABE 2004, p. 16). Paris has a goal that all citizens can live within 500 m of a green space, which has resulted in a programme to create new green spaces within identified areas of deficiency, including buying derelict houses to create small green spaces (CABE 2004, p. 82–83).

Climate change creates additional factors to be taken into consideration and will require special solutions. The Stern review on the economics of climate change recognized the need for flexible policies whose aim is to reduce fragmentation and encourage movement and migration of species by making use of wildlife corridors (Stern 2007, p. 481). The construction of 'green bridges' across roads and railway lines at key locations would provide a means of reducing habitat fragmentation and would make it easier for species to move in response to climate change. Moreover, Article 10 of the European Community Habitats & Species Directive (Council Directive 92/43/EEC) encourages the management of features of major importance for wildlife, such as those which have a linear and continuous structure or a function as stepping stones and are essential for the migration, dispersal, and genetic exchange of wild species.

5.6.4 Conclusions

Urban design, planning, and land management need to recognize the economic benefits and critical importance of the ecosystem services model together with the natural capital that sustains the ecosystem functions from which the goods and services are derived. More widespread use is required of methodologies for quantifying and costing ecosystem services—and applying them to urban areas where the context relates directly to human needs for services such as flood regulation, noise reduction, or air quality improvement.

Ecosystem services, natural capital, and biodiversity need to be accounted for in balance sheets and commercial markets which deal with economic services and manufactured capital. Losses and gains from policies, plans, and projects can then be costed. Realistic cost/benefit analyses should inform the introduction of new planning policies and fiscal regimes, which could have beneficial or adverse effects on the biodiversity and ecosystem services of urban areas. Ecosystem services are starting to be costed (e.g. Durban, Philadelphia) and the initial results indicate the order of magnitude benefits of these services; developers need to begin to take these into account. Designers and developers are taking notice of costing carbon as part of new projects, and costing ecosystem services and biodiversity losses and gains will start once the values begin to be quantified.

Land-use planning and all projects, from individual developments to major infrastructure projects, should not compromise ecosystem services. Local planning authorities and other regulators giving land-use consents should be able to require developers to cost for loss of ecosystem services without appropriate mitigation and/or compensation. Compensation could take the form of enhancing ecosystem services elsewhere to compensate for a loss of specific services in a location resulting from a particular project.

Ecological resources and biodiversity need to be conserved for their ecosystem functions. This will require new and radical actions during construction and development projects to counteract the inevitable losses of habitats and ecological features which contribute to ecosystem services. There should be a presumption in favour of moving habitats and ecological features, such as hedges, trees, and ponds, to new locations (habitat translocation) in preference to their loss and subsequent mitigation by habitat creation. Habitat translocation retains structural components of ecological resources which can regenerate more quickly than newly planted habitats (Box & Stanhope 2010). Habitats which cannot be retained nor translocated should be recreated within new urban developments, and this should include innovative techniques such as creating green bridges to link habitats and creating green roofs and green walls on buildings (Dunnett & Kingsbury 2004).

Vibrant, innovative national programmes are required in relation to increasing urban biodiversity, with the health, enjoyment, and well-being of all the urban population at their core. Such national programmes should draw on international knowledge and practical experience of biodiversity and urban greenspaces. The ecology of urban areas, the study of the effects of their surroundings on human well-being, and the economics of ecosystem services are relatively new fields of scientific enquiry, and our state of knowledge is incomplete. Beautiful butterflies, dashing dragonflies, and ornate orchids will enthuse local residents and those who work in urban areas. But it is the fiscal incentives, regulation, and targets that have the capacity to really change behaviour amongst the public, developers, local authorities, and statutory bodies in order to provide urban biodiversity and maintain ecosystem services in urban areas.

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John Box

Delivering urban greenspace for people and wildlife

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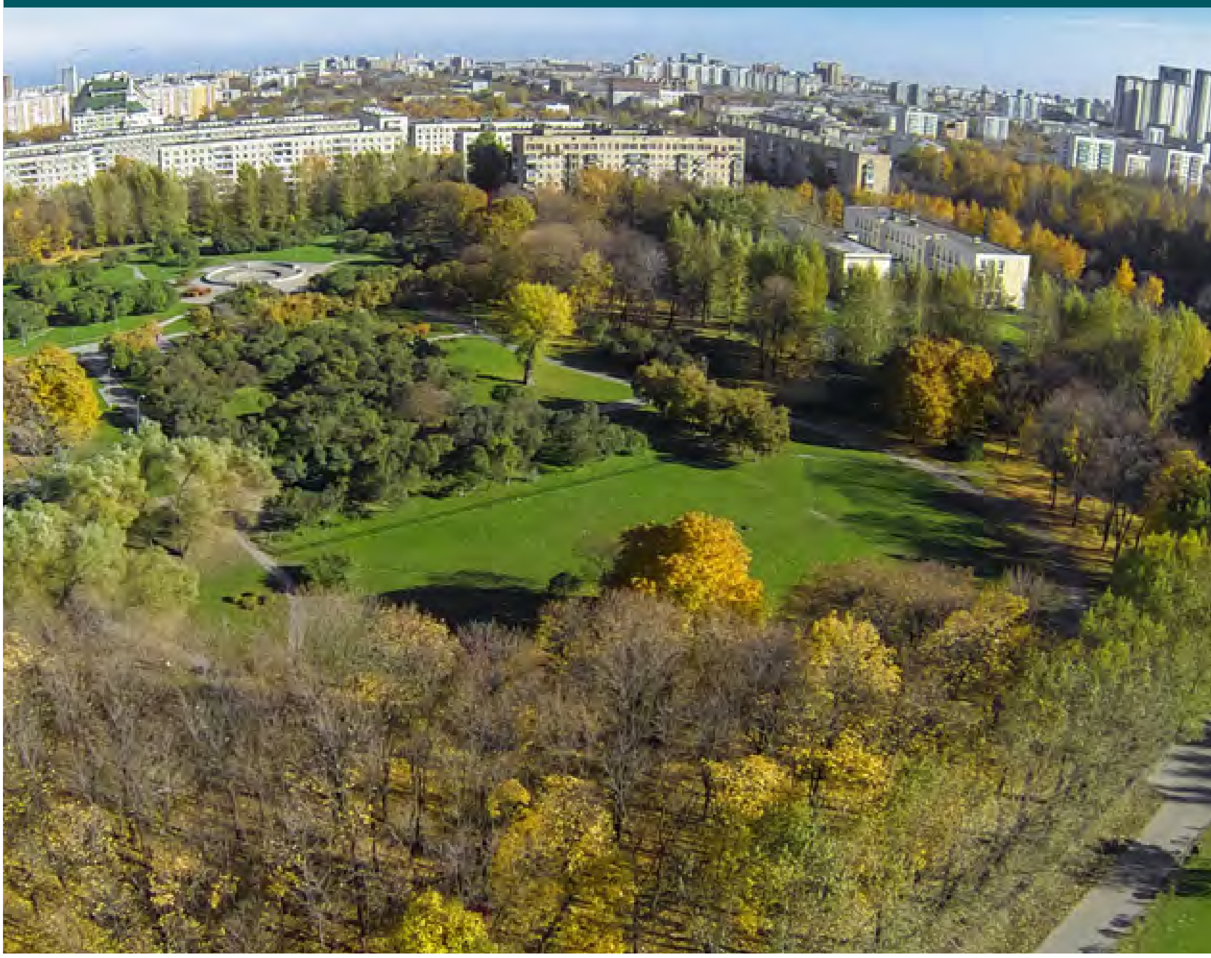

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Delivering urban greenspace for people and wildlife

John Box

Introduction

Greenspace (including parks and wildspaces) in urban areas are the places where people have the contact with nature that is important for well-being and quality of life (Rohde and Kendle 1997; Douglas 2008; Maller *et al.* 2008; see also Chapters 32 and 33 this volume). Therefore, ensuring adequate opportunities for people to come into contact with nature in their everyday lives should result in direct benefits to their health and happiness and hard evidence for this is becoming available (Fuller *et al.* 2007; Pretty *et al.* 2007; Mitchell and Popham 2008).

High urban land values require a significant commitment by the landowner for land where the primary function is nature conservation. It is unusual for a private landowner to set aside land for wildlife purposes due to the high value of urban land and, therefore, most urban natural greenspaces to which the public have access are found on land owned by a public body such as a local authority or local council or a voluntary organisation such as a wildlife trust. But much can be achieved for wildlife and people through the promotion of multifunctional urban greenspace where multiple land uses are recognised (Barker 1997; Commission for the Built Environment (CABE) 2004). The value of such multifunctional urban greenspaces can therefore be costed in terms of environmental services (e.g. flood regulation, air quality amelioration), thus increasing the notional or theoretical land value of a given urban greenspace.

An excellent example of the results of costing the environmental services provided by greenspace in an urban area has been set out in the pioneering open space and environmental services plans for Durban in South Africa, the Durban Metropolitan Open Space System (or D'MOSS) (1999); eThekwini Environmental Services Management Plan (2001, 2003); eThekwini Municipality and Local Action for Biodiversity (2007)].

Urban greenspace provision is usually seen in terms of quantitative standards (unit area of greenspace per resident or household), or accessibility standards (set areas of greenspace within set distances from every resident). For example, the 31 largest towns and cities in the Netherlands have agreed on a guideline of 75 m^2 green space per dwelling (van Egmond and Vonk 2007). Aarhus, the second largest city in Denmark, set a standard defined in the Green Structure Plan that no dwelling should be more than 500 m from a green area of at least 6,000 m^2 (reported in

Figure 41.1 Landscape structure planting around Telford Central station provides a sense of arrival in a green town as well as screening and noise reduction.

CABE 2004: 25). However, quantity and accessibility are not the whole story because the quality of the resource is also significant in terms of the benefits derived by the public. For example, research for the Scottish government into minimum standards for open space has proposed that open space standards should address a qualitative standard as well as a quantitative standard and an accessibility standard (Ironside Farrar Ltd 2005).

Aspirational open space standards are commonplace in strategies, plans and frameworks for guiding the spatial planning of towns, cities and regions. Standards and targets for urban greenspace are ideal for the initial stages of planning large-scale development at a regional or sub-regional scale which incorporates green networks and green infrastructure (TCPA 2004). Examples include the Green Network of Telford in the West Midlands of England (Box *et al.* 2001), the East London Green Grid Framework (Greater London Authority 2008), and the Green Space Plan 2000 for Tokyo (CABE 2004: 16).

The implementation of such open space standards may be planned and strategic or be opportunistic and piecemeal – or some combination of the two depending on circumstances. What is less commonly undertaken are studies of the results of the incorporation of such standards into spatial plans or strategies. The lack of mechanisms for sharing information on implementation is recognised as a weakness by the research undertaken by CABE Space into the experiences in urban greenspace management of eleven cities across the world (CABE 2004: 90).

This chapter examines the evolution of a set of standards for the provision of natural urban greenspace in towns and cities in England and assesses one of them – the supply of designated nature reserves by local authorities – to see how effective it has been over the period from 1993 to 2006.

Accessible natural greenspace standards – a case study

A set of targets (standards or benchmarks) for accessible natural greenspace in towns and cities has been promoted since 1996 by Natural England (previously English Nature), which is the official nature conservation agency in England (English Nature 1996; Natural England 2010):

- an accessible natural greenspace, of at least two hectares in size, no more than 300 metres (five minutes' walk) from home;
- at least one accessible 20 hectare site within two kilometres of home;
- one accessible 100 hectare site within five kilometres of home;
- one accessible 500 hectare site within ten kilometres of home;
- statutory Local Nature Reserves at a minimum level of one hectare per thousand population.

These targets for ensuring access by people living in towns and cities to places of wildlife interest include a mixture of quantitative and accessibility standards, as well as a qualitative standard in Local Nature Reserves which are a statutory designation where the primary land use must be

Figure 41.2 A wooden dragon constructed by children at Plants Brook LNR in Birmingham shows that all forms of wildlife can be appreciated even in a high quality designated site.

nature conservation which is managed both for its inherent qualities and also for enjoyment by the public and local residents. These targets are derived from the UK tradition of urban planning, open-space hierarchies and recreational standards and also take account of the emerging understanding of the ecology of urban areas and the need to conserve important wildlife habitats and geological features.

The term 'natural greenspace' in the targets includes the full range of richness and diversity of urban greenspaces which can range from small sites awaiting redevelopment and which have been colonised by spontaneous assemblages of plants and animals to much larger areas such as the substantial islands of countryside surrounded by urban development found in most urban areas in Britain. Such urban greenspaces are likely to be multifunctional by providing ecosystem services (such as flood regulation, amelioration of temperature, noise and air quality), recreational areas, landscape quality and places for plants and animals to live.

These targets in relation to people and wildlife in urban areas were novel in the UK when they were first published in 1993 (Box and Harrison 1993) and the intellectual foundation was undertaken by work undertaken through the UK Man and the Biosphere Urban Forum (www.ukmaburbanforum.co.uk). Subsequent research refined the targets (Harrison *et al.* 1995) which were adopted by the statutory nature conservation agency for England (English Nature 1996, 2004; Natural England 2010) and their use disseminated in various publications (Barker 1997; TCPA 2004). Technical and institutional barriers for the implementation of such an urban greenspace model have been identified (Handley *et al.* 2003). A toolkit was produced for local authorities (Handley *et al.* 2003) who are envisaged as being the key agencies for applying the targets at a local level through local planning policies and local development frameworks.

A broadly similar process is being undertaken in Wales where accessible natural greenspace standards established by the Countryside Council for Wales (Centre for Urban and Regional Ecology 2002; Countryside Council for Wales 2006) are being promoted by the Welsh Assembly Government through the environment strategy for Wales (Welsh Assembly Government 2006: 42 and 43) and planning advice for open spaces (Welsh Assembly Government 2009: 9). These accessible natural greenspace standards are not yet given such official recognition in Scotland where there is guidance on greenspace quality (Greenspace Scotland 2008). However, research for the Scottish government into minimum standards for open space (Ironside Farrar Ltd 2005) proposed that open space standards should address a qualitative standard, a quantitative standard and an accessibility standard – these standards are addressed by the accessible natural greenspace standards used in England and Wales.

Setting targets is the easy part. Their implementation by local authorities may be visible at a local level but they are hard to monitor at a regional or national level because of a lack of appropriate mechanisms in the UK. However, one target – the provision of designated nature reserves by local authorities in England – can be measured over time because the data is collected both locally and nationally.

Local Nature Reserve is a statutory designation in the UK whose origin lies with the enjoyment of nature (Wild Life Conservation Special Committee 1947; English Nature 1991). Local Nature Reserves (LNRs) are designated by local authorities and can be chosen to reflect local priorities as opposed to the national priorities reflected in the selection of National Nature Reserves (Barker and Box 1998). Indeed, local authorities can hold the view that LNRs should be established because the natural features of a site are of special interest 'by virtue of the use to which the public puts them for quiet enjoyment and appreciation of nature' (English Nature 1991: 3). The current position in respect of LNRs across the UK is set out by Box *et al.* (2007).

The target for statutory LNRs at a minimum level of one hectare per thousand population is a simple and appealing measure that allows local authorities to establish a nature reserve on a formal statutory basis on land that they own, lease or over which they have a long-term management agreement. Funding to assist this programme was established by English Nature through the *Wildspace!* grants programme for LNRs in 2001, financed largely by a National Lottery award from the New Opportunities Fund (now the Big Lottery Fund) under its Green Spaces and Sustainable Communities programme. By the time the programme ended in October 2006, almost £7 million of *Wildspace!* grants had been spent to encourage more and better LNRs to be established by projects working for people, places and nature (English Nature 2005).

Local Nature Reserves are best seen as nodes in multifunctional green networks. Such a view places them in a landscape context, values them as part of the environmental resources of the county or district and draws attention to their excellence as sites of nature conservation value (Barker 1997). LNRs are usually identified in the local development plans produced by local authorities in the UK which have a statutory basis and demonstrate the existing land uses in the area of a local authority as a means of guiding the location of future developments. The demonstration of a positive landuse for LNRs has important practical benefits by clearly indicating that there is no potential for other land uses, such as built development, on these sites. Such a positive land use allocation helps to move away from the idea, particularly in urban areas, that nature conservation only occurs on land which has no other use or which no one wants.

The original article by Box and Harrison (1993) set out data from a sample of 25 urban local authorities in England whose provision of LNRs in 1993 ranged from 1 ha of LNR for 889 residents (Canterbury) to 1 ha of LNR for 170,500 residents (Camden). This baseline dataset was updated just over ten years later with data on the number and area of LNRs in each of the same sample of 25 urban local authorities as at December 2006 (Table 41.1) (Box 2007).

By 2006, there were significant improvements in the supply of Local Nature Reserves with some local authorities achieving order of magnitude or even greater increases in their provision over a period of little more than a decade (Barnet, Derby, Gloucester, Leicester, Newcastle-upon-Tyne). Of these, Leicester City Council has increased its provision of LNRs by a factor of 67 from 1 ha for 135,300 residents in 1993 to 1 ha for 2,014 residents in 2006. For some local authorities, the population has increased but the total area of LNRs has remained essentially unchanged and the provision per thousand residents has therefore actually decreased (Haringey, Portsmouth, Southampton, Southwark). It is notable that the provision in Leeds has remained static since 1993 at just over 1 ha for 1,100 local residents, but the total area of over 600 ha of LNRs in Leeds in 1993 was far ahead of other local authorities in England at that time and still remains exceptional.

Conclusions

Standards, targets and guidelines can all be used to turn policy into practice. But implementation is the key to real success and legislation and regulation or financial incentives are the most effective drivers. Costing the environmental services provided by multifunction greenspaces in urban areas can be an effective way of countering arguments that built development is required to realise the inevitably high urban land values (e.g. eThekwini Municipality and Local Action for Biodiversity 2007 and associated references; The Trust for Public Land and Philadelphia Parks Alliance 2008). Another driver can be competition – in this case, increasing the supply of Local Nature Reserves in some towns and cities through monitoring of the results of a sample of local authorities over a period of time – as long as the results can be published in places where the results can be readily seen by the target organisations, in this case the planning journals that local authority planners read.

Table 41.1 Provision of Local Nature Reserves in a selection of urban local authorities in England in 1993 and 2006

Local authority	*1993*			*2006*			
	Population[1]	*LNRs (total area, number)*[2]	*Population/ area*	*Population*[3]	*LNRs (total area, number)*[4]	*Population/area*	*Comments*
<1,000 residents per ha LNR (in 2006)							
Gloucester	91,800	4.3 ha (2)	21,349	109,885	169.5 ha (7)	648	Large improvement and achieved target
Canterbury	127,100	143 ha (3)	889	135,278	177.7 ha (10)	761	Improving and achieved target
Wakefield	306,300	313 ha (7)	979	315,172	401.5 ha (10)	785	Improving and achieved target
Norwich	120,700	52.5 ha (5)	2299	121,550	136.2 ha (8)	892	Improving and achieved target
Stoke-on-Trent	244,800	82 ha (1)	2985	240,636	246.4 ha (9)	977	Improving and achieved target
Range 1,000:1 to 5,000:1 (2006)							
Dudley	300,400	181.7 ha (4)	1653	305,155	274.6 ha (7)	1111	Improving and target in sight
Leeds	674,400	605.4 ha (5)	1114	715,402	613.0 ha (8)	1167	Static – but there was a very large area of LNR in 1993
Sandwell	282,000	30.3 ha (2)	9307	282,904	205.8 ha (9)	1375	Large improvement
Coventry	292,500	48 ha (3)	6094	300,848	216.7 ha (14)	1388	Improving
Derby	214,000	9.3 ha (1)	23,011	221,708	143.2 ha (7)	1548	Large improvement
Portsmouth	174,700	119 ha (1)	1468	186,701	119.0 ha (1)	1569	Getting worse

Plymouth	238,800	105 ha (5)	2274	240,720	146.1 ha (7)	1648	Improving
Peterborough	148,800	51.4 ha (2)	2895	156,061	81.2 ha (5)	1922	Improving
Barnet	283,000	4.9 ha (1)	57,755	314,564	158.5 ha (6)	1985	Large improvement
Leicester	270,600	2 ha (1)	135,300	279,921	139.0 ha (7)	2014	Large improvement
Newcastle-upon-Tyne	263,000	8 ha (1)	32,875	259,936	113.0 ha (6)	2300	Large improvement
Liverpool	448,300	21 ha (1)	21,348	439,473	134.1 ha (3)	3277	Large improvement
Hereford	49,800	6.1 ha (2)	8164	50,149	14.4 ha (3)	3483	Improving
Range 5,000:1 to 10,000:1 (2006)							
Haringey	187,300	36.2 ha (3)	5174	216,507	32.6 ha (3)	6641	Getting worse
Southwark	196,500	29.9 ha (1)	6572	244,866	32.4 ha (4)	7558	Getting worse
Birmingham	934,900	39.5 ha (4)	23,668	977,807	102.6 ha (7)	9530	Large improvement
Range 10,000:1 to 50,000:1 (2006)							
Southampton	194,400	14 ha (1)	13,886	217,445	14.0 ha (1)	15,532	Getting worse
Oxford	109,000	2.2 ha (2)	49,545	134,248	6.4 ha (3)	20,976	Improving
Islington	155,200	2.5 ha (1)	62,080	175,797	5.3 ha (3)	33,169	Improving
Greater than 50,000:1 (2006)							
Camden	170,500	1 ha (1)	170,500	198,020	1.85 ha (4)	107,038	Improving

Notes
1 Population data are preliminary 1991 Census figures (*Whitaker's Almanac* 1993).
2 LNR areas and numbers for April 1993 (English Nature data).
3 Population data are 2001 Census figures.
4 LNR areas and numbers for December 2006 (Local Authority data).

Like air and water, wildlife is assumed to be a free resource that we take for granted and which can be adversely affected without direct economic payment. However, the protection and continued enjoyment of natural resources does entail costs to individuals and to society. Legislation, planning guidance and public attitudes are continually driving the burden of these costs away from the victim and the taxpayer and onto the consumer and the shareholder where they rightfully belong. Over the past 40 years the conservation of nature in Europe has focused on the protection of rare habitats and species, rather than on the overall losses of biodiversity due to specific developments. Wildlife legislation has not yet set limits for changes in species or populations in relation to the development of individual sites – either due to the initial land-take or to subsequent disturbance. Increased emphasis on environmentally sustainable development may offer a more appropriate mechanism to achieve net gains in biodiversity for individual projects.

Environmentally sustainable development demands that environmental capital is not diminished from one generation to the next. The next generation will only know what it finds and will not be able to fully comprehend past losses. Therefore, important urban greenspaces that are rich in wildlife need systems which can deliver good site management in order to maintain the quality of the resource in the long term. Large sites are more likely to be able to accept multiple use without damage and can provide a greater variety of opportunities for local people to use and enjoy. But in many urban areas the severe constraints of high land values and existing land uses mean that only small sites are practicable as urban greenspaces. It is increasingly being recognised however that even very small urban greenspaces are valuable not only in terms of their ecological and educational benefits but also in supporting more sustainable communities, for example through their contribution to people's health and well-being.

Some may argue that there is no room for more publicly accessible urban greenspace in crowded urban areas. But why not create these areas? A number of nature conservation strategies in the UK have recognised the concept of areas which are deficient in wildlife habitats to which the public have reasonable access. Indeed one of the main aims of the nature conservation strategy produced 25 years ago for the metropolitan county of the West Midlands (West Midlands County Council 1984) was to ensure that all residents had reasonable access to wildlife habitats. The strategy identified 'Urban Deserts' based on areas where residents were more than 1 km away from accessible wildlife habitats; habitat creation was seen as being very important in these areas which were called 'Wildlife Action Areas'. Such a methodology has been used in other nature conservation strategies for urban areas, for example London (Greater London Authority 2002) and Birmingham (Birmingham City Council 1997).

The challenge is for local authorities and public bodies to turn places that they own, such as mown amenity grassland, into more interesting and stimulating natural greenspace and to incorporate accessible natural greenspace into new developments through spatial planning and through working with those involved in the new developments.

Creative management of biodiversity at the local level is demonstrated on an international scale by the ICLEI (Local Governments for Sustainability) initiative known as Local Action for Biodiversity which is supported by the UNEP Urban Environment Unit and IUCN (www.iclei.org/index.php?id=lab). Tokyo has significantly less greenspace per person (6.1 m^2/person) than London (26.9 m^2/person) and the Green Space Plan 2000 for Tokyo aims to develop 400 ha of green space by 2015 (CABE 2004: 16). Paris has a goal that all citizens can live within 500 m of a greenspace which has resulted in a programme to create new greenspaces within identified areas of deficiency, including creating small greenspaces by buying derelict houses (CABE 2004: 82–3). Some 16,000 ha of public greenspace are proposed to be created by 2013 in the Randstad, the major urban area in the west of the Netherlands that includes Amsterdam,

Figure 41.3 Regularly mown grassland in the Town Park in Telford in 1990.

Figure 41.4 The same area in 2005 showing scrub habitats created by natural regeneration after mowing ceased.

Rotterdam, The Hague and Utrecht (van Egmond and Vonk 2007). Rennes in France has pioneered a programme of differential management regimes for greenspace in the city which is based on ecological awareness and results in the development of wildlife areas within the city, control of water and soil pollution and increasing biodiversity while reducing the overall cost of open space management.

In conclusion, standards for environmental quality and targets for enhancing and protecting biodiversity which include both wildlife and people – such as those for accessible natural greenspace in the UK – can be powerful levers for change and their use to influence behaviour should not be underestimated.

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Briefing: Keeping up with the Suds revolution and legislative evolution

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Briefing: Keeping up with the Suds revolution and legislative evolution

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The past ten years has seen progressive integration of sustainable drainage systems into the fabric of development and planning policy across the UK. With increasing urbanisation, exceptional flood events and the potential changes in our climate behaviour, there is a need for careful management of surface water and consideration of its quantity, quality and a symbiotic need to combine this with amenity and biodiversity within a hardening, simplified environment. From the implementation of the water framework directive, through to the recent implementation of the Flood and Water Management Act, sustainable drainage systems have evolved from a conceptual, novel idea to a design of principal importance for most new developments. This paper reviews the progression of sustainable drainage systems from a little-applied soft drainage technique to an essential feature of flood risk assessments and planning policy through responsive and strategic legislative development that also lead to new roles for all flood management authorities.

A combination of severe drought conditions (2004–2006) and three significant flood events (1998, 2000 and 2007) that affected large areas of the UK has raised the profile of flood risk and sustainable water management over the past 12 years. During this time we have also seen continuing urban expansion and growing concerns about climate change, particularly as we develop a greater understanding of the implications of both. The importance of 'sustainable development' and the increasing need to manage surface water has re-emphasised the importance of sustainable drainage systems (Suds). Whilst the fundamental designs of some Suds are not necessarily new (Kirby, 2005), the legislative mechanisms by which they are now being required for all types of development are evolving at a pace.

By definition, Suds incorporate a portfolio of flow control techniques that allow urban runoff to be managed in a more sustainable manner including measures to keep flows above ground longer than traditional piped systems. Suds seek to incorporate runoff into the urban form rather than see it as a nuisance, although in some catchments traditional drainage solutions may still be the most sustainable system for effective surface water management. Understanding catchment drainage characteristics is crucial to drainage design.

Suds mimic natural drainage processes to facilitate infiltration (where ground conditions are suitable) and provide storage or attenuation above or below ground, depending on the design. They can contribute to peak flow reduction while also improving water quality through particulate removal or pollutant reduction. Above-ground solutions can create diverse wetland features in otherwise urbanised areas. The main processes Suds are designed to manage involve 'quantity, quality, biodiversity and amenity' (Woods-Ballard *et al*., 2007).

Since its introduction, the water framework directive (WFD) (European Community, 2000) has influenced the way in which the quantity, quality and biodiversity of our water environment are managed, by promoting sustainable development and a greater consideration of flood risk and the future effects of climate change. The role of Suds in contributing to improved water quality aspects is just part of the applicable legislative framework.

Planning policy guidance note 25 (PPG25) (DTLR, 2001) was constructed with flood risk management and climate change in mind but was consistent with WFD ideals. It promoted the 'precautionary principle' for selecting development sites, advocated sustainable development and better management of surface water through the use of 'soft sustainable drainage systems.' It was, however, understood that these drainage systems were only as effective as the standard to which they were designed. At the time, policies advising the use of Suds required further development and the issues of adoption and future maintenance were seen as constraints to their wider implementation.

Later, 'Making space for water' (Defra, 2005) set out the government's 20-year plan for a holistic approach to fluvial flood and coastal erosion risk management. It emphasised the importance of a coordinated approach to land use, planning policy and urban design and highlighted the developing importance of flood risk assessments (FRAs) in coordinating effective planning for sustainable development.

Planning Policy Statement 25 (PPS25) replaced PPG25 in 2006 (Communities and Local Government, 2006) and outlined the roles and responsibilities for developers, regional planning bodies, local planning authorities (LPAs) and the Environment Agency (EA). At the time, amendments to article 10 of The Town and Country Planning (General Development Procedure) Order 1995 meant that LPAs were required to consult with the EA on all (except minor) development in fluvial or coastal flood risk areas and on any development exceeding 1 ha.

The role of the FRA has since provided a mechanism for LPAs and the EA to promote Suds and initiate pre-planning application liaison with developers. As Suds policies have been increasingly promoted within regional and local strategies/plans and strategic FRAs in line with PPS25, the inclusion of Suds at the earliest stages of development and planning has become an essential part of planning policy. Developers are increasingly expected to include Suds and a 'lack of space in the design' is not an acceptable reason for their exclusion.

The increased consideration of flood risk and surface water issues to this point would seem to have been timely foresight given the flooding events witnessed in 2007. The Pitt Review (Pitt, 2008) that followed was a comprehensive review of the events, and among its 92 recommendations it stated the government should decide where Suds responsibilities lay and that partnerships should be developed between all flood risk authorities. The report asked probing questions about Suds that would need addressing if surface water issues were to be adequately managed without duplication or inconsistency in the future. Many of the report's recommendations were or have already been implemented, although it was understood that others would require changes in the law to take effect.

The government's water strategy for England, 'Future Water' (Defra, 2008) set out plans and practical steps for securing a sustainable approach to the way we use and are affected by water to find a balance between people and nature. The importance of the 'water cycle' from rain drops to treated sewage effluent were clear drivers, as were the implications of climate change, but it also recognised that no regulatory authority as yet had an overarching responsibility for Suds.

In April 2009, the Draft Flood and Water Management Bill was released for public consultation to demonstrate the government's commitment to the previous reports and policies (Defra and Welsh Assembly Government, 2009). The Bill set out reasons why changes to the law were required for the management of our water environment. It also responded to the WFD and the EU floods directive (European Community, 2007) for the assessment of flood risk from watercourses (river basin) and coastlines, with a focus on prevention, protection and preparedness, which was subsequently transposed into English and Welsh law as the Flood Risk Regulations (Defra, 2009).

The Flood and Water Management Act received Royal Assent on 8 April 2010 and some elements were implemented in September (Defra, 2010a), with other sections to come into effect at a later date following current and future consultations. The Act not only consolidates the importance of Suds within developments but it will be, by far, the most influential in terms of formalising the roles and responsibilities of developers and flood management bodies as it progresses to become law.

The key proposed legislative mechanisms for Suds are detailed within part 2 of the Act (under the section entitled miscellaneous), but Defra (Department for Environment, Food and Rural Affairs) has produced a series of briefing notes to summarise what the Act will mean for each management body.

Unitary and County Councils will form the lead local flood authorities (LLFA) with a coordination role to manage flood risk in their areas that is consistent with local and national strategies (Defra, 2010b). LLFAs will investigate all flooding incidents and maintain a register of significant flood management structures, with powers to undertake works for surface or groundwater flood risk. Perhaps most importantly, the LLFA will establish its own Suds approving body (SAB) to review and approve Suds in all developments (to ensure compliance with forthcoming 'national standards' for Suds). Once approved, the LLFA will be required to adopt and maintain Suds that serve more than one property.

All other regulatory bodies (local and district councils, the EA, internal drainage boards (IDBs), highways authorities, water and sewerage companies) are to be referred to as 'risk management authorities' (RMAs). Highways authorities will be responsible for including and maintaining Suds in public roads that meet the national standards (Defra, 2010b). The EA will have an overview of all flood and coastal erosion risk management and will be required to develop a national flood and coastal erosion risk management strategy for England and a separate strategy for Wales. IDBs will be required to act consistently with local strategies with regard to the management of watercourses in their control and have key roles in partnership with other RMAs (Defra, 2010c).

For developers (Defra, 2010d), all new developments or re-developments will require Suds that meet the national standards for approval by the SAB. A 'non-performance bond' may be payable to ensure that, should the construction not meet the standards, it will not cost the taxpayer. The SAB will also consult with the sewerage undertaker before making its determination and adoption agreements will be required between the developer and sewer providers.

Defra recognises the potential burdens that the Act may mean for LLFAs and is committed to funding new duties and training, and to publishing a clear way forward (Defra, 2010e). The draft national standards for Suds is imminently due for consultation and the Construction Industry Research and Information Association (Ciria) will continue to develop and update its invaluable Suds guidance manuals and its model agreement procedures to help steer us through the growing and evolving legislative requirements. A key starting point for most would be *The Suds Manual* (Woods-Ballard *et al*., 2007) and the new planning for Suds document (Dickie *et al*., 2010), with further insights into their management through information on model agreements (Shaffer *et al*., 2004).

In 2001, the Institution of Civil Engineers reported that people with appropriate Suds and modelling skills were lacking within the industry (Fleming *et al*., 2001). Since then, the development of both has been swift and expertise has emerged and been synthesised with skills from a range of professions to keep pace with the changing needs of planning policies. PPS25, along with its accompanying practice guide (Communities and Local Government, 2008), form living documents that are continuously being updated to reflect developing policies and improvements in computer modelling of rivers, rainfall-runoff and climate change. The 2010 updates (Communities and Local Government, 2006; revised 2010) refine flood zone definitions and vulnerability classifications and with the advancement of the Act, these documents will no doubt be updated again soon.

As we move towards a more coordinated and formalised approach to managing surface water flood risks and improving water quality to contribute to the achievement of WFD targets, the specific needs for Suds expertise within LPAs will need to expand. Indeed, some have already created specific roles for Suds specialists in readiness. Councils will need to assess their existing capacity, skills and resources to develop viable options to enable fulfilment of their duties that will be required by the new legislation.

The Suds revolution of the past decade has been significant, as has the evolution of the legislation and policy drivers for its usage across England and Wales. It is now apparent that there is still a lack of long-term knowledge on the longevity of Suds and the potential future maintenance and management implications for adopting bodies, despite the development of National Standards. The potential cost burden at a time of substantial spending cuts across the economy signifies a real need to promote partnership working among all flood management bodies to maximise efficiencies and effectiveness for future implementation wherever possible.

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Assessing the Environmental Impact: Are Stormwater Ponds More Effective Than Presumed?

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Assessing the Environmental Impact: Are Stormwater Ponds More Effective Than Presumed?

David A. Tomasko, Emily H. Keenan, Shayne Paynter, and Megan Arasteh

Stormwater detention is a serious concern for communities in Florida. On average, the state receives between 50 and 65 in. of rainfall every year, from about 120 storms. The drinking water for more than 90 percent of Floridians comes from groundwater, so pollutant loads from runoff (which commonly include such chemical nutrients as nitrogen and phosphorus) must be managed to prevent them from entering the water supply.

A key element in stormwater management is the design and construction of wet detention ponds, which have been found to be an affordable and viable system for pollutant removal. But exactly what quantity of pollutants can wet detention ponds remove? Are wet detention ponds any more effective than is currently believed? Will new nutrient-removal requirements proposed by the Florida Department of Environmental Protection (FDEP) be too stringent or too costly for wet detention ponds to comply?

The study described here attempts to answer these questions.

Regulatory Background

The FDEP has developed draft rules that, if implemented, would require many current stormwater treatment systems to be modified (FDEP, 2010). To avoid water quality violations, these same rules may require more stringent nutrient (nitrogen and phosphorus) removal, particularly in areas where downstream waters have been "verified impaired" due to nutrient-related water quality concerns.

Under the modified rule, a minimum level of stormwater treatment would be required to meet the new performance standards. One of the following two options would be required:

1. An 85 percent reduction of the postdevelopment average annual loading of nutrients from a site.
2. A reduction in nutrient loads such that the postdevelopment average annual nutrient loading would not exceed the amount expected from the site's former natural landscape.

Currently, wet detention systems for stormwater treatment are designed and permitted under the assumption that the volume of water they receive during storm events can be held on site for enough time to reduce incoming loads of total nitrogen (TN) by approximately 30 percent. But, based on conventional assumptions, the proposed new stormwater rules would make it nearly impossible for wet detention ponds—an affordable and widely used stormwater treatment system in Florida—to comply.

In addition, the current regulatory guidelines for "impaired water" require nutrient loading calculations to demonstrate no additional impairment by any proposed construction project, which can require larger, more expensive, ponds. In some cases, the required pollution reduction is greater than a wet pond can provide, which can require the construction of a dry pond or other more costly options.

A study was conducted in an attempt to determine if wet detention ponds, *as currently designed*, are:

- More effective in pollutant removal than is commonly assumed.
- Able to comply with the intent of FDEP's proposed new performance criteria.

Figure 1. Tampa Bay Region of Florida Showing Hillsborough Bay and Tampa Bay.

Nutrients and Water Quality Impacts

All water bodies in Florida are evaluated by either the U.S. Environmental Protection Agency (EPA) or FDEP to assess their water quality status. An excess of nitrogen and/or phosphorus can result in the overproduction of phytoplankton (algae), which is measured in units of chlorophyll-a (a pigment found in all plants).

With the adoption of numeric nutrient concentration criteria (NNC) by both FDEP and EPA, water bodies are now characterized based on chlorophyll-a and nutrient concentrations *combined*. In estuarine systems, nitro-

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gen is the typical nutrient of concern, as is the case in the Tampa Bay region (Figure 1). But in freshwater systems, phosphorus is normally the greater concern.

Nitrogen, which is generally more difficult to remove by means of stormwater detention ponds, was the focus of the study.

A body of water and its stormwater inflows can be characterized by TN concentration. The TN can be subdivided into two broad categories: dissolved inorganic nitrogen (DIN) and organic nitrogen (ON). The DIN is made up of three primary forms of nitrogen: ammonium, nitrite, and nitrate. These forms are readily available for assimilation by phytoplankton populations, which have mechanisms that enable the direct uptake and assimilation of these nitrogen forms into compounds such as the nitrogenous bases of DNA, amino acids (the building blocks of proteins), and the photosynthetic pigment chlorophyll-a (Seitzinger et al., 2002; Bronk et al., 2006; Urgun-Demirtas et al., 2008).

However, the dominant form of nitrogen in stormwater runoff is ON—not DIN—and ON is not readily or immediately available for use by phytoplankton (Seitzinger et al., 2002; Bronk et al., 2006; Urgun-Demirtas et al., 2008). The ON can be further subdivided into two categories:

- *Particulate organic nitrogen* (PON) is comprised of small organisms (alive and dead), fragments of organisms, and organic debris—all of which are greater than 0.45 microns in size. The PON is not readily available for biological assimilation until those particulate forms are first broken down and their nitrogen becomes biologically available. Such processes can take days, weeks, or even months.
- *Dissolved organic nitrogen* (DON) is a mixture of compounds less than 0.45 microns in size, such as amino acids and tannins. Depending on their specific characteristics, DON components may eventually become available for phytoplankton uptake, in which case DON is considered "labile." On the other hand, when its components do not become available for algal uptake, DON is considered "recalcitrant."

In a tidally flushed system such as Tampa Bay, the degradation processes necessary for DON to become DIN may take longer than the average amount of time a given water mass resides in the bay. In Barnegat Bay, N. J., a scenario was outlined (Seitzinger et al., 2002) whereby the residence times of river waters discharging into the bay were less than the period over which DON would become biologically available. As a result, both the PON fraction and some of the larger DON compounds would not be in the bay long enough for their nitrogenous compounds to become available for algal uptake and assimilation.

Thus, DIN is likely to have a greater affect on algal growth in well-flushed water bodies. Stormwater treatment systems—and the regulatory basis for stormwater treatment rules—are therefore best considered in light of the differing abilities of DIN, DON, and PON to stimulate algal growth.

In assessing the implications of this information, it was concluded that nitrogen loading models that focus only on TN are likely to overestimate the biological impacts of modeled nutrient loads, as not all forms of nitrogen within the TN category are equally able to stimulate algal growth. It was similarly concluded that nutrient-loading models that consider only DIN are likely to underestimate biologically available nutrient loads because they do not consider the role of labile DON.

The implications of differences in the biological availability of different nitrogen forms—and how these forms of nitrogen are modified in typical stormwater treatment systems—should therefore be considered when establishing any need to adjust the regulatory criteria related to stormwater treatment ponds.

Thus, it is possible that the impacts of discharging treated stormwater into well-mixed water bodies such as Tampa Bay *could be less than expected*. With this possibility in mind, a study was designed to answer two primary questions:

1. Do wet detention ponds managed by the Florida Department of Transportation (FDOT) in the Tampa Bay region remove DIN and TN at rates similar to what has been previously documented?
2. If they do, does the elevated rate of DIN removal mean that water leaving these stormwater treatment ponds has less of an impact to receiving water bodies than would be predicted based solely on TN reduction rates?

Nutrient Removal Efficiencies in Stormwater Treatment Systems

Smith (2010) summarized the nitrogen makeup of more than 900 Florida stormwater samples. The average sample predominantly contained DON (69 percent of TN by mass), with DIN making up the remaining 31 percent. Those numbers compare favorably with values found (Rushton et al., 1997), where DON made up 72 percent of TN by mass, with the remaining 28 percent in the form of DIN (Table 1).

In examining previous assessments, it was found that typical wet detention stormwater ponds reduce TN concentrations by about 32 percent and reduce DIN concentrations by 68 percent (data from Southwest Florida Water Management District, 1997, and Johnson Engineering, 2009a, 2009b, 2006, and 2008). Also, various FDEP-developed total maximum daily load (TMDL) reports indicate that wet detention ponds are expected to reduce stormwater TN loads by about 30 percent (FDEP, 2008). This expected load-reduction efficiency is similar to that found in the pollutant loading assessments developed for the Sarasota Bay National Estuary Program (Heyl, 1992) and the Charlotte Harbor (Coastal Environmental Inc., 1995) National Estuary Program (Table 2).

In addition, the Tampa Bay Estuary Program (1996) concluded that although stormwater treatment ponds are highly effective in reducing sediment and toxin loads, "...

Table 1. Nitrogen makeup of Florida stormwater samples

Study	DON concentration of TN (average sample by mass)	DIN concentration of TN (average sample by mass)
Smith (2010)	69 percent	31 percent
Rushton et al. (1997)	72 percent	28 percent

Table 2. Typical nitrogen reductions of wet-detention stormwater treatment ponds

Study	Reduction in TN concentration	Reduction in DIN concentration
1. SWFWMD (1997) 2. Johnson Engineering (2009a, 2009b, 2006, 2008)	32 percent	68 percent
3. FDEP (1997) 4. Heyl (1992) 5. Coastal Environmental Inc. (1995)	30 percent	N/A

wetland retention/detention is not as effective for reducing nitrogen."

While these conclusions are accurate, the *form* of nitrogen in stormwater ponds is just as important as the total amount of nitrogen, if not more so, which renders such conclusions incomplete.

Current evaluations of the effectiveness of stormwater treatment ponds focus *only* on TN removal. However, DIN and labile DON are the nitrogen forms that are more biologically relevant to phytoplankton production.

Stormwater Treatment Pond Efficiencies: Biologically Relevant Versus Total Loads

As an example of the potential difference in presumed efficiencies, consider a hypothetical scenario in which 100 metric tons (MT) of nitrogen enter a stormwater treatment pond. As noted in Table 2, the widely accepted expectation is that about 30 percent of that load would be reduced through in-pond processes such as burial, uptake by littoral vegetation, denitrification, and so forth. Therefore, an estimated 70 MT of nitrogen would be left in the pond (100 - 30 = 70).

Of the 70 MT of TN leaving the pond in outflows, studies show that about 10 percent (7 MT) would be expected to be in the form of DIN, with the remaining 63 MT in the form of DON (Rushton et al., 1997). Of the 63 MT of DON, the amount of DON that would ultimately be considered biologically available for phytoplankton uptake would be 30 percent, on average (Wiegner et al., 2006). Therefore, the amount of labile DON would be about 19 MT (0.3 x 63 = 18.9, rounded to 19).

The result is that 26 MT of TN would potentially be biologically available for phytoplankton assimilation, or 19 MT of labile DON plus 7 MT of DIN (Figure 2). Considering that 100 MT of total nitrogen entered the pond in this hypothetical example, a typical stormwater treatment pond could convert 100 MT of TN into 26 MT of potentially available nitrogen in its discharge, which yields an nutrient-reduction efficiency of *74 percent*, not the widely accepted value of 30 percent that is used in loading models and other guidance documents.

Therefore, existing and planned stormwater treatment ponds may be more efficient at reducing nutrients than their presumed efficiencies would suggest, which means that the impact of treated stormwater on algal populations in a well-mixed water body (such as Tampa Bay) could be minimal.

This hypothesis is based on the following logic:

- Water discharging from stormwater treatment ponds has much lower levels of inorganic nutrients than water entering such ponds.
- The organic forms of nutrients that characterize the majority of nutrients discharged from these ponds are much more refractory than inorganic forms of nutrients.

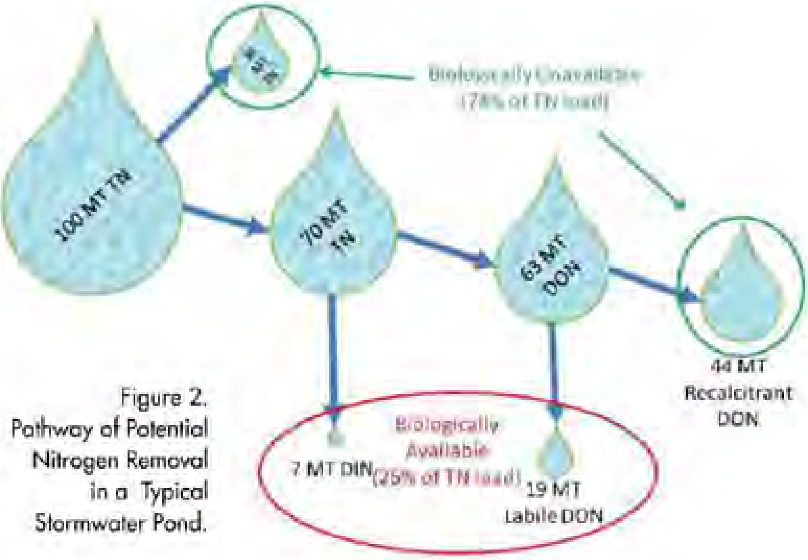

Figure 2. Pathway of Potential Nitrogen Removal in a Typical Stormwater Pond.

A Study of Wet Detention Ponds in Florida's Tampa Bay Region

In 2011, the FDOT District 7 funded a study in Florida's Tampa Bay region to test the real-world nutrient-removal efficiency of stormwater detention ponds.

One of the study objectives was to quantify the biologically relevant nutrient-removal efficiency of typical wet detention ponds; the method used was to measure phytoplankton responses to nutrient additions from both direct and treated stormwater runoff.

Three stormwater ponds where selected for the source of incubation waters from both inflows and outflows (ponds referred to herein as D, 3S, and 1). The drainage basin for each pond was comprised solely of transportation infrastructure. Each stormwater pond is located in Tampa and discharges to a portion of Tampa Bay (Figure 1). Samples from Hillsborough Bay (a subsection of nitrogen-limited Tampa Bay) were used to represent receiving water and potential phytoplankton responses to treated and untreated stormwater runoff, and the study itself was comprised of several project phases.

Phase I: Determining Methodology

Phase I of the study consisted of evaluating methodologies (using data from only Pond D) and refining techniques, one of which was used for the remainder of the study. Pond D stormwater inflows were collected during a rain event on March 2, 2011. A fixed volume of Hillsborough Bay water (365 ml) was inoculated with various quantities of water (1 to 50 ml) from Pond D inflow and incubated for various periods of time (8 to 24 hours) while suspended in the water column of Hillsborough Bay (Figure 3).

Both the initial and final chlorophyll-a concentrations from each inoculation/incubation scenario were evaluated to identify the best methodology to use for the rest of the study. Based upon that initial evaluation, 15 ml of stormwater inflow or outflow to inoculate 365 ml of Hillsborough Bay water were used, with an incubation period of 24 hours (Figure 4).

Phase II: Ensuring the Methodology Addresses the Hypothesis

In Phase II, the ability of the preferred technique to address the proposed hypothesis was assessed, namely, that *stormwater ponds treat water in such a way that biologically relevant nutrient-load reductions exceed presumed efficiencies*. Phase II used stormwater inflow and outflow collected on March 29, 2011 (again, using data from only Pond D).

Phase III: Testing Stormwater Inflow/Outflow From Three Ponds

Phase III determined the DIN, DON, and chlorophyll-a concentrations in stormwater inflow and outflow from three ponds (Ponds D, 3S, and 1) using samples collected on Aug. 29, 2011 (during Florida's "wet" season) and Oct. 9, 2011 (the beginning of Florida's "dry" season).

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Figure 3. Study Apparatus With Bottles Used to Suspend Stormwater Samples in Water Column (left). Apparatus With Samples Incubating in Hillsborough Bay (right).

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Phase IIIM: Testing Filtered and Unfiltered Stormwater From Two Ponds

Finally, a modified version of Phase-III testing—called Phase IIIM—was conducted using filtered and unfiltered stormwater inflow and outflow from only two ponds (1 and D); the samples were collected on Jan. 11, 2012 (in the middle of Florida's dry season).

For Phases III and IIIM, the initial nutrient and chlorophyll-a concentration of sample bottles were quantified and the bottles were then suspended in the water column of Hillsborough Bay for the 24-hour incubation period. After incubation, the final nutrient and chlorophyll-a concentrations of each bottle were measured.

Results and Discussion

Nutrient concentrations of the stormwater pond inflow and outflow were measured during Phases II, III, and IIIM (Table 4). These measurements reveal that:

1. Inflows were dominated by DON, not DIN. This suggests that TN loads from road runoff are mostly comprised of nitrogen forms that are not as biologically available for algal assimilation as nutrient loads with higher DIN contents.
2. Pond outflows became even more dominated by DON as DIN was removed from the water column.
3. The FDOT ponds reduced DIN concentrations at rates consistent with existing literature.

The DIN concentrations were greatest in the inflow when compared to the outflow of the ponds for all sampling events (Table 4). In pond inflows, DIN comprised from 13 to 46 percent of TN. On average, DIN comprised 29 percent of the TN load.

Outflow DIN represented between 1 and 8 percent of the TN load, with an average of 3 percent. Water discharging from the ponds contained a much smaller percentage of TN, in the form of the more biologically available DIN fraction.

Phytoplankton Response to Stormwater Input

Initial and final chlorophyll-a concentrations were measured during every phase of the stormwater inoculation study. The results suggest that neither inflows nor outflows to and/or from the ponds tested were consistently capable of stimulating phytoplankton growth in bottles filled with ambient water from Hillsborough Bay (Table 5).

For the seven events where pond inflows were tested, chlorophyll-a concentrations increased three times. But, for two of those times, the increase was 2 µg/L or less—a value not much greater than the detection limit itself.

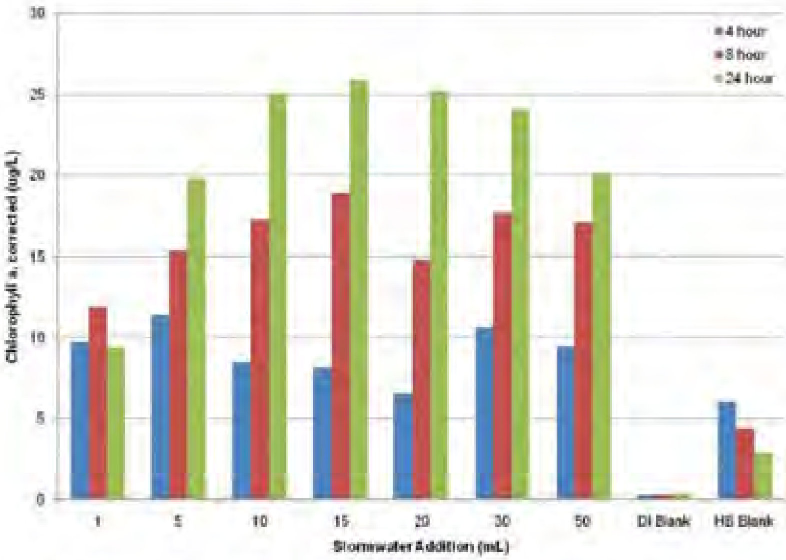

Table 3. Chlorophyll-a Results From Phytoplankton Response Evaluation Experiment ["DI Blank" refers to "laboratory blank" samples containing no runoff or bay waters; "HB Blank" refers to "experimental blank" samples from Hillsborough Bay containing no added runoff].

For the seven events where pond outflows were tested, chlorophyll-a concentrations also increased three times, but for only one of those times was the increase 2 μg/L or less. A phytoplankton response as a result of bay-water inoculation using pond outflow was observed during the Phase-III sampling event in October 2011, which is what led to the Phase-IIIM portion of the study.

Evaluating the Role of Pond Phytoplankton on Incubation Bottles

It was thought that the unexpected phytoplankton response observed using stormwater pond outflows in October 2011 could be caused by the presence of saline-tolerant phytoplankton from the ponds that continued to grow in the estuarine waters within the incubation bottles. Commonly, phytoplanktons found in stormwater ponds are from the class *Cyanophyceae*, specifically in the genera *Microcystis* and *Oscillatoria* (Wanielista et al., 2006; Drescher, 2011), which have been reported to survive in both freshwater and marine environments, with some species being able to tolerate salinity ranges as broad as 0 to 30 ppt (Liu, 2006; Dube et al., 2010; Mur et al., 1999; Tonk et al., 2007).

An evaluation of nitrogen concentrations from Hillsborough Bay during the August and October 2011 experiments indicates significantly more inorganic nitrogen in the bay in October, when a chlorophyll-a increase was observed in the incubation bottles; DIN values of 0.06 mg/L were measured in August, but the value rose to 0.40 mg/L in October. It was surmised that the phytoplankton response observed in October was likely related to saline-tolerant phytoplankton from the stormwater pond assimilating the abundant supply of inorganic nitrogen in Hillsborough Bay.

To determine if their assumption could explain the October 2011 results, the authors performed a modified version of their Phase-III experiment (Phase IIIM). Before incubation, inflow and outflow samples were filtered using a syringe and 0.45 micron filter to remove phytoplankton.

The final chlorophyll-a concentrations measured during Phase IIIM showed a lack of phytoplankton response when using filtered pond outflow samples (Table 6). Specifically, outflow from Pond 1, which exhibited elevated initial chlorophyll-a concentrations (75 μg/L) during Phase III, showed a decrease in chlorophyll-a concentration when compared to unfiltered inoculation. This was expected, and is consistent with the hypothesis that phytoplankton from the stormwater ponds had grown in incubation bottles during the October 2011 experiment. These results support the contention that stormwater pond discharges did *not* cause the growth of phytoplankton within Hillsborough Bay during the October 2011 experiment. Rather, pond algae grew in October as a result of elevated DIN levels in Hillsborough Bay.

Table 4. Average DIN, DON, and DIN concentrations for stormwater pond inflow/outflow (study Phases II, III, and IIIM).

Phase	Date	Pond	No. of samples	Inflow DIN (mg/L)	Inflow DON (mg/L)	Inflow Percent DIN	Outflow DIN (mg/L)	Outflow DON (mg/L)	Outflow Percent DIN
II	Mar 2011	D	3	0.42	0.78	35	0.07	2.11	3
III	Aug 2011	D	5	0.22	0.80	22	0.03	0.89	3
III	Aug 2011	3S	5	0.63	1.19	34	0.07	0.74	8
III	Aug 2011	1	5	0.44	0.66	40	0.07	1.90	3
III	Oct 2011	D	5	0.03	0.12	22	0.05	1.46	3
III	Oct 2011	3S	5	0.07	0.26	20	0.02	1.29	1
III	Oct 2011	1	5	0.02	0.14	13	0.09	1.64	5
IIIM	Jan 2012	D	5	0.62	0.74	46	0.01	0.94	1
IIIM	Jan 2012	1	5	1.01	2.91	26	0.03	1.32	3

Table 5. Average initial and final Chlorophyll-a concentrations from Phases II and III after inoculation of Hillsborough Bay water with stormwater pond inflow/outflow.

Phase	Date	Treatment	No. of samples	Pond inflows Chlorophyll-a (μg/L)		Pond outflows Chlorophyll-a (μg/L)	
				Initial	Final	Initial	Final
II	Mar 2011	Hillsborough Bay (HB) with Pond D	3	13	15	9	10
III	Aug 2011	HB/Pond D	5	22	12	21	12
III	Aug 2011	HB/Pond 3S	5	19	14	21	11
III	Aug 2011	HB/Pond 1	5	20	21	21	15
III	Oct 2011	HB/Pond D	5	13	11	15	58
III	Oct 2011	HB/Pond 3S	5	13	20	13	14
III	Oct 2011	HB/Pond 1	5	11	8	18	34

Table 6. Average initial and final chlorophyll-a concentrations measured in Phase IIIM after inoculating Hillsborough Bay water with stormwater pond inflow/outflow

Phase	Date	Treatment	No. of samples	Pond inflows Chlorophyll-a (μg/L)		Pond outflows Chlorophyll-a (μg/L)	
				Initial	Final	Initial	Final
III(M)	Jan 2012	Hillsborough Bay with Pond 1	5	5	7	8	8
III(M)	Jan 2012	HB/Pond 1 (filtered)	5	5	7	4	4
III(M)	Jan 2012	HB/Pond D	5	4	6	5	4
III(M)	Jan 2012	HB/Pond D (filtered)	5	3	5	2	4

Conclusion

Taken together, the study results suggest that:

- The findings complement existing literature, confirming that wet stormwater detention ponds reduce TN concentrations by approximately 30 percent—but they also reduce DIN concentrations by more than *80 percent.*
- In most cases, stormwater runoff from transportation land use was not able to stimulate algal growth in Hillsborough Bay.
- Also, in most cases, water discharging from FDOT stormwater ponds was not able to stimulate algal growth in Hillsborough Bay. However, there is evidence that algal populations in these ponds include genera with fairly wide salinity tolerance levels, and it is possible for these algae to survive in the higher-salinity waters of Hillsborough Bay—at least over a 24-hour period.

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The results indicate that the three FDOT ponds studied may be more efficient at reducing downstream environmental impacts than their presumed TN load-reduction efficiency of 30 percent. It appears that additional modifications to wet stormwater detention pond designs may not be needed—at least for FDOT projects—because the ponds may be better at removing biologically relevant forms of nutrients (average of 86 percent) than is currently assumed.

Most importantly for FDOT, wet detention ponds appear to provide sufficient environmental benefits when taking into account the biologically relevant forms of nitrogen in stormwater runoff that are found in well-flushed and nitrogen-limited water bodies. Given the hydraulic grade-line limitations faced by many FDOT projects, the ability to use wet detention ponds offers substantial cost savings over other stormwater detention solutions.

Proposed statewide stormwater rules could require detention solutions to remove 85 percent of nitrogen loads. The good news is that wet detention ponds appear to be able to comply with the *intent* of Florida's proposed new rules—if the conversion of nitrogen into biologically less available forms is considered. That, in turn, may eliminate the need to build larger, more costly dry retention ponds or dry-wet treatment trains in many (if not most) situations.

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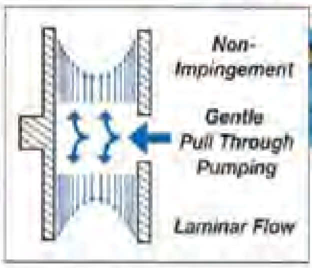

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Experimental studies on the effects of nutrient loading and sediment removal on water quality in Lake Hancock

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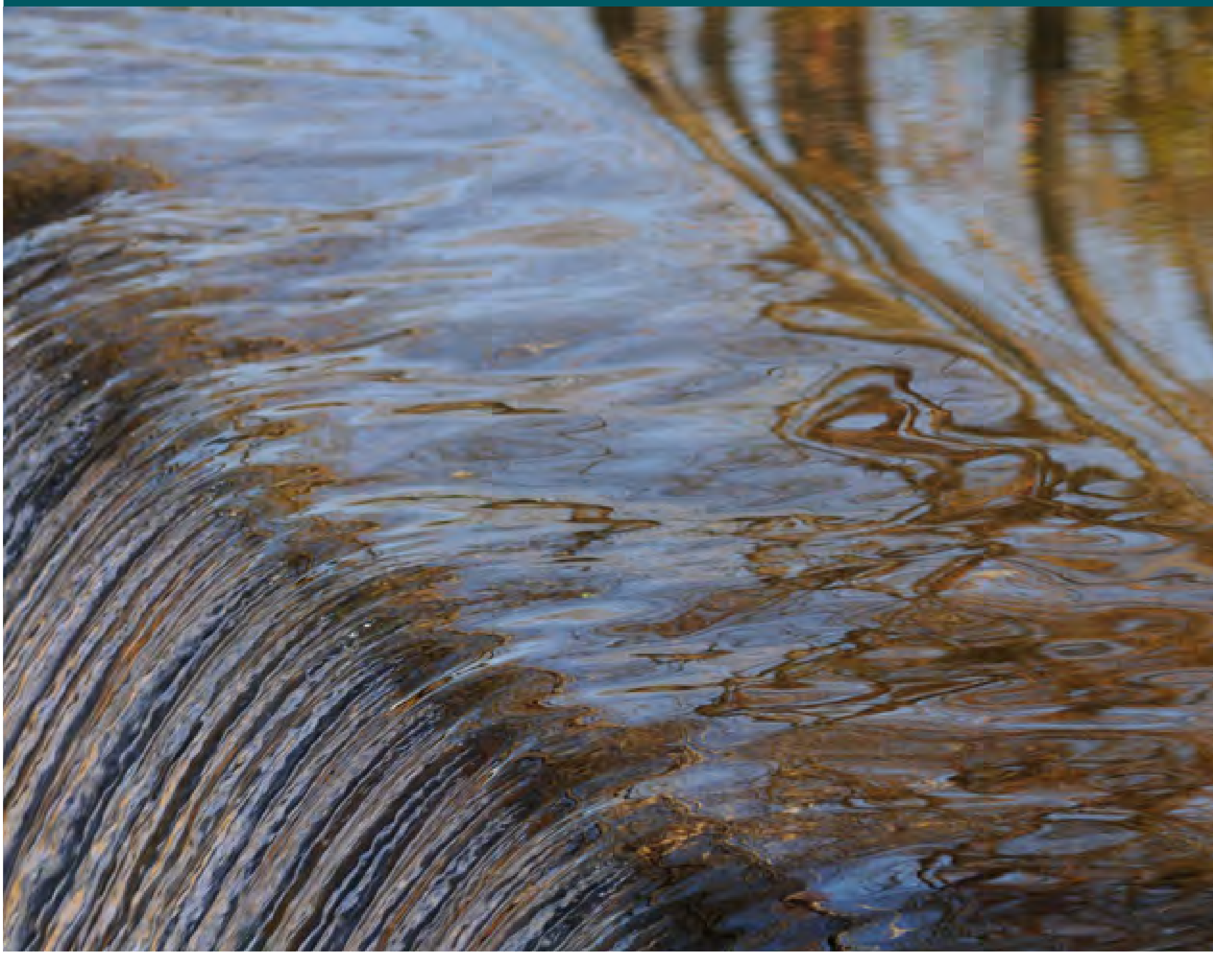

EXPERIMENTAL STUDIES ON THE EFFECTS OF NUTRIENT LOADING AND SEDIMENT REMOVAL ON WATER QUALITY IN LAKE HANCOCK

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Abstract: *Lake Hancock is situated in the headwaters of the Peace River, and is identified as having an impact on water quality within the Peace River and Charlotte Harbor. Water quality in Lake Hancock has been verified as impaired for nutrients by the Florida Department of Environmental Protection, due to levels of total nitrogen, total phosphorus and biological oxygen demand that exceed the State of Florida's threshold values for impairment. This study examined the potential benefits of sediment removal on water quality in Lake Hancock. The project included two field experiments: (1) a manipulative experiment to determine if a reduction in nutrient availability could result in decreases in blue-green algal vigor and/or abundance and (2) a manipulative mesocosm experiment designed to evaluate whether partial removal of the lake's sediments would result in improvements in water quality. The findings were consistent with the hypothesis that Lake Hancock is a net sink for phosphorus loads that accumulate in the sediments and the lake is a net source of nitrogen which is exported from the lake. Only by reducing TP loads from both external and internal loading sources could TP levels be reduced to biologically relevant levels.*

Key Words: Lake Hancock, hypereutrophic, cyanobacteria, sediments, pollution, Charlotte Harbor

Lake Hancock, with a surface area of approximately 18.4 km^2, is the third largest lake in Polk County, Florida (ERD, 1999). The contributing watershed is approximately 339 km^2 in size, for a watershed to open water ratio of 18:1. The major tributaries to Lake Hancock are the Banana Creek sub-basin (55 km^2), the Lake Lena Run sub-basin (48 km^2 acres) and the North Saddle Creek sub-basin (198 km^2). The only discharge from Lake Hancock is South Saddle Creek, which discharges to the Peace River which then flows 171 kilometers to Charlotte Harbor.

Lake Hancock has been characterized as having "poor" water quality, using the State of Florida's Trophic State Index (TSI), since at least 1970 (Polk County, 2005), and concerns over poor water quality in the lake have existed as far back as the 1950s (ERD, 1999). More recently, Lake Hancock's water quality was verified as impaired for nutrients based on data collected between January 1997 and June 2004 (FDEP, 2005). Concentrations of total nitrogen

(TN), total phosphorus (TP) and biochemical oxygen demand (BOD) exceeded the State of Florida's threshold screening values by considerable amounts (FDEP, 2005). Poor water quality in Lake Hancock has prompted a number of reports focusing on strategies to improve its condition.

Zellars-Williams Company (1986) developed a nutrient budget and restoration strategy for Lake Hancock that pointed to the potential benefits of sediment removal on lake water quality. In 1999, the consulting firm ERD completed a nutrient budget and water quality modeling effort that concluded that even if nutrient loads from base flow and stormwater runoff were reduced by 75 percent, water quality in the lake would remain "poor" based on TSI values (ERD, 1999). In that study, it was concluded that an estimated 50 percent reduction in nutrient loads from baseflow and stormwater, in combination with a whole-lake dredging effort, could potentially improve the mean TSI value to 70, the threshold between "fair" and "poor" water quality. The report also highlighted the significant reductions in pollutant loads to the Peace River that would be expected to occur by various outfall treatment alternatives. As an example, the least effective outfall treatment alternative investigated by ERD (1999) would still reduce nitrogen loads leaving the lake by $>$ 90,000 kg/year, an amount greater than the 20 year horizon for population-growth related nitrogen increases expected for the entire Peace River watershed (81,500 kg/yr; Coastal Environmental, Inc., 1995).

In 1999, a draft report titled "Lake Hancock Habitat Enhancement Plan" was prepared by the Florida Fish and Wildlife Conservation Commission (FFWCC, 1999). A restoration plan was included in the report and a number of activities, including re-establishing historical aquifer levels to restore seepage into North Saddle Creek, and re-establishing native vegetation along the lake's shoreline and in the lower reaches of Banana Creek, were identified as in the ERD (1999) report. This plan also highlighted the need to construct a downstream treatment wetland for lake discharges, particularly during the latter phases of lake restoration efforts, when the lake's sediments would be removed and transported offsite.

In a report conducted for Polk County and the Florida Department of Environmental Protection (FDEP) by CDM (2002), a variety of lake restoration techniques were assessed. A number of options were developed, and the highest-rated options all included the following components: 1) wetland treatment of discharges from the lake, 2) chemical treatment/settling pond treatment of inflows from North Saddle Creek, 3) wetland treatment of inflows from Banana Creek, and 4) passive drawdown and mechanical excavation of bottom sediments in nearshore areas. Regardless of the techniques employed to restore water quality within the lake, CDM (2002) concluded that the wetland treatment of lake discharges component should be incorporated into any potential restoration strategies, as it would both further improve the quality of water leaving the lake, and also provide an "environmental safeguard" for potential downstream impacts from activities such as hydraulic dredging of the lake's sediments.

Using methods outlined in the Impaired Waters Rule (IWR- Florida Administrative Code 62.303) the FDEP concluded that both Lake Hancock and Lower Saddle Creek were impaired for nutrients and dissolved oxygen (DO). Lake Hancock was verified as impaired in 2003 for nutrients based on an annual average Trophic State Index value of 83.5. This value which considers nitrogen, phosphorus and chlorophyll *a* (Chl-a), exceeded the IWR threshold value of 60 during the verification period (January 1, 1997 to June 30, 2004). The median values for TN, TP, and BOD for the lake for this period were 4.46, 0.49, and 9.9 mg/liter, respectively (FDEP, 2005). In comparison, 90 percent of Florida lakes have a median TN or TP concentration below 2.5 and 0.29 mg/liter, respectively (FDEP, 2005). Ninety percent of Florida streams have a median BOD below 5.1 mg/liter (FDEP, 2005). In addition to being listed as impaired for nutrients, FDEP listed Lake Hancock as impaired for DO because more than 19 percent (27/140) of DO values collected during the verification period were below the freshwater DO standard of 5 mg/liter (FDEP, 2005).

The poor water quality of Lake Hancock is due either directly or indirectly to the very high levels of phytoplankton in the lake. In the lake, values of Chl-a can reach levels as high as 800 µg/liter (Parsons Water & Infrastructure, 2005). In 2004, the annual average Chl-a level in Lake Hancock was 152 µg/liter (Polk County, 2005) more than seven times higher than the 20 µg/liter level that FDEP uses to classify lakes as "impaired" for nutrients.

Based on criteria developed by the State of Florida, ratios of TN to TP by weight are used to infer the nutrient that is most likely to limit algal biomass. Ratios of 10 or less suggest limitation by nitrogen, ratios greater than 30 suggest limitation by phosphorus, and ratios between 10 and 30 suggest co-limitation. Recent data (Polk County, 2005) show an annual average TN:TP ratio of 9.6. At three sites visited monthly, Parsons Water and Infrastructure (2005) reported TN:TP ratios that ranged between 8.6 and 8.7. And in a report that focused on the isotopic signature of Lake Hancock's sediments, Brenner and co-workers (2002) concluded that these sediments appeared to be associated with algal detritus from nitrogen-fixing cyanobacteria (aka blue green algae) from the overlying water column.

These results suggest that nitrogen is the nutrient limiting algal production in Lake Hancock and that the "surplus" nitrogen exported from the lake is due to nitrogen-fixation by the lake's high levels of blue green algae (Brenner et al., 2002). Not all the surplus nitrogen is exported from the lake, as Brenner and co-workers (2002) found that sediment accumulation in the lake was a significant destination for an unknown portion of the lake's nitrogen budget. The conclusion that high rates of nitrogen fixation are at least partly responsible for the very high levels of nitrogen in Lake Hancock appears to be supported by recent water quality modeling work (FDEP, 2005) wherein "internal loading" of TN was required to account for the lake's measured phytoplankton biomass. It is therefore important to determine what factors are most likely responsible for the dominance of cyanobacteria in Lake Hancock's waters.

Early work by Smith (1983) suggested that low TN:TP ratios (<29), which could be caused by excessive phosphorus loading, favor dominance by cyanobacteria. However, a broader and Florida-based assessment by Canfield and co-workers (1989) found little evidence for a dominant role of TN:TP ratios in determining cyanobacteria abundance. Instead, Canfield and co-workers (1989) concluded that cyanobacteria become proportionally more important in Florida lakes as overall phytoplankton biomass increases. Other researchers (e.g., Downing et al., 2001) also concluded that overall nutrient supply, rather than ratios of nitrogen to phosphorus, was the best predictor of the dominance of phytoplankton communities in lakes by cyanobacteria.

Polk County and FDEP contracted with PBS&J to conduct a study to determine the potential benefits to water quality in Lake Hancock from sediment removal activities, within the context of the pending FDEP-derived TMDL (Total Maximum Daily Loads) determinations for Lake Hancock and its tributaries. The overall project design involved the implementation of two concurrent field studies. The first study (Task 1) was a manipulative experiment to determine if variation in the levels of nutrient availability could bring about decreases in cyanobacteria vigor and/or abundance. The second study (Task 2) was a manipulative mesocosm experiment designed to determine if the partial removal of the lake's sediments would likely result in significant improvements in water quality. Data derived from the two experimental field studies and a review of historical data sets were analyzed to address the questions listed below.

1. Is there a phosphorus and/or nitrogen level below which cyanobacteria in Lake Hancock no longer dominate the phytoplankton assemblage?
2. Can removal of the organic sediments in Lake Hancock reduce phosphorus and/or nitrogen concentrations below these threshold levels?
3. Will the removal of organic sediments from Lake Hancock likely result in a measurable improvement in water quality (e.g., total nitrogen (TN), total phosphorus (TP), Chl-a, BOD, TSI), and a reduction in the abundance of nuisance cyanobacteria?

METHODS—*Task 1 - Manipulative Phytoplankton Response Study*—On two occasions (December 2006 and July 2007), a series of experiments were carried out to measure rates of dinitrogen gas (N_2) and carbon dioxide (CO_2) fixation in Lake Hancock. In December 2006, three sites were selected within Lake Hancock (North-LH1, Central-LH2 and South-LH3) (FIG. 1). However, in July 2007 the lake's water level was exceptionally low and the sampling and incubation equipment could not be realistically taken out to the center of the lake. Instead, water samples were taken from a jetty on the west side of the lake where the incubation experiments were completed. This process used a high-sensitivity stable isotope tracer method (Montoya et al., 1996) to quantify rates in bottles suspended from the jetty. In brief, lake water was transferred to 250-ml polycarbonate bottles equipped with septum caps. All bottles were filled to overflowing and care was taken to eliminate all bubbles from each bottle as it was capped and sealed. Tracer additions of $^{15}N_2$ gas (Isotech, 250 µL at 1 atm pressure) and $NaH^{13}CO_3$ (New England Nuclear, 30 µL of a 0.1 M solution in DIW) were made using gas-

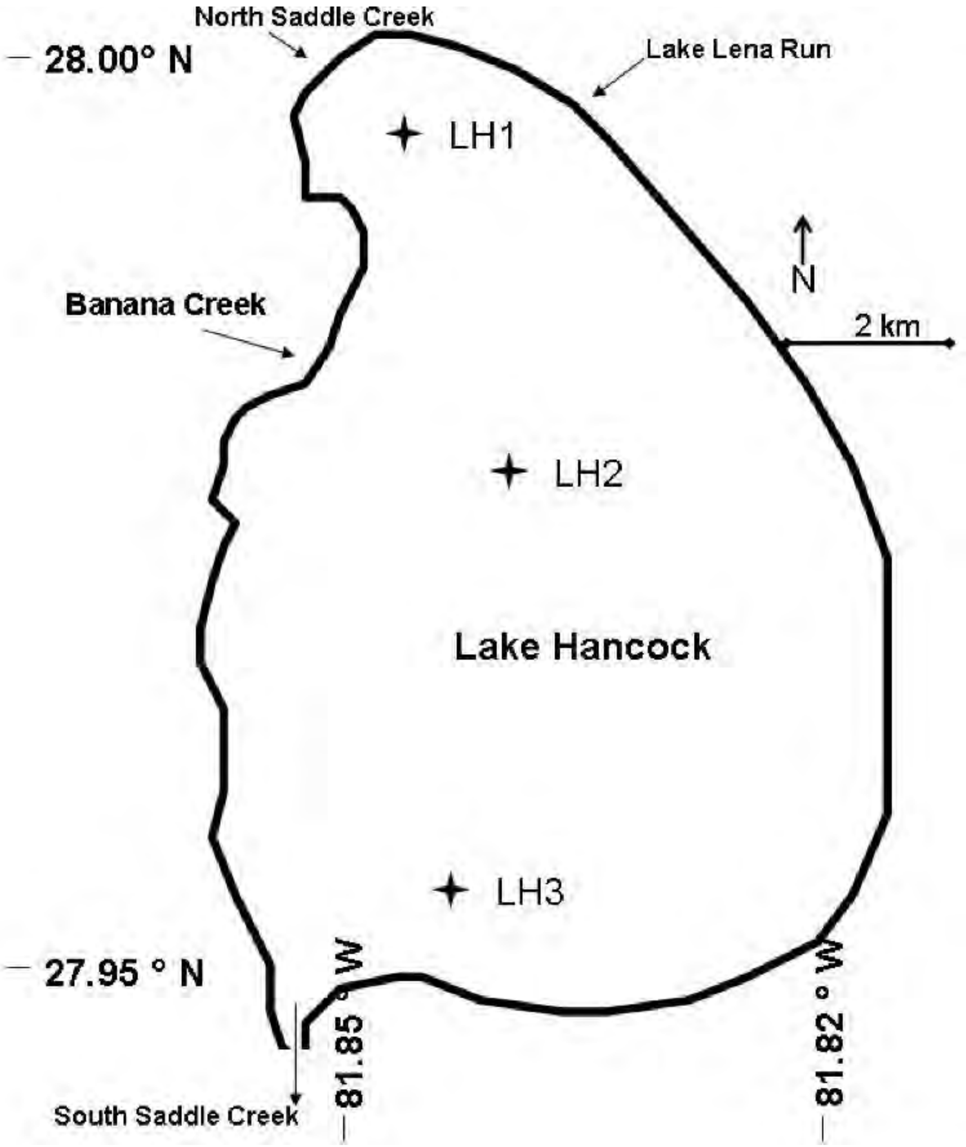


FIG. 1. Map of Lake Hancock showing approximate location of three sampling stations.

tight syringes. The syringe used for the $^{15}N_2$ injection was fitted with a valve, which wasn't necessary for the liquid injection. P amendments (1 mM solution of KH_2PO_4 in DIW) to incubation bottles were also carried out using gas-tight syringes.

All bottles were incubated in situ, with the bottles suspended so that they rested just below the surface of the lake. Experiments were terminated by filtration of water from the bottles through pre-combusted GF/F filters, which were kept on ice for transport to the lab, where they were dried and stored over desiccant. Sample filters were trimmed, and then sub-sampled for isotopic analysis, which entailed pelletization in a tin capsule (Costech) and flash combustion followed by continuous-flow isotope ratio mass spectrometry using a Carlo Erba NC2500 elemental analyzer interfaced to a Micromass Optima mass spectrometer. We used the method outlined by Montoya and co-workers (1996) to calculate rates of N_2- and CO_2-fixation.

The December 2006 experiment determined rates of N-fixation from the three locations throughout the lake (described in Task 2 below). The second experiment (July 2007) involved estimations of N-fixation rates within the lake, but also two additional experiments – an assessment of the role (if any) of light levels on rates of N-fixation, and an assessment of the role (if any) of levels of phosphorus availability on N-fixation rates.

The phosphorus effects study was carried out in July 2007 experiments to test the effect of P availability on metabolic rates on short (hour) and longer (day) time scales:

1. On 10 July, large (4.5L) bottles were filled with lake water and amended with 0, 5, and 10 µM PO_4^{3-}. These bottles were incubated at the surface of the lake through 12 July and subsamples were withdrawn for tracer measurements of N- and CO_2-fixation rates as described above on 11 and 12 July (i.e., after roughly 24 and 48 h of preincubation).
2. On 11 and 12 July, N- and CO_2-fixation rates were measured in lake water as described above with addition of 0, 5, and 10 µM PO_4^{3-} to the incubation bottles at the start of the experiment.

Task 2 - Manipulative Sediment-Water Quality Response Study—For the second study, three locations within Lake Hancock (North-LH1, Central-LH2 and South-LH3) were selected to allow for heterogeneity in water and sediment depth (FIG. 1). At each location, water and sediment depth were measured using a clear 5-cm diameter acrylic tube. Large (1.8 m diameter) aluminum mesocosms were assembled from two sections. The bottom portion ranged from 0.9 to 1.07 m tall, depending on sediment depth. This section of the mesocosm was designed to prevent exchange with the surrounding sediment matrix. The cylinder had an open top, but its bottom portion was solid aluminum. The cylinder was pushed through the organic sediment layer into the sandy bottom of the lake to ensure complete isolation. The top portion ranged from 1.5 to 1.7-m tall, depending on water depth. The top section was comprised of an open aluminum frame covered with a double layer of reinforced 10-mm clear polyvinyl material. The construction of the top portion of the cylinders thus resulted in the isolation of the water column within the cylinders while still allowing for natural wave action and light penetration. The maximum height of each mesocosm was 2.6 m. Additionally, a reference site (control) was established at each location and marked with a tethered buoy to evaluate the lake under "normal" conditions (without manipulation).

The experiment was conducted twice, December 2006 and May 2007. Two cylinders were deployed at each of the three replicate locations on November 30, 2006 and May 2, 2007. The cylinders were removed between the December and May sampling periods to reduce the influence of the cylinder affect on water quality. One cylinder at each location was randomly selected for dredging and subsequently dredged on December 1, 2006 or May 3, 2007, using a hydraulic pump. The other cylinder was not dredged with a hydraulic pump. Reference sites were established in close proximity to each of the three pairs of dredged and undredged mesocosms. Three replicate cores were taken within each cylinder (dredged and undredged) from each site, and the sediment and water depths were recorded.

TABLE 1. Sampling dates for the mesocosm experiments.

Condition	Sampling Dates	
Not-Mixed	December 6th, 2006	May 7th, 2007
Mixed	December 6th, 2006	May 9th, 2007
Not-Mixed	December 11th, 2006	May 14th, 2007
Mixed	December 13th, 2006	May 16th, 2007

For both the December and May experiments, water samples were collected and water quality parameters estimated at each of the nine sites on four sampling dates (Table 1). A suite of physical, chemical and biological parameters were assessed (Table 2). The water quality laboratory of the Polk County-Natural Resources Division analyzed all parameters except BOD, which was analyzed by PhosLab Co. in Lakeland, Florida. All samples were collected 20 cm below the water surface within each cylinder and at each reference site. Additionally, a profile of the water column was completed using a HydroLab®. On the first and third sampling dates, water samples were collected without manipulation of the water column. On the second and fourth sampling dates, a paddle was used to mix the isolated water column to simulate windy day conditions on the lake. Initially, the undredged cylinder at each location was "stirred" to determine the amount of effort necessary to

TABLE 2. Water quality parameters measured for each sample.

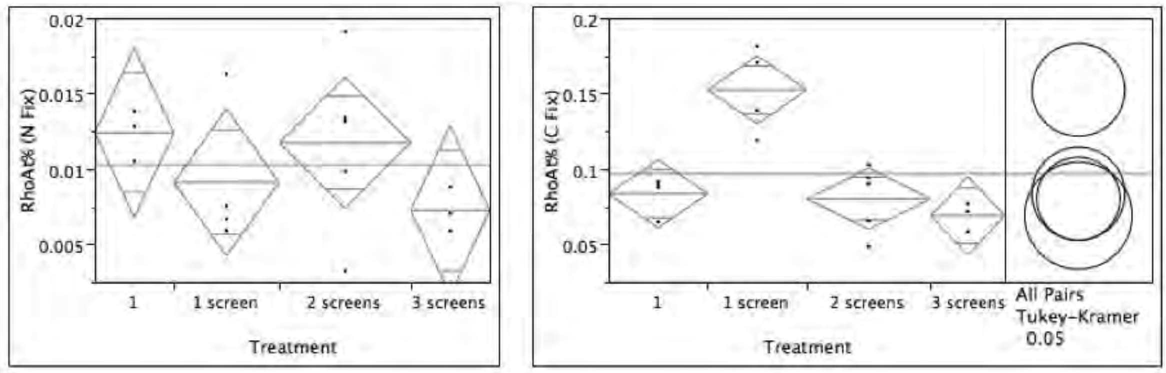
Parameter	Unit	Method
Physical		
pH		HydroLab
Temperature	°C	HydroLab
Dissolved Oxygen (DO)	mg/liter	HydroLab
Specific Conductivity	umhos	HydroLab
Secchi Depth	cm	Secchi Disk/ measuring tape
Turbidity (Field)	NTU	Hach 2100P Turbidimeter
Turbidity (Lab)	NTU	2130 B
Color	Pt-Co	2120 B
Total Suspended Solids (TSS)	mg/liter	2540 D
Chemical		
Total Nitrogen (TN)	mg/liter	351.2 & 4500-NO3F
Total Kjeldahl (TKN)	mg/liter	351.2
Total Phosphorus (TP)	mg/liter	EPA365.4
Nitrate+Nitrite (NOx)	mg/liter	4500-NO3F
Ortho-Phosphate (SRP)	mg/liter	4500 P-F
Biological		
Chlorophyll *a*-corrected (Chl a)	µg/liter	102000 H
Biological Oxygen Demand (BOD)	mg/liter	SM 5210B

reach high turbidity conditions. Initial and post-stirring turbidity readings were taken to establish the increase in turbidity due to mixing, using a Hach® 2100P Turbidimeter. The same level of effort (i.e., time spent stirring the water column) was then used at the dredged and reference sites at each location. Water quality samples and water quality profile readings were taken after the mixing was completed. Statistical analyses used either parametric or non-parametric techniques, as was appropriate based on determination of normality and/or homoscedasticity. Statistical significance was set, *a priori*, at $p < 0.05$.

RESULTS—*Task 1 - Manipulative Phytoplankton Response Study*—Baseline N_2-fixation rates found during this study of 16.2 ± 4.5 nmol N L^{-1} h^{-1} (mean ± SE, N=9) were comparable to rates measured in December 2006, but were lower than measured rates from March 2005 (133 ± 22 nmol N L^{-1} h^{-1}, N=7). The baseline CO_2-fixation rates were 130 ± 27 nmol C L^{-1} h^{-1} (mean ± SE, N=9). These rates are quite low relative to the very high biomass observed (particulate nitrogen [PN] levels of 705 ± 34 µmol N L^{-1} and particulate carbon [PC] of 5100 ± 260 µmol C L^{-1} (mean ± SE, N=9).

The ratio of specific CO_2- and N_2-fixation rates was relatively low (V_C:V_N = 12.4 ± 3.5), implying that N_2-fixers made up a modest fraction (ca. 8 percent) of the photosynthetically active biomass at this time of year. Both instantaneous rates were very low relative to the biomass of the lake, perhaps reflecting the high temperatures and low lake levels in July 2007.

An experiment was carried out during the July 2007 experiments to test the effect of light intensity on both N- and C-fixation rates (FIG. 2). The



Differences not significant

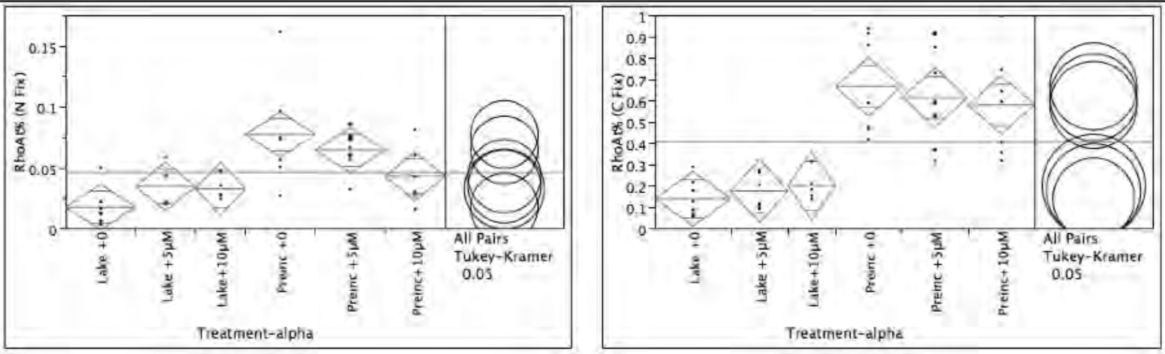
Level	Group	Mean
1 screens	A	0.152
0 screens	B	0.086
2 screens	B	0.080
3 screens	B	0.069

FIG. 2. Effect of light intensity on N- and CO_2-fixation rates in July 2007. Results of a Tukey's HSD multiple comparisons test are shown below the CO_2-fixation results. The differences among the N_2-fixation rates were not significant.

experiment used neutral density screening to vary the light intensity in bottles suspended just below the surface of the lake. These treatments (0, 1, 2, and 3 screens) correspond roughly to 100 percent, 50 percent, 25 percent, and 12.5 percent of ambient irradiance. Results (FIG. 2) indicate no significant effect of light intensity on N-fixation rates. In contrast, light intensity had a significant effect on CO_2-fixation rates (also FIG. 2), with higher rates in the bottles exposed to 50 percent of ambient irradiance. The other three light treatments did not differ significantly.

These results show clear evidence for photoinhibition (reduced photosynthetic rates with increased light levels) of primary production at surface irradiances (100% ambient irradiance). The CO_2-fixation rate in the ~50 percent irradiance treatment is roughly double the rate under full surface light. In contrast, results showed no evidence for light limitation or light inhibition of N-fixation.

Results from the P enrichment study showed a mixed response (FIG. 3). Addition of P had no significant immediate effect on N_2- and CO_2-fixation rates, though N_2-fixation rates were higher in bottles amended with P. Our preincubation treatment affected both N_2- and CO_2-fixation rates with no significant difference between days (i.e., with the length of the preincubation, 1 or 2 days). Our preincubated samples showed uniformly elevated rates of CO_2-fixation and elevated rates of N_2-fixation in our control and_5 μM PO_4^{3-} preincubations.



Level	Group			Mean
Preinc +0	A			0.077
Preinc +5µM	A	B		0.064
Preinc+10µM	A	B	C	0.043
Lake +5µM		B	C	0.034
Lake+10µM		B	C	0.032
Lake +0			C	0.017

Level	Group		Mean
Preinc +0	A		0.665
Preinc +5µM	A		0.614
Preinc+10µM	A		0.579
Lake+10µM		B	0.199
Lake +5µM		B	0.176
Lake +0		B	0.139

FIG. 3. Effect of P amendment on N-fixation (left) and CO_2-fixation (right) rates in July 2007. The left half of each panel shows the immediate effect of P addition (0, 5, 10 µM PO_4^{3-}) on rates, while the right half of each panel shows the results of preincubation following addition of 0, 5, or 10 µM PO_4^{3-} (see methods for details). Results of a Tukey's HSD multiple comparisons are shown below the plots.

Task 2 - Manipulative Sediment-Water Quality Response Study—Three cores were taken at each location to measure the water and sediment profile during both December and May sampling events. The average core profile for each site is displayed in Figures 4 and 5. During December, sediment depth in the dredged cylinders was significantly less than the reference and undredged cylinders (as was anticipated). There was a 35–52 cm change in sediment depth in dredged vs. undredged cylinders. Additionally, a 46 to 86 percent increase in water volume was found in the dredged cylinders compared to the undredged cylinders and reference sites. The total volume of water within the cylinders ranged from a minimum of 7,949 liters (undredged) to a maximum of 19,306 liters (dredged). In comparison, the volume of sediment ranged from a minimum of 1,136 liters (dredged) to a maximum of 7,571 liters (undredged). In May, sediment in the dredged cylinders was significantly less than the reference and undredged cylinders. There was a 31–50 cm change in sediment depth due to dredging compared to the undredged cylinders in May. Additionally, a 79 to 151 percent increase in water volume in the dredged cylinders compared to the undredged cylinders and reference stations was found. The total volume of water within the cylinders ranged from a minimum of 3,558 liters (undredged) to a maximum of 12,870 liters (dredged). In

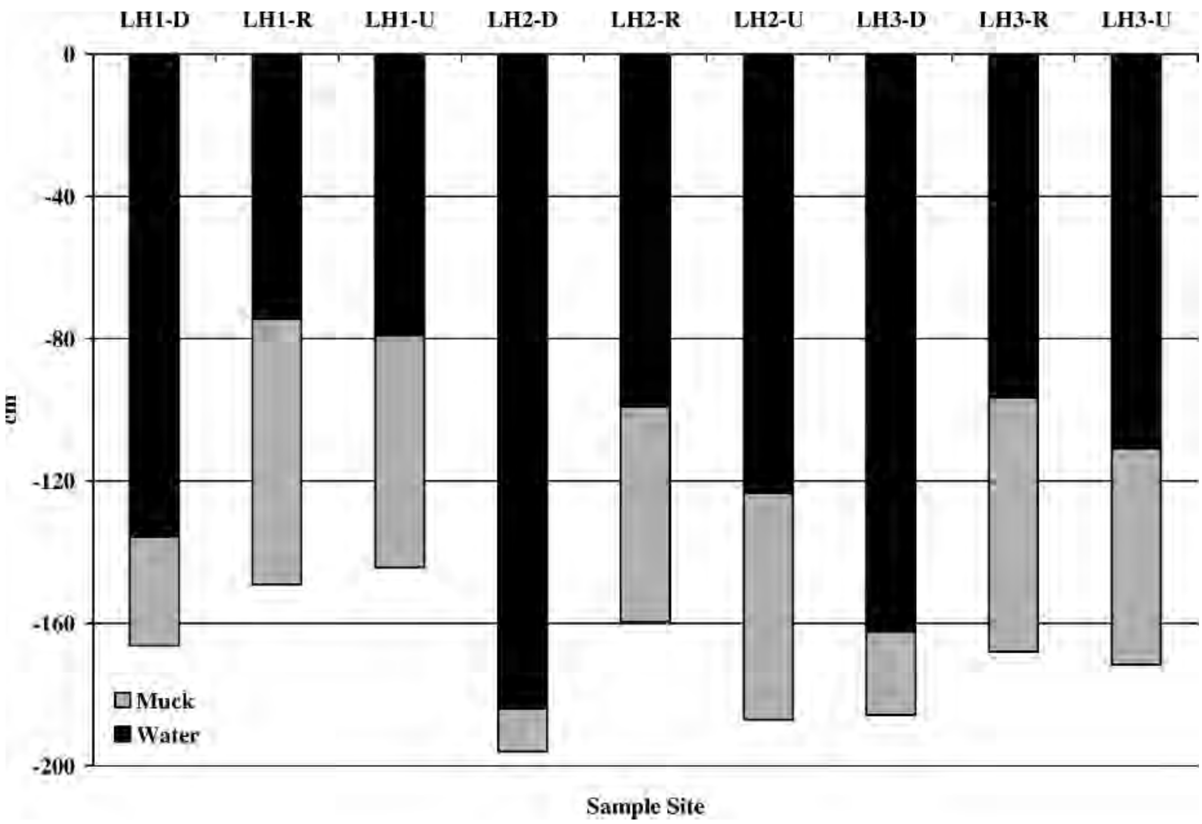


FIG. 4. December 2006 average muck and water depth at each site.

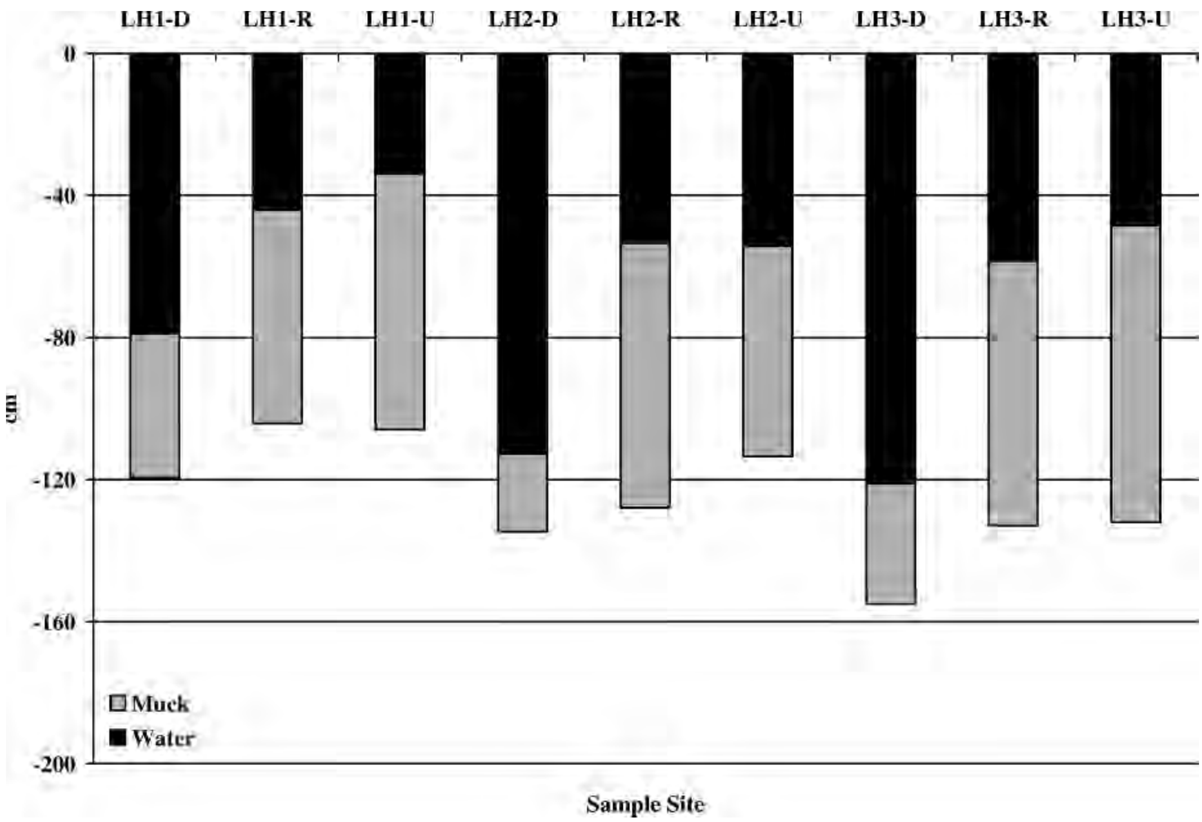


FIG. 5. May 2007 average muck and water depth at each site.

comparison, the volume of sediment ranged from a minimum of 2,309 liters (dredged) to a maximum 8,707 liters (undredged).

A total of 16 chemical and biological water quality parameters were then compared for dredged vs. undredged cylinders, under mixed vs. non-mixed conditions, for both November and May sampling events. To display these results in their entirety, a total of 32 figures would be required. The raw data alone would require a table with 192 row and column combinations (i.e., 16 parameters at each of 6 treatments [dredged, reference, undredged] under both mixed and unmixed conditions, on 2 occasions). Rather than present such a massive amount of data, we have presented illustrative figures for TN, TP, and Chl-a and also a single table that summarizes the percent change in parameters for each combination of treatment, condition and occasion. More detailed results are available in the report prepared by PBS&J (2008).

The average TN concentrations for each treatment, for both December and May, are shown in Figures 6 and 7, respectively. In December, TN concentrations in the unmixed cylinders were reduced by 20 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, TN concentrations in December were 53 percent lower in dredged vs. undredged cylinders. In May, TN concentrations in the unmixed cylinders were reduced

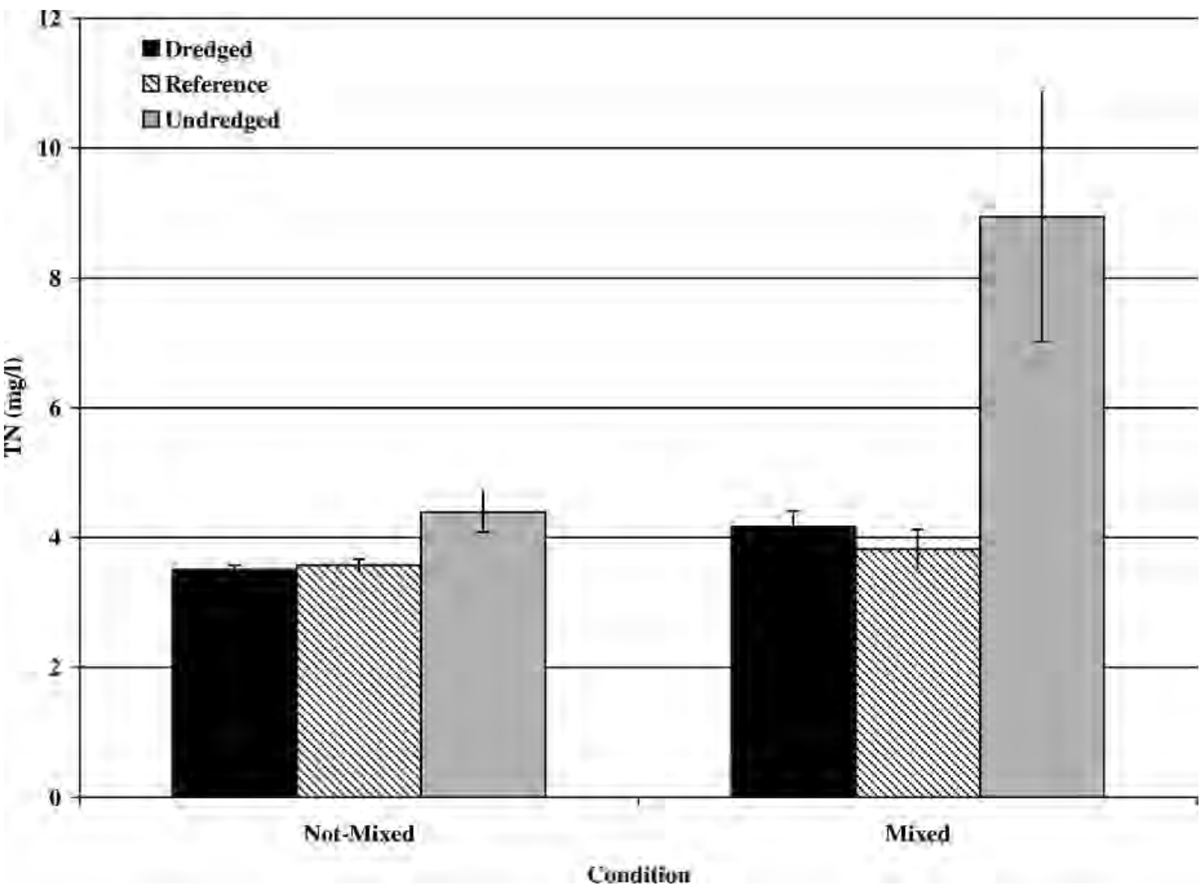


FIG. 6. December 2006 average total nitrogen results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

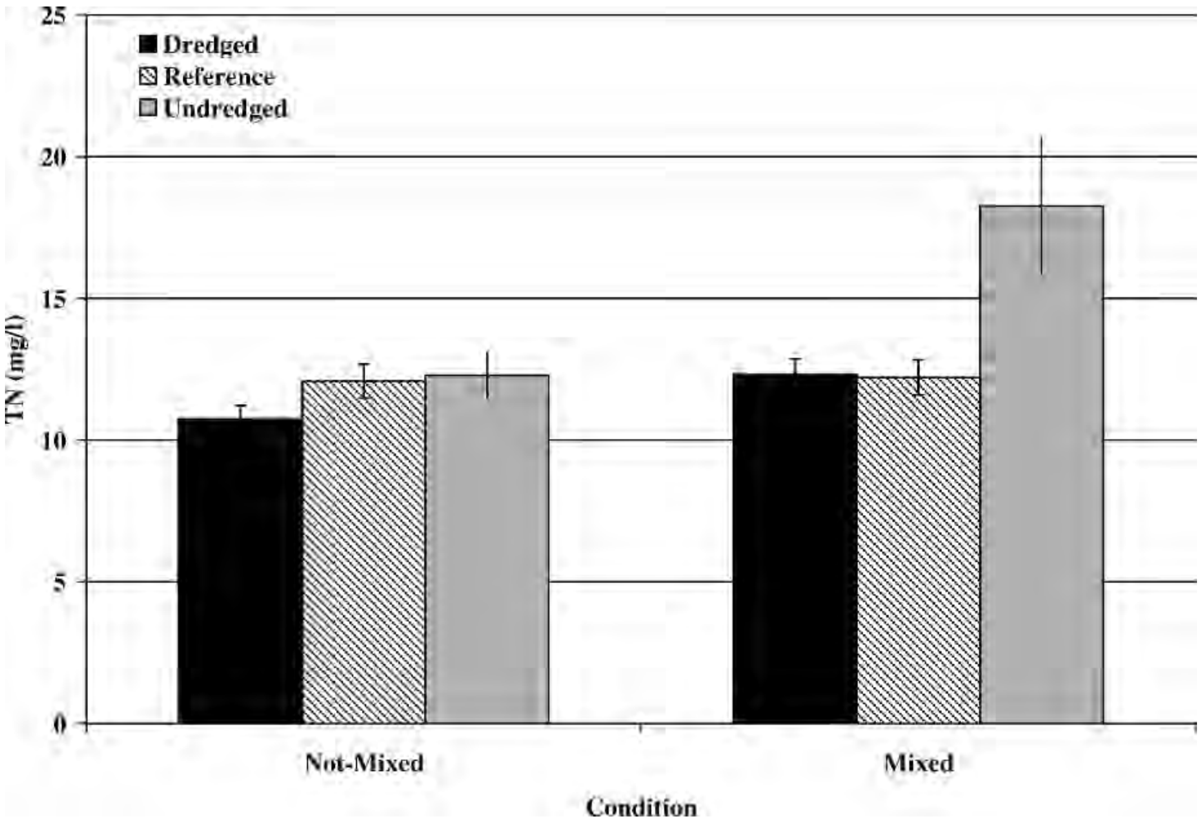


FIG. 7. May 2007 average total nitrogen results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

by 13 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, TN concentrations in May were 33 percent lower in dredged vs. undredged cylinders.

Mean TP concentrations for each treatment, for both December and May, are shown in Figures 8 and 9, respectively. In December, TP concentrations in the unmixed cylinders were reduced by 37 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, TP concentrations in December were 77 percent lower in dredged vs. undredged cylinders. In May, TP concentrations in the unmixed cylinders were reduced by 21 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, TP concentrations in May were 69 percent lower in dredged vs. undredged cylinders.

The mean Chl-a concentrations for each treatment, for both December and May, are shown in Figures 10 and 11, respectively. In December, Chl-a concentrations in the unmixed cylinders were reduced by 21 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, Chl-a concentrations in December were 31 percent lower in dredged vs. undredged cylinders. In May, Chl-a concentrations in the unmixed cylinders were reduced by 5 percent, comparing the dredged vs. undredged cylinders. For the mixed cylinders, Chl-a concentrations in May were 13 percent lower in dredged vs. undredged cylinders.

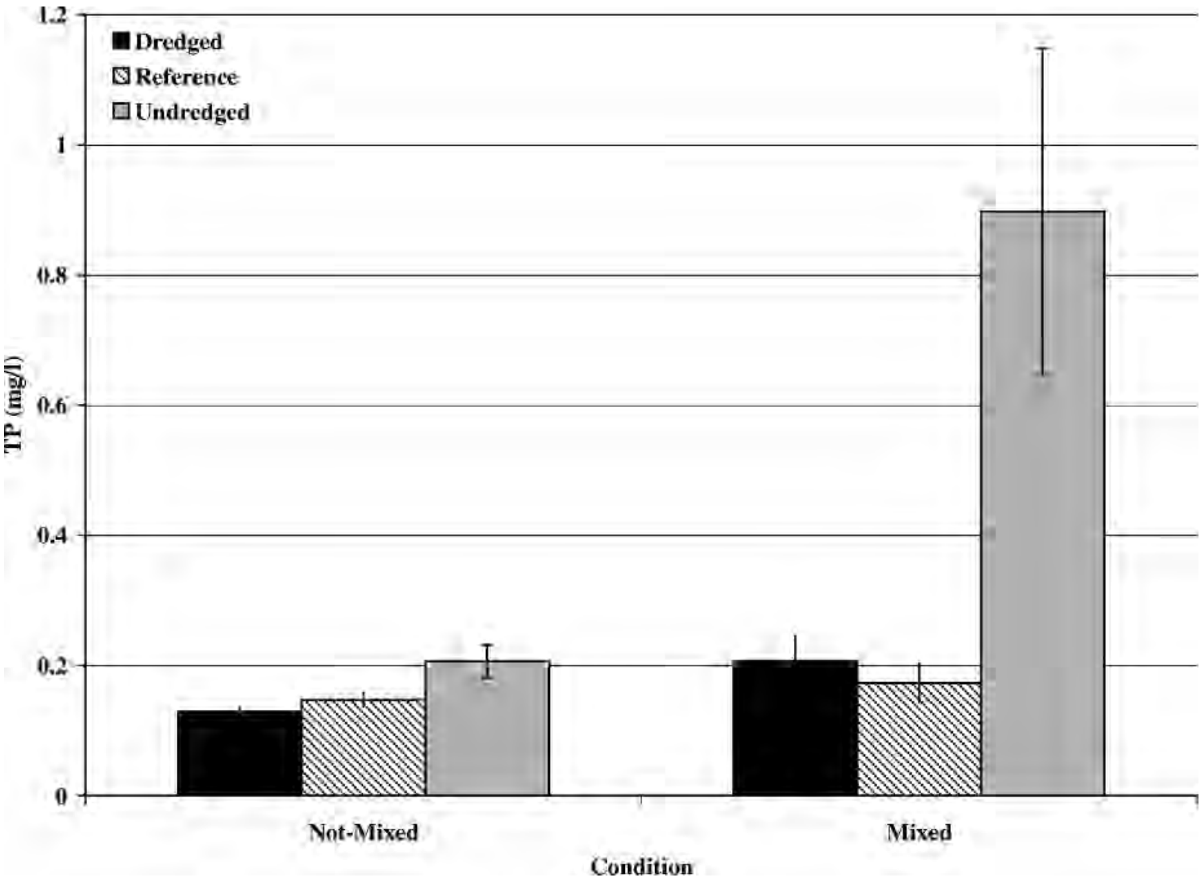


FIG. 8. December 2006 average total phosphorus results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

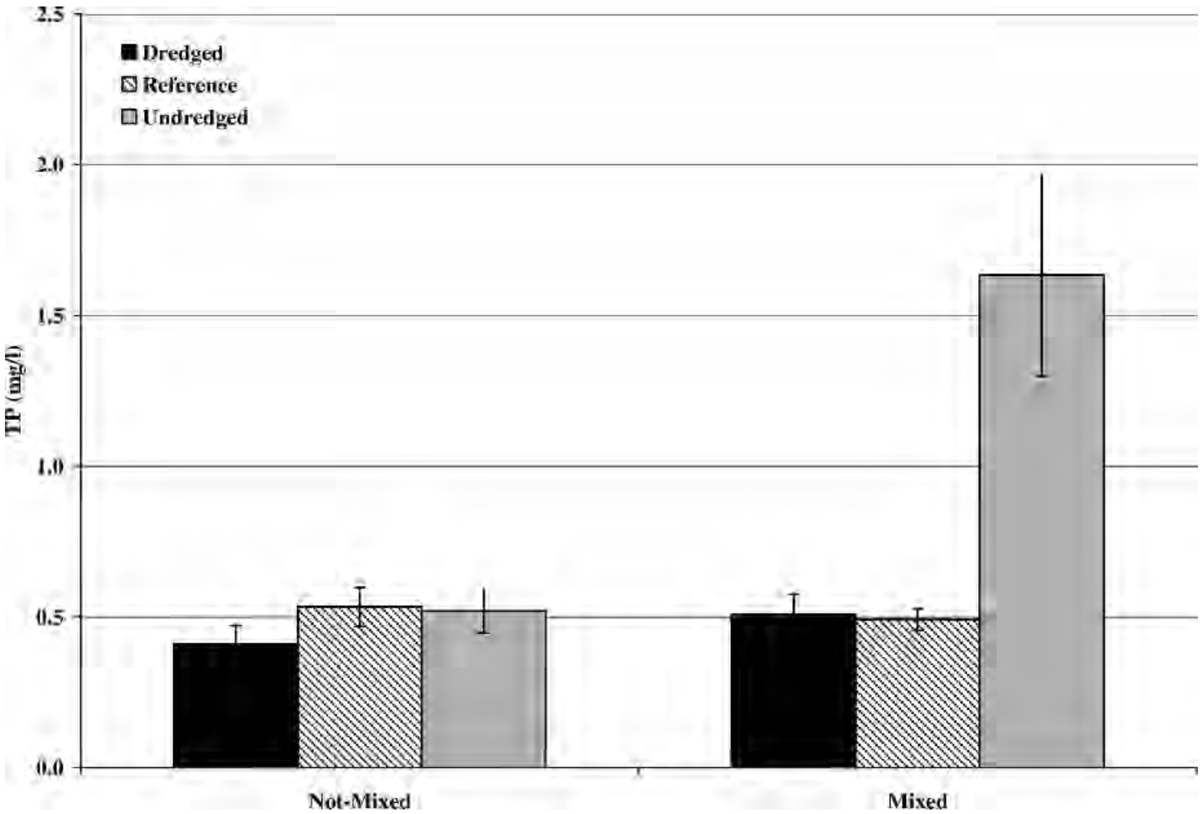


FIG. 9. May 2007 average total phosphorus results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

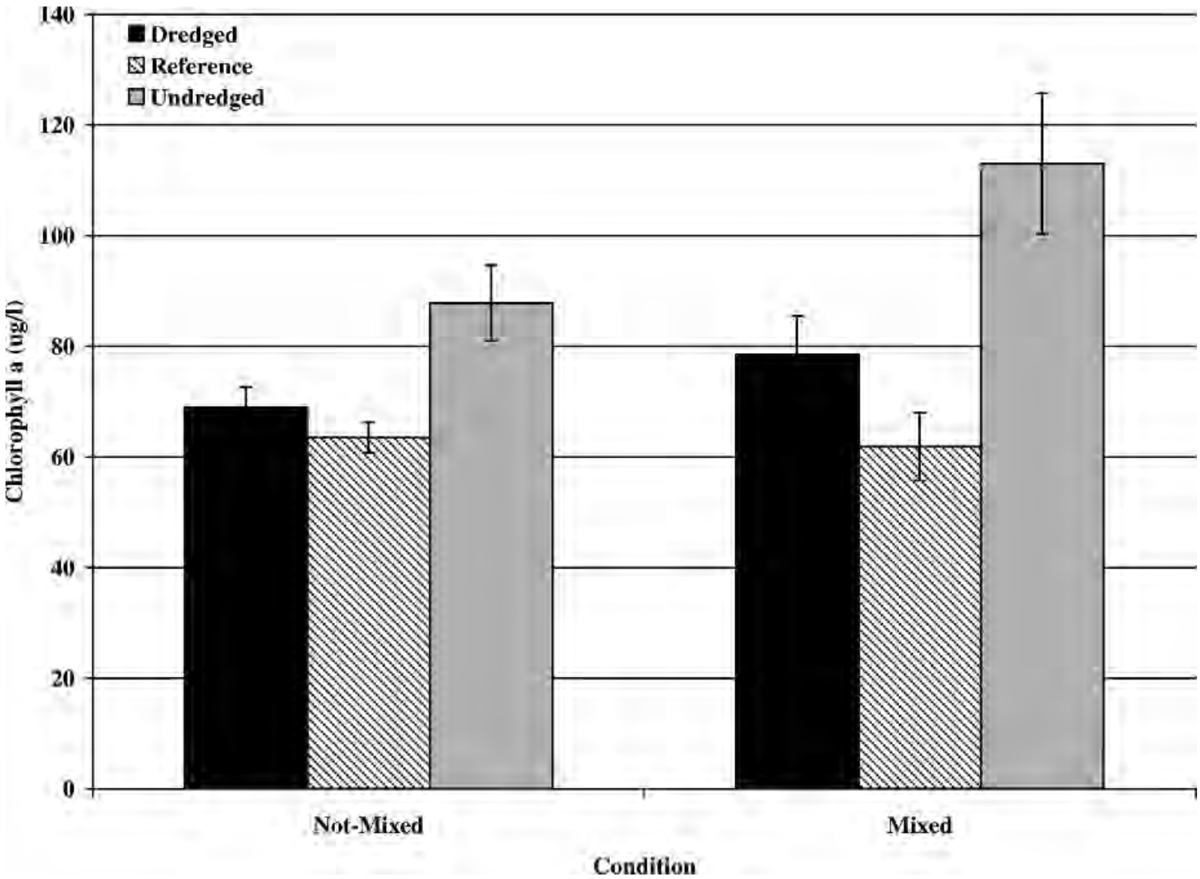


FIG. 10. December 2006 average Chl-a results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

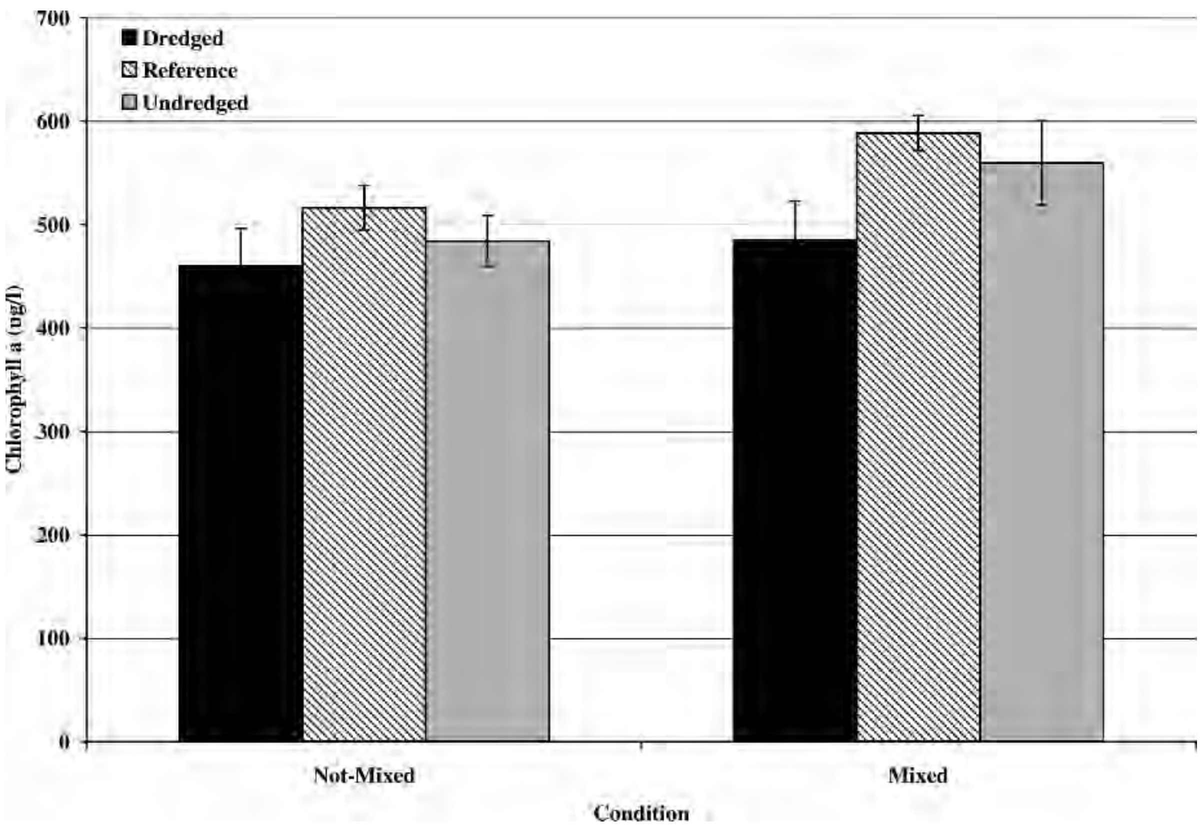


FIG. 11. May 2007 average Chl-a results at the dredged, reference and undredged sites under not-mixed and mixed conditions.

TABLE 3. Percent change of dredged versus undredged cylinders.

	December 2006		May 2007	
	Not-Mixed	Mixed	Not-Mixed	Mixed
Chl a (corrected)	−21	−31	−5	−13
TN	−20	−53	−13	−33
TP	−37	−77	−21	−69
TSS	−35	−72	−42	−78
Turbidity	−41	−75	−14	−60
TN:TP	26	101	11	111
BOD	−13	−13	3	−6
TSI	−4	−12	−1	−7

Table 3 summarizes the comparison of dredged versus undredged cylinders, for both mixed and non-mixed treatments, for both December and May. Levels of TN were reduced between 13 and 53 percent with sediment removal, while levels of TP were reduced between 21 and 77 percent. TSS was reduced between 35 and 78 percent, with turbidity down by 14 to 75 percent. Chl-a concentrations were reduced, comparing dredged versus undredged cylinders, by 5 to 31 percent. BOD was reduced by 13 percent for December samples, but showed either a slight increase (not-mixed) or a small decrease (6 percent) in May. Ratios of TN to TP increased between 26 and 111 percent, indicating conditions would tend toward those less favorable for cyanobacteria, which may benefit from low TN:TP ratios. However, TSI values exhibited a lesser degree of change, compared to both nutrient and Chl-a concentrations, with reductions of 1 to 12 percent found, when comparing dredged to undredged cylinders.

DISCUSSION—Data derived from the two experimental field studies and a review of historical data sets were analyzed to address the three questions previously presented in the introduction. The questions focused on the nutrient thresholds for cyanobacteria competitive advantage, the relationship of organic sediments on nutrient concentrations, and the effect of the removal of organic sediments on the overall water quality of the lake. Each question is addressed below:

Is there a "threshold" value for TN and/or TP for cyanobacteria?—Smith (1983) suggested that low TN:TP ratios (<29), brought about by excessive phosphorus loading, produce conditions that favor dominance by cyanobacteria. However, Canfield and co-workers (1989) found little evidence for an effect of TN:TP ratios on cyanobacteria abundance in Florida lakes. Instead, Canfield and co-workers (1989) concluded that cyanobacteria become proportionally more important in Florida lakes simply as overall phytoplankton biomass increases. This view has been supported by Downing and co-workers (2001) who concluded that overall nutrient supply, rather than ratios of nitrogen to phosphorus, was the better predictor of blue-green algal abundance.

In terms of TN:TP ratios, it is important to recognize that levels of nitrogen in Lake Hancock are not simply a function of external loading. High levels of nitrogen fixation can occur in Lake Hancock (FIG. 2 and 3) as has also been reported in another Florida lake with a similar degree of poor water quality (Lake Jesup in Seminole County) (PBS&J, 2006). Nitrogen fixation via photosynthetic cyanobacteria is likely to be a major, but not necessarily the only, source of nitrogen in Lake Hancock, as water column values of TN are much higher than levels of TN in stormwater runoff entering the lake. For example, levels of TN in Lake Hancock between 1992 and 2004 averaged 4.57 mg/liter (FDEP, 2005), while TN levels of runoff from North Saddle Creek, which supplies 87 percent of the tributary inflow volume into Lake Hancock, were measured at 1.24 mg/liter (ERD, 1999).

In contrast, the disparity between in-lake levels of TP and stormwater runoff is much less than the disparity between in-lake levels of TN and TN levels in the stormwater entering the lake. Between 1992 and 2004, TP values in the lake averaged 0.52 mg/liter (FDEP, 2005), while TP levels of runoff from North Saddle Creek were measured at 0.44 mg/liter (Table 5-4 in ERD, 1999). Perhaps associated with the removal of point source discharges from the cities of Auburndale and Lakeland (ERD, 1999), levels of TP in Lake Hancock declined from an average of 0.96 mg/liter in the 1980s to an average of 0.48 mg/liter in the 1990s (ERD, 1999), a 51 percent decline. In contrast, levels of TN decreased by only 10 percent during that same time period. Chl-a concentrations appear to have declined by only 5 percent, when comparing values in the 1980s to values in the 1990s (ERD, 1999).

Thus, recent reductions in nutrient loads comparing data from the 1980s to data from the 1990s, has been mostly associated with reductions in levels of TP, rather than TN. This has resulted in an increase in TN:TP ratios from a decadal average of 6.7 in the 1980s to 12.2 in the 1990s (ERD, 1999), a shift that if it continues, would increase TN:TP ratios to levels that could be less favorable for cyanobacteria, if the arguments of Smith (1983) have merit.

For reasons presented above, levels of TN within Lake Hancock are higher than what can be explained by loading models such as those constructed by ERD (1999) and FDEP (2005). Consequently, stormwater retrofit activities designed to reduce nitrogen loads to the lake might have little impact, as much of the nitrogen in the lake appears to be generated internally within the lake via nitrogen fixation or from resuspension of the lake's organic-rich sediments. The draft TMDL for Lake Hancock (FDEP, 2005), which calls for a 75.2 percent reduction in stormwater loads of TN, cannot be supported, as it does not properly account for the role of internal (from within the lake itself) sources of nutrients.

While our findings do not contradict the value of reducing external phosphorus loads to Lake Hancock, they do suggest that internal nutrient loads from remineralization and resuspension of bottom sediments are a significant and continuing source of TN and TP. Further, these results suggest that sediment removal activities could reduce in-lake TP values to levels close to, but not at, potential "thresholds" that could restrict the abundance of cyanobacteria.

In their nutrient budget for Lake Hancock, ERD (1999) measured TN and TP loads into the lake from stormwater runoff, direct rainfall, base flow, and in-lake seepage. Loads were also measured as they left the lake via South Saddle Creek. When loads into the lake from these four loading sources were totaled, 175,879 kg TN was calculated to enter the lake annually. Measured loads leaving the lake via South Saddle Creek totaled 271,700 kg/yr for TN. Measured TP loads into the lake were estimated at 35,086 kg TP per year, with TP loads leaving the lake via South Saddle Creek estimated at 23,180 kg/yr.

Based on these calculations, approximately 34 percent of the TP loads into the lake remain in the lake. In contrast, there is a surplus of 95,821 kg TN leaving the lake on an annual basis, compared to the amount calculated to enter the lake. This represents a "surplus" of nitrogen equal to 35 percent of the estimated load leaving the lake.

These findings are consistent with the hypothesis that Lake Hancock is a net sink for phosphorus loads, which accumulate in the sediments, while simultaneously being a net source for nitrogen (created via nitrogen fixation and organic sediments), which is then exported out of the lake, and eventually into the Peace River and Charlotte Harbor via South Saddle Creek.

Can removal of the organic sediments in Lake Hancock reduce phosphorus and/or nitrogen concentrations below threshold levels?—Activities that would reduce phosphorus loads, both internal and external, to the lake might in turn increase TN:TP ratios to levels that would no longer favor the dominance of cyanobacteria, if Smith (1983) is correct.

Our P amendment rate experiments revealed a complex response pattern to changes in P availability (FIG. 3). Our general expectation was that P amendments would increase the rates of N_2- and CO_2-fixation. In fact, addition of P had only a modest immediate impact on the rate of N_2-fixation and no effect on the rate of CO_2-fixation immediately after the nutrient amendment (FIG. 3, left half of each panel). In contrast, our pre-incubated samples showed higher rates of both N_2- and CO_2-fixation in all treatments, including the unamended controls (FIG. 3, right half of each panel). This pattern suggests that the pre-incubation may have led to a change in community composition and/or activity over the 1 to 2 days of bottle containment.

Our data do suggest that the P effect on the rate of N-fixation appears to be "saturated" at TP levels somewhat less than 0.16 mg TP/liter (0.5 µM, see FIG. 3). The actual saturation threshold could not be resolved with our data, but rates of N-fixation were not further increased by TP levels higher than 0.16 mg/liter. It would seem reasonable to conclude that an appropriate target for TP levels in Lake Hancock would somewhat less than 0.16 mg/liter. In contrast, TP levels within the lake averaged 0.52 mg/liter between 1992 and 2004 (FDEP, 2005).

Phosphorus levels within dredged cylinders in December 2006 averaged 0.13 and 0.21 mg TP/liter for non-mixed and mixed treatments, respectively. These results represent TP reductions of 37 and 77 percent, compared to non-dredged cylinders. In May 2007, phosphorus levels within dredged cylinders averaged 0.41 and 0.51 mg TP/liter for non-mixed and mixed treatments, respectively. Although these levels were higher than those found in December 2006, they still represented reductions of between 21 and 69 percent, compared to undredged cylinders (see Table 3).

The results from the mesocosm experiments suggest that sediment removal could result in substantial reductions in TP levels within Lake Hancock. If these results can be scaled up to whole-lake expectations, TP levels could be reduced to potentially biologically relevant concentrations (levels that may affect cyanobacteria abundance), although they would be higher than the potential TP target value of 0.16 mg/liter derived from the manipulative phytoplankton study. The results of these studies cannot answer the question as to whether or not sediment removal activities would drop TP levels low enough to result in a lake no longer dominated by cyanobacteria. It would seem that only by reducing TP loads from internal loading sources could TP levels be further reduced, and load reduction strategies that do not appropriately incorporate sediment nutrient availability might not be able to create desired water quality.

Will the removal of organic sediments from Lake Hancock likely result in a measurable improvement in water quality for other water quality parameters?— As a final test of the potential benefits to water quality associated with sediment removal, the Dynamic Ratio concept outlined by Bachmann and co-workers (2000) was examined. In a comprehensive survey of 36 Florida Lakes, Bachmann and co-workers (2000) found that water quality could be predicted, with fairly good results, based on the "Dynamic Ratio". The Dynamic Ratio is calculated as the square root of the size of the open water of the lake (in square kilometers) divided by the average depth of the lake (in meters). Using information within ERD (1999) the Lake Hancock Dynamic Ratio is thus:

$$\text{Dynamic Ratio} = \text{Square Root of } 18.4\ \text{km}^2/1.5\ \text{m} = 2.86 \qquad (1)$$

The Dynamic Ratio was then calculated for Lake Hancock under a potential scenario wherein 1 m of muck would be removed and the lake's seasonal high water level would be raised by approximately 0.3 m (equating to an average water depth of 2.8 m). The estimate of a 1 m removal of muck is simply an estimate of the amount of organic matter in the lake bottom, based on results summarized in Figures 2 and 3. The potential lake level elevation by 0.3 m is based on a project being pursued by the Southwest

Florida Water Management District to increase wet weather storage volumes within Lake Hancock. The Dynamic Ratio for Lake Hancock under this potential scenario is thus:

$$\text{Dynamic Ratio} = \text{Square Root of } 18.4\ \text{km}^2/2.8\ \text{m} = 1.53$$

Within Bachmann and co-workers (2000), equations are derived that allow for the estimation of levels of TN, TP, and Chl-a as related to Dynamic Ratios. However, Lake Hancock's water quality is sufficiently poor that it is an "outlier" to their data set, with higher levels of TN and TP, and much higher levels of Chl-a, than most of the lakes they examined. Other equations in Bachmann and co-workers (2000) can be used to calculate estimates of the amount of time that the entirety of the lake bottom would be expected to be disturbed by wind-driven waves. This equation is:

$$y = -2.45 + 6.47x; \qquad (2)$$

Where,

y = percent of time that 100 percent of the lake bottom would be disturbed, and
x = Dynamic Ratio

Using this equation, the percent of time that 100 percent of the bottom of Lake Hancock would be disturbed under present conditions calculates out to 16.1. Under the proposed sediment removal and water level elevation scenarios described above, with an increase in the effective water depth to 2.8 meters, the percent of time that 100 percent of the bottom of Lake Hancock would be disturbed calculates out to 7.5. Taking the inverse of these two numbers, present day conditions in Lake Hancock are such that 100 percent of the bottom of the lake would be expected to be disturbed every 6.2 days (100 / 16.1 = 6.2). Under the proposed sediment removal and water level elevation scenario this time interval would increase to once every 13.4 days (100 / 7.5 = 13.4).

If the Dynamic Ratio developed by Bachmann and co-workers (2000) accurately reflects the probability of sediment resuspension in Lake Hancock, the removal of 1 m of organic muck, combined with an increased water level of 0.3 m, could reduce the frequency of whole-lake sediment resuspension events from approximately once a week to just under once every two weeks. As nitrogen fixation and carbon fixation rates respond quite rapidly to pulses of phosphorus enrichment (see FIG. 3) and as resuspension of sediments brings about an increased level of TP within the water column, it would seem reasonable to conclude that increases in the effective water depth of Lake Hancock would likely bring about an improvement in water quality, due in part to conditions being less favorable for resuspension of bottom sediments.

Reductions in the internal loads of TP and increased water depths would be expected to decrease levels of Chl-a in Lake Hancock. These actions would also be expected to decrease levels of total suspended solids and turbidity within the lake, perhaps allowing for increased water clarity. Increased water clarity could potentially allow submerged vegetation to become established, at least in the shallowest fringes of the lake. Additionally, reductions in BOD might be expected to occur with sediment removal, especially on windy days. A reduction in BOD levels would not only benefit the lake itself, but it would likely benefit the downstream water bodies of South Saddle Creek to portions of the Upper Peace River.

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Managing Water Quality in Huntsman Lake (Virginia, USA)—Development and Implementation of Restoration Strategies

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Managing Water Quality in Huntsman Lake (Virginia, USA)—Development and Implementation of Restoration Strategies

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Abstract: Huntsman Lake is located within the Middle Run of the Pohick Creek watershed, which is itself located within the much larger Chesapeake Bay watershed. Data collected from both the water column and the lake's sediments indicate that phosphorus-rich bottom sediments are an important internal loading source, and these internal phosphorus loads would continue to adversely impact water quality until and unless sediments are removed or inactivated. The implementation of artificial circulation was anticipated to be able to increase the sequestration of phosphorus within better oxygenated bottom sediments, and was the first lake management strategy deployed in Huntsman Lake. In the first two years after the installation of a whole-lake circulation system, the lake's waters are no longer stratified, and the bottom waters are no longer hypoxic and/or anoxic. While there is no evidence of a subsequent reduction in concentrations of nitrogen or chlorophyll-a, average phosphorus concentrations have decreased. However, high variability in the phosphorus data decreases our confidence that this is a sustained improvement. These results are consistent with prior findings, including those from downstream systems, that the reversal of the symptoms of eutrophication can involve lag-periods up to several years, if they are successful at all.

Key words: Lake water quality, eutrophication, phosphorus, restoration strategies.

1. Introduction

Pohick Creek is a 95 km^2 watershed located within the much larger (38,000 km^2) Potomac River watershed. The Potomac River then discharges into the Chesapeake Bay, the largest estuary in the US. Beginning in the early 1960, the Pohick Creek watershed started to transform from a mostly rural and agricultural landscape to one where residential, commercial and transportation features became dominant. The human population of the Pohick watershed increased 20-fold from 1965 to 2005, from just under 5,000 to in excess of 100,000 people [1]. The increased impervious nature of the watershed that accompanied this development resulted in greater amounts of stormwater runoff in local waterways. This in turn increased the erosion of stream channels, and increased the amount of siltation in these same waterways. Increased siltation of waterways not only was a concern for its environmental impacts, but also because of concerns that sediment build up could adversely affect the downstream conveyance of flood waters.

The Watershed Protection and Flood Prevention Act of 1954 (also known as the Small Watershed Program) provided local governments with assistance for flood protection during major storm events within identified sub-basins of the major river systems of Virginia. In 1965, the Pohick Creek Watershed Protection and Flood Prevention Project (the Project)

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was developed under the authority of the Small Watershed Program. Following the recommendations outlined in the Project, six flood control dams were built in the Pohick Creek watershed. Huntsman Lake is impounded by the Pohick Creek No. 8 Dam, constructed in 1973. The open waters of Huntsman Lake average approximately 12 ha in size.

Since the time of construction of the dam, the technology for evaluating dam safety has improved, and based on these methods, the auxiliary spillway at Huntsman Lake needs to be improved. Based on current flood risk analyses, the auxiliary spillway for Huntsman Lake is expected to erode during a flood-flow event, potentially causing a breach and raising the possibility of an uncontrolled and rapid release of impounded waters. In response to such concerns, improvements to the flood protection and sediment retention capacities of Huntsman Lake have been scheduled. This paper summarizes the potential water quality benefits to the lake itself and to downstream waters that would be expected to occur with implementation of various lake management activities.

2. Materials and Methods

In August 2011, nine locations were sampled throughout Huntsman Lake (38.754° north latitude, 77.255° west longitude). Based on an evaluation of water quality data for two lakes listed on Virginia's 303 (d) list of impaired water bodies (little creek reservoir and lake Smith) the month of August was chosen as the time of the year with the greatest probability of exhibiting poor water quality. Samples were collected from Huntsman Lake in August to capture its likely worst water quality conditions.

Seven of the nine sites visited were close to the shoreline, in waters less than 1.2 m in depth. The other two sites were located west of the dam itself, in waters greater than 2.5 m in depth. Water quality data (field and laboratory parameters) were collected at nine surface locations throughout the lake, while sediment data were collected from eight locations throughout the lake. At each site, grab samples were collected at the water surface for the following laboratory parameters: TN (total nitrogen), TKN (total Kjeldahl nitrogen), NO_x (nitrate + nitrite), TP (total phosphorus), inorganic phosphorus, TSS (total suspended solids), mineral suspended solids (with VSS (volatile suspended solids) calculated as the difference between total and mineral suspended solids), and chlorophyll-a (Chl-a). A water quality probe was used to collect data for the field parameters of water temperature, specific conductance, and DO (dissolved oxygen). The probe was calibrated with standard solutions prior to collecting data, then was post-calibrated at the end of the day following standard QA/QC (quality assurance/quality control) procedures and guidance. Sediment samples were collected using a Ponar sampler, and composite samples from two replicates were analyzed for TKN, TP, organic content, and grain size analyses.

Data were analyzed and stations were compared against each other by calculating means and standard deviations; due to the lack of replication, no inferential statistical analyses were conducted. When examining the potential for relationships between independent variables such as water depth and potentially significant dependent variables, regression techniques were based on standard equations such as linear, power and logarithmic regressions. When shown, lines of best fit represent statistical significance at levels of $p < 0.05$ or better [2].

Artificial circulation was employed in Huntsman Lake beginning in July 2012, in response to conclusions reached based on these efforts. A 38,000 L per minute solar-powered water circulator (SolarBee© model SB 10000) was installed and programmed to run for the duration of the growing season. Following the installation of the circulator, water quality in Huntsman Lake was monitored (at a reduced number of stations) during the summer months in 2012 and 2013 to document any effects of artificial circulation.

3. Results

3.1 Initial Water Quality

Water depths at the nine sites visited varied between 0.49 m and 3.6 m. Locations along the periphery of the lake had water depths less than 1.2 m. Sites in the open waters of the lake west of the dam had the greatest depths.

Huntsman Lake exhibited some evidence of thermal stratification (Fig. 1) with an apparent inflection point located at a depth of approximately 1.5 m.

There also appears to be evidence of stratification in terms of specific conductance (Fig. 2), with a similar inflection point at a depth of approximately 1.5 m.

There is a strong pattern of decreasing DO values with increasing depth in Huntsman Lake (Fig. 3). This pattern is associated, at least in part, with physical stratification (warmer and less dense surface waters perched on top of colder bottom waters). However, there is a strong downward trend in DO at depths above the apparent stratification layer at 1.5 m, suggesting that factors other than stratification alone are impacting DO values.

Findings shown in Fig. 3 suggest that at least in August of 2011, the State of Virginia's Water Quality standard for dissolved oxygen (4 mg/L) is not met at

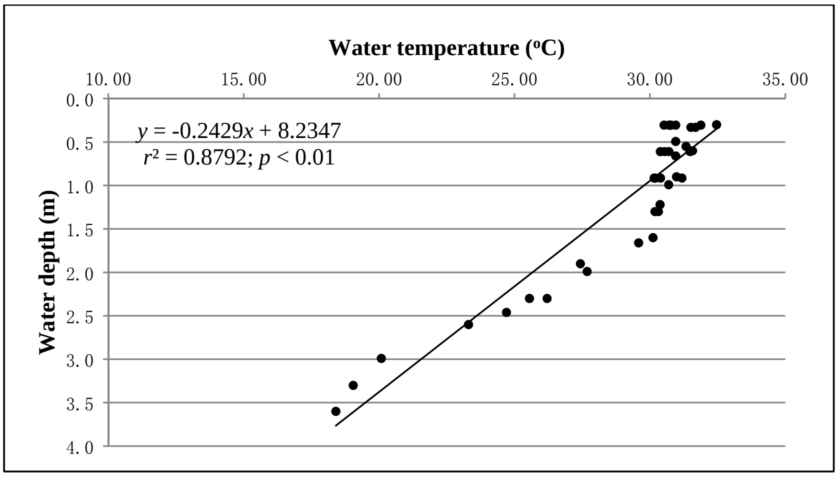

Fig. 1 Relationship between water temperature and water depth in Huntsman Lake.

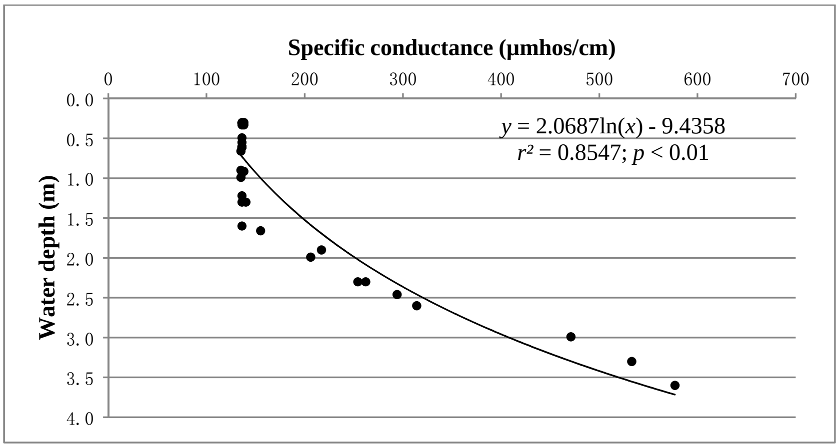

Fig. 2 Relationship between specific conductance and water depth in Huntsman Lake.

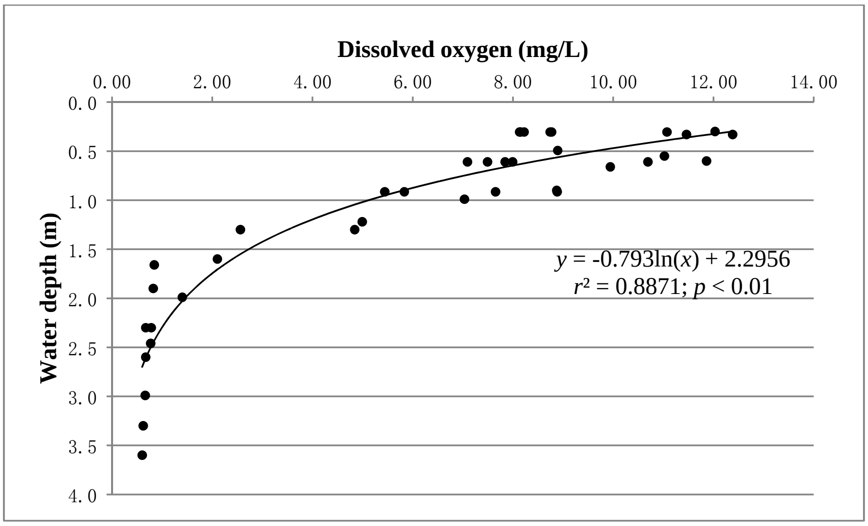

Fig. 3 Relationship between dissolved oxygen and water depth in Huntsman Lake.

Table 1 Summary of water quality data for Huntsman Lake.

Site	TKN (mg/L)	NO_x (mg/L)	TN (mg/L)	TP (mg/L)	Chl-a (μg/L)	TSS (mg/L)	VSS (mg/L)	MSS (mg/L)	Turbidity (NTU)	TN:TP	VSS %
1	1.71	0.01	1.72	0.150	147.00	29.00	15.00	14.00	29.20	11.45	51.72
2	1.48	0.00	1.48	0.156	109.00	26.20	15.70	10.50	20.70	9.49	59.92
3	1.01	0.00	1.01	0.106	46.60	22.10	11.80	10.30	23.30	9.53	53.39
4	1.00	0.00	1.00	0.110	81.40	23.10	13.00	10.10	23.30	9.07	56.28
5	0.85	0.00	0.85	0.102	66.50	20.40	11.60	8.80	15.70	8.28	56.86
6	1.10	0.00	1.10	0.078	64.20	12.60	10.60	2.00	13.40	14.10	84.13
7	0.83	0.00	0.83	0.096	65.30	18.20	12.40	5.80	18.20	8.65	68.13
8	0.83	0.00	0.83	0.096	61.50	17.20	11.40	5.80	18.10	8.69	66.28
9	1.08	0.00	1.08	0.084	38.30	9.10	7.10	2.00	17.00	12.86	78.02
Mean	1.10	0.00	1.10	0.109	75.53	19.77	12.07	7.70	19.88	10.24	63.86
Std. dev.	0.30	0.00	0.31	0.027	33.54	6.30	2.51	4.08	4.80	2.08	11.26

depths greater than about 1.2 m, and the deepest portions of Huntsman Lake (i.e., greater than 2 m) are hypoxic or anoxic.

A summary of laboratory parameters is shown in Table 1. Values of TN, TP and chlorophyll-a were highest at sites located near inflow in the northwest corner of the lake. This location could be a "hot spot" for watershed loading. The low NO_x and low to undetectable inorganic phosphorus levels could be indicative of low loads of these inorganic nutrient forms; however, the very high levels of chlorophyll-a suggest that inorganic nutrients are low because they are rapidly taken up by the elevated levels of algal biomass. Based on TN:TP ratios, it could be concluded that algal biomass in Huntsman Lake is co-limited by both nitrogen and phosphorus. In the water column, the majority (61%) of TSS appears to be VSS (12.07/19.77 = 0.61) suggesting that most of the suspended material within the water column is organic (i.e., algal biomass) and not suspended sediments such as sand and/or silt and clay.

A comparison of water quality from Huntsman Lake to proposed guidance criteria developed for the VDEQ (Virginia Department of Environmental Quality) [3] and also developed by the United States EPA (Environmental Protection Agency) [3] shows that Huntsman Lake has much higher levels of chlorophyll -a, TN and TP than what is considered appropriate for lakes in this region of Virginia (Table 2). Average levels of TN, TP and chlorophyll-a exceed guidance criteria for Virginia lakes by factors of approximately two, five and 15, respectively (Table 2).

The finding, based on TN:TP ratios, that Huntsman Lake is likely co-limited by both nitrogen and phosphorus is also supported by the statistically significant relationships that were found plotting chlorophyll-a against both TN and TP (Figs. 4 and 5, respectively).

Table 2 Comparisons of Huntsman Lake mean concentrations with proposed guidance for Virginia lakes and reservoirs in Eco Region Nine [3].

Source	Chl-a (μg/L)	Secchi depth (m)	TN (mg/L)	TP (mg/L)
Proposed VDEQ guidance	3.99	1.45	0.44	0.020
EPA guidance	4.93	1.53	0.36	0.020
Huntsman Lake (mean values)	75.53	0.36	1.10	0.109

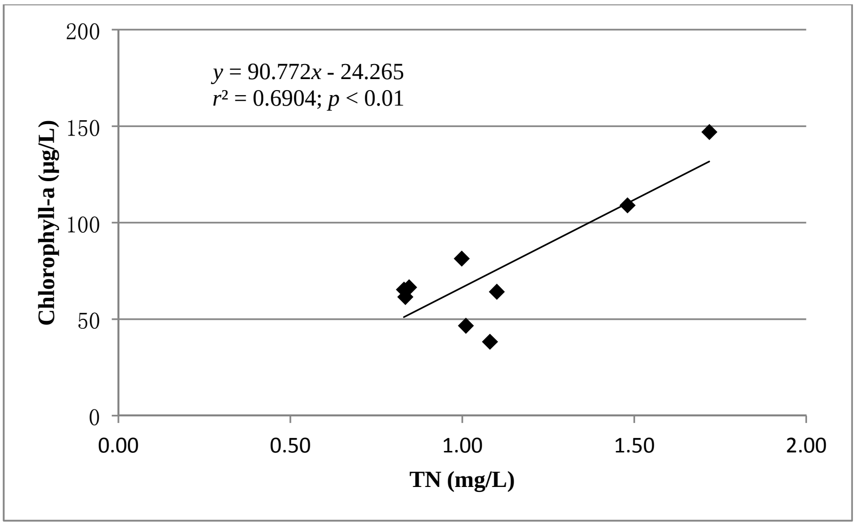

Fig. 4 Relationship between chlorophyll-a and TN in Huntsman Lake.

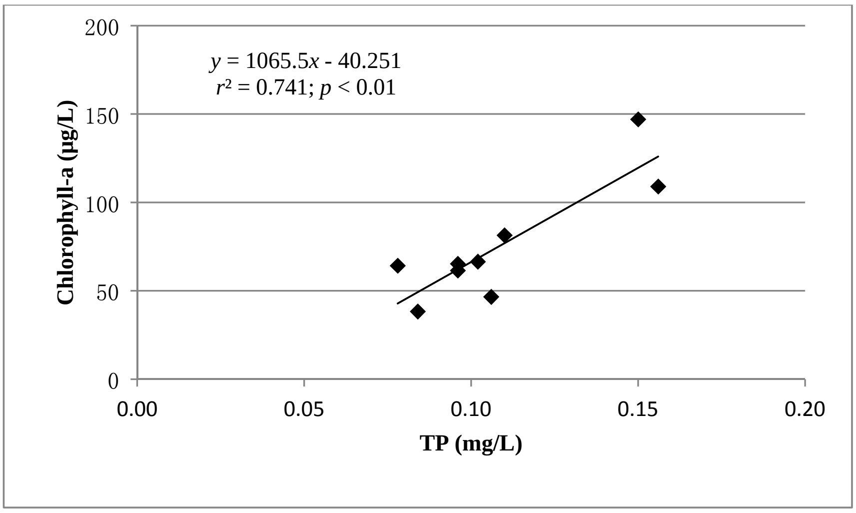

Fig. 5 Relationship between chlorophyll-a and TP in Huntsman Lake.

Secchi disk depths in Huntsman Lake ranged between 0.30 m and 0.38 m, values well below guidance criteria for Virginia lakes (Table 2). To determine the major contributors to a lack of water clarity, optical models are often constructed based on multiple regression of potential contributors to light attenuation (i.e., chlorophyll-a, non-chlorophyll TSS, tannins) vs. the potentially dependent variables of light attenuation coefficients or Secchi disk depths. However, because Secchi disk depths did not vary in any significant manner, this approach could not be used. Instead, the basis for the reduced water clarity in Huntsman Lake was assessed via the construction of an optical model based on equations in the scientific literature. As a first step, light attenuation coefficients were derived from Secchi disk depths [4] by solving for attenuation coefficients as below:

$$Z = -k/1.7$$

where: Z = Secchi disk depth (in m);

k = light extinction coefficient (loss in available light per meter);

1.7 = derived conversion factor.

Then, a previously derived equation [5] was used to derive a partial attenuation coefficient for chlorophyll-a from chlorophyll-a concentrations based on the equation:

$$K\ (\text{chl-a}) = 0.041 \times Chla$$

where: K(chl-a) = partial attenuation coefficient of chlorophyll-a (loss in available light per meter);

Chla = chlorophyll-a concentration (in µg/L);

0.041 = derived conversion factor.

These two equations combined allow for the estimation of the amount of light attenuation that is likely associated with algal biomass (as opposed to suspended inorganic sediments and/or tannins). The results of this assessment are shown in Table 3.

The techniques for developing predictions of light attenuation coefficients from chlorophyll-a concentrations, and for converting Secchi disk depths to light attenuation coefficients, are both imperfect and uncertainty is expected. However, results from most locations suggest that algal biomass is most likely the dominant factor influencing water clarity in Huntsman Lake.

3.2 Initial Sediment Quality

The physical characteristics of the sediments in Huntsman Lake are shown in Table 4.

Sand and silt make up the majority of the sediments in Huntsman Lake, and the mean grain size for lake sediments at all locations is 52.2 µm, which falls in the category of silt. Clay is a relatively minor component of bottom sediments, averaging only 6.4% across the lake. The percent of sand and silt (the major components of bottom sediments) and mean grain size are plotted against water depth in Figs. 6-8, respectively.

Table 3 Chlorophyll-a concentrations, predicted partial attenuation coefficients for chlorophyll-a, and predicted (based on algal biomass) and observed water clarity for Huntsman Lake (*K*-chla = partial attenuation coefficient due to phytoplankton, *Z* = Secchi disk depth in m).

Site	Chl-a (µg/L)	*K*-Chla	Predicted *Z* (m)	Actual *Z* (m)
1	147.0	6.03	0.28	0.30
2	109.0	4.47	0.38	0.30
3	46.6	1.91	0.89	0.38
4	81.4	3.34	0.51	0.38
5	66.5	2.73	0.62	0.38
6	64.2	2.63	0.65	0.38
7	56.6	2.32	0.73	0.38
8	65.3	2.68	0.63	0.38
9	61.5	2.52	0.67	0.38
10	38.3	1.57	1.08	0.38
Mean	73.6	3.02	0.56	0.36

Table 4 Physical characteristics of bottom sediments in Huntsman Lake.

Site	Sand content (%)	Silt content (%)	Clay content (%)	Mean grain size (μm)
1	58.7	38.2	3.1	78.5
2	38.5	57.4	4.0	45.5
3	58.6	38.3	3.1	80.4
4	63.9	32.9	3.2	98.2
5	25.7	67.0	7.3	27.7
6	49.4	46.1	4.7	56.6
7	11.7	77.5	10.9	16.5
8	12.4	73.0	14.5	14.5
Mean	39.9	53.8	6.4	52.2
Std. dev.	21.1	17.3	4.3	31.5

Figs. 6-8 display a pattern whereby larger grain-size particles dominate the shallow portions of Huntsman Lake. These coarser particles appear to settle out in shallower regions of the lake, with silt and (to a much lesser extent) clay comprising the sediments in deeper parts of the lake. This could be indicative of the settling out of sand from stormwater runoff and stream bank erosion in areas close to their inputs and then a reworking of sediments in shallow regions by wave action and subsequent transport of finer-grained sediments to deeper regions of the lake. Chemical characteristics of bottom sediments are shown in Table 5.

The sediments of Huntsman Lake have an average percent organic content of 11.0%, with a relatively narrow range between 8.4% and 13.9%. There was no statistically significant correlation between water depth and percent of organic content. TKN and TP contents averaged 0.168% and 0.063% of dry weight, respectively, with an average sediment TKN:TP ratio of 2.89.

3.3 Post-Circulation Water Quality

Artificial circulation was employed in Huntsman Lake in July 2012. Following the installation of the artificial circulation device, water quality in Huntsman Lake was monitored during summer months in 2012 and 2013 to document any effects of circulation.

As shown below, artificial circulation had a distinct

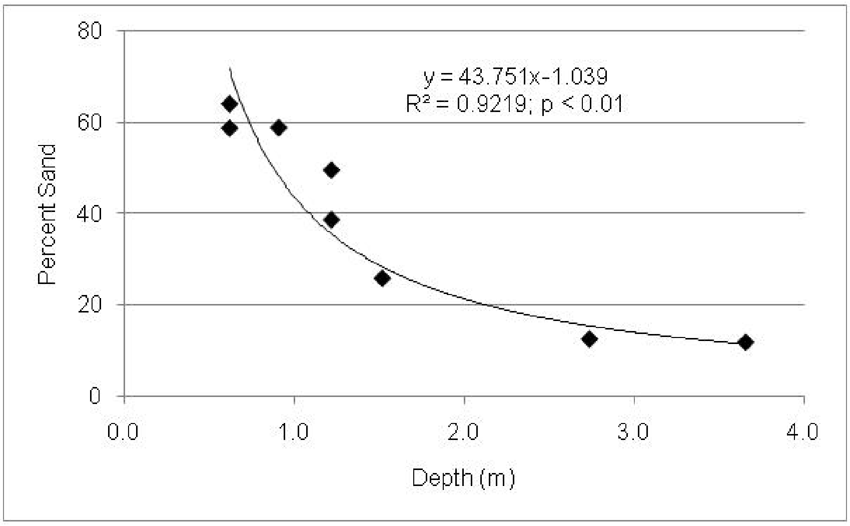

Fig. 6 Relationship between sand content (percent of dry weight) of bottom sediments and water depth in Huntsman Lake.

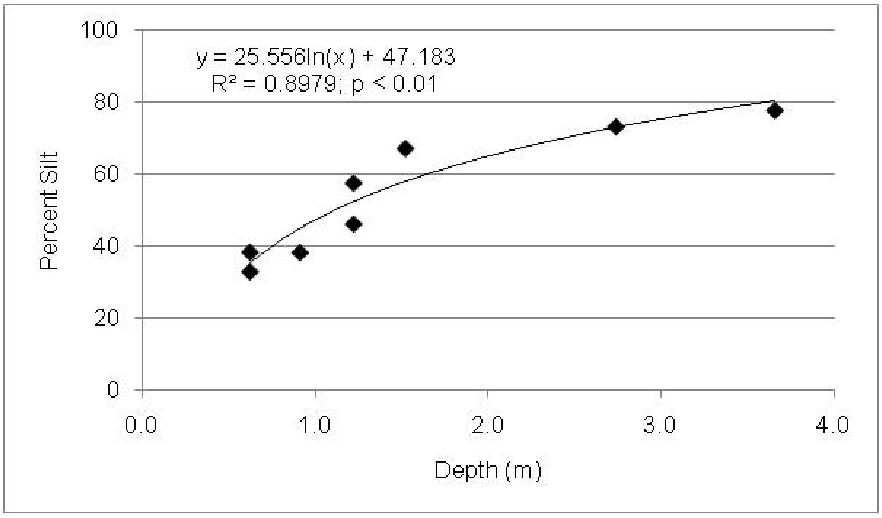

Fig. 7 Relationship between silt content (percent of dry weight) of bottom sediments and water depth in Huntsman Lake.

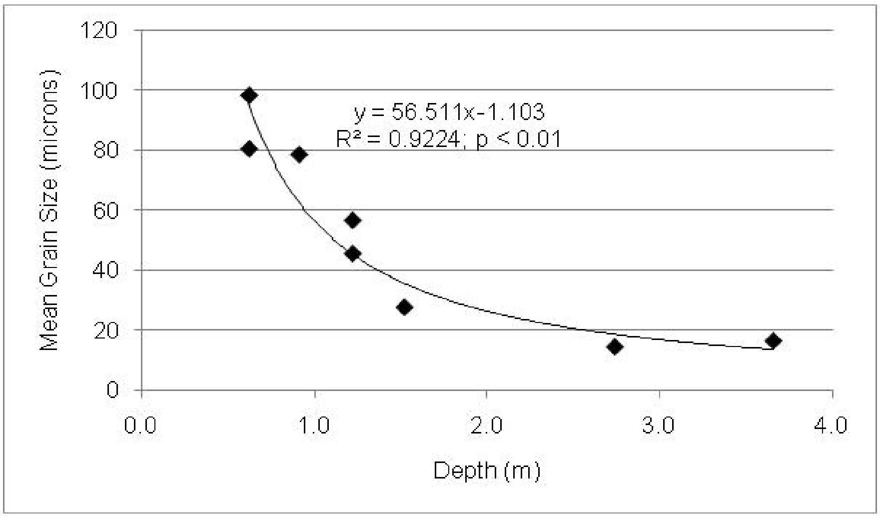

Fig. 8 Relationship between mean grain size of bottom sediments (μm) and water depth in Huntsman Lake.

Table 5 Chemical characteristics of bottom sediments in Huntsman Lake.

Site	TKN (% dry weight)	TP (% dry weight)	TKN:TP	Organic content (%)
1	0.202	0.053	3.81	10.2
2	0.180	0.079	2.27	10.0
3	0.146	0.049	2.99	11.6
4	0.152	0.046	3.33	11.5
5	0.166	0.047	3.56	8.4
6	0.157	0.040	3.92	11.0
7	0.291	0.115	2.54	13.9
8	0.050	0.074	0.67	11.7
Mean	0.168	0.063	2.89	11.0
Std. dev.	0.067	0.025	1.07	1.6

effect on temperature, dissolved oxygen and specific conductance. Pre-circulation conditions for these parameters exhibited moderate to strong stratification patterns (Figs. 1-3). Figs. 9-11 show a much more homogenous distribution for these parameters after artificial circulation was employed.

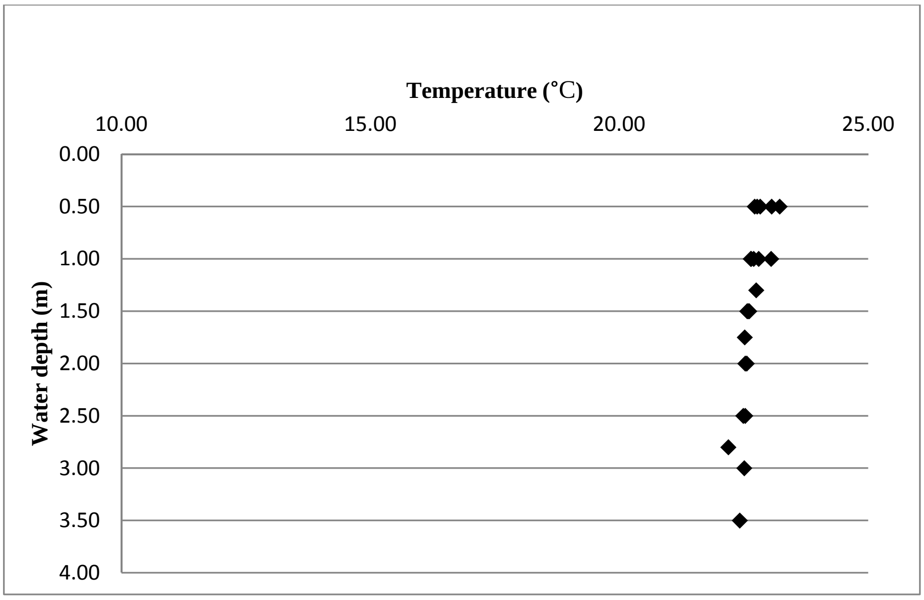

Fig. 9 Relationship between water temperature and water depth in Huntsman Lake (post-circulation) on September 17, 2012.

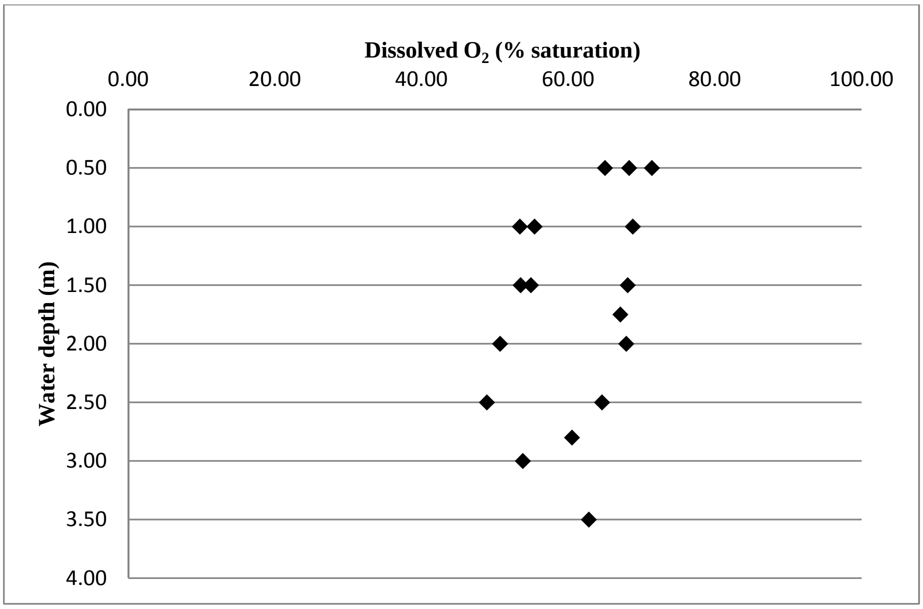

Fig. 10 Relationship between dissolved oxygen and water depth in Huntsman Lake (post-circulation) on September 17, 2012.

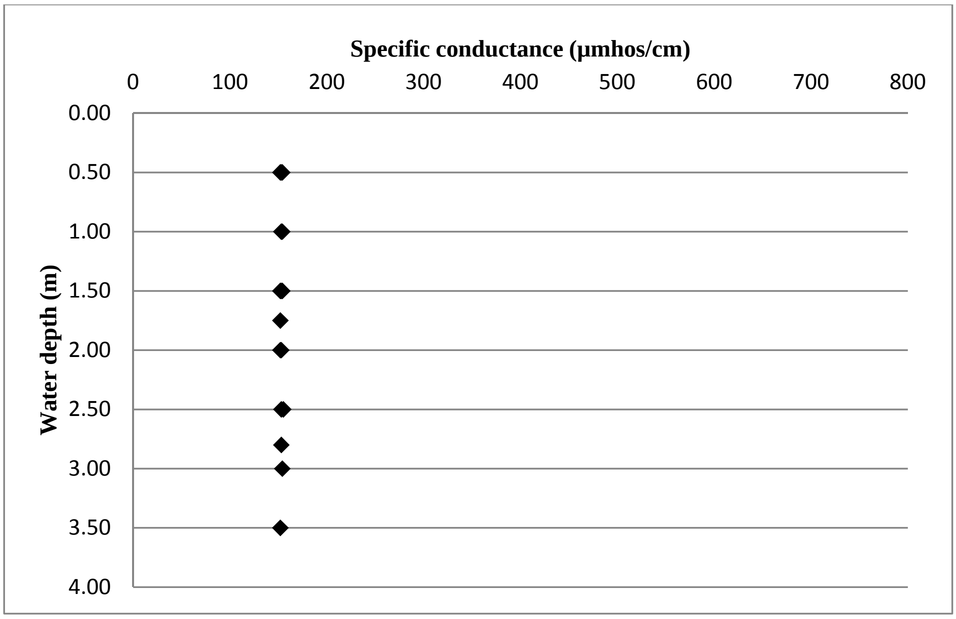

Fig. 11 Relationship between specific conductance and water depth in Huntsman Lake (post-circulation) on September 17, 2012.

Table 6 Water quality data (mean and standard deviation) for total nitrogen, TP and chlorophyll-a for pooled samples collected before and after initiation of artificial circulation.

Parameter	Pre-circulation mean (mg/L)	Pre-circulation std. dev. (mg/L)	Post-circulation mean (mg/L)	Post-circulation std. dev. (mg/L)
Total nitrogen	1.19	0.32	1.12	0.18
Total phosphorus	0.137	0.146	0.071	0.013
Chlorophyll-a	0.057	0.023	0.056	0.021

Prior to initiating artificial circulation, water quality data were obtained in August 2011 and June 2012. After initiating artificial circulation, the same data were collected in July and September of 2012, and also in May and July of 2013. For those six dates (two before and four after circulation was initiated) there are water quality data from five location and depth combinations that were sampled each time. These data are summarized in Table 6.

Results in Table 6 suggest that there was no change in concentrations of TN or chlorophyll-a, using pooled samples from before and after the initiation of artificial circulation. There is an apparent reduction in TP concentrations of 52%, which is in line with expectations in the literature. This reduction in TP concentrations, if sustained, would also be a predicted outcome related to the more thoroughly oxygenated bottom waters seen after implementing whole-lake circulation (comparing Figs. 3 and 10). However, there is a high degree of variability in TP values in the pre-circulation time period, and if a potential "outlier" TP value is excluded from analysis, the change in TP concentrations would be minimal. Water quality monitoring is ongoing to ascertain sustained water quality improvement.

4. Discussion

4.1 Initial Water Quality

Based on the ratio of TN to TP in the water column, phytoplankton biomass in Huntsman Lake would appear to be co-limited by both nitrogen and phosphorus [3]. Chlorophyll-a levels were correlated with both TN and TP, which is consistent with the suggestion of co-limitation of algal biomass. However, there is debate as to whether or not lakes with abundant populations of cyanobacteria (blue green algae) can ever be limited by nitrogen, as they have the ability to utilize atmospheric di-nitrogen gas, thus giving them access to a pool of nitrogen not available to other algal species. Although there is no direct evidence to support or refute the contention that cyanobacteria are in Huntsman Lake, excessive phosphorus loading and elevated phytoplankton biomass are known to enhance the dominance of cyanobacteria in lakes [6-8], and Huntsman Lake has TP and chlorophyll-a values well in excess of proposed lake criteria for this region of Virginia (Table 2).

While various lake restoration activities have been able to improve water quality in lakes with very high levels of phytoplankton—such as Huntsman Lake—those activities have typically involved intensive actions such as a combination of reducing external loads from the watershed and acting on internal loads through processes such as sediment removal, sediment inactivation via whole-lake alum treatment, and artificial circulation [9].

4.2 Initial Sediment Quality

The sediments of Huntsman Lake reflect a pattern wherein coarser-grained particles settle out in shallow areas (i.e., less than 1.5 m in depth) along the periphery of the lake, while finer-grained silt and clay particles—mostly silt—accumulate in the deeper portions of the lake. Despite the dominance of organic compounds in the water column of Huntsman Lake, the bottom sediments of the lake appear to be dominated by inorganic compounds.

Although silt and clay levels within the lake are relatively low in shallow areas, they comprise more than 80% of the dry weight of sediments in deeper areas of the lake, where clay contents alone exceed 10%. In the adjacent Accotink Creek watershed, it was found [10] that more than 70% of the watershed had surface soils with high clay contents [10]. The elevated clay content in deeper waters of Huntsman Lake would thus suggest that stream bank erosion of soils with elevated clay contents could be an important source of sediment loading to the lake. This is also consistent with the conclusion that stream bank erosion was the source of 64% of the TSS loads in the Pohick Creek watershed [11].

The lake’s sediments appear to contain concentrations of TP similar to or higher than levels found in other eutrophic Virginia lakes. Levels of TP in sediments in Huntsman Lake averaged 0.063% of dry weight, while sediment TP values in eutrophic Lake Drummond, in southeast Virginia, averaged 0.25% of dry weight [12]. Sediment TP levels in the hypereutrophic lake Matoaka, also in southeast Virginia, averaged 0.083% of dry weight [13], a value similar to those found in Huntsman Lake. It appears that the phosphorus content of sediments in Huntsman Lake is similar to or higher than levels found in other lakes in Virginia with documented water quality concerns related to nutrient over-enrichment.

4.3 Proposed Lake Water Quality Paradigm

The hypoxic to anoxic bottom waters of Huntsman Lake (Fig. 3) can be problematic not only to the biology of the lake, but to water quality as well. As outlined in a report to VDEQ [3], *“Hypolimnetic oxygen depletion in stratified water bodies may lead to increases in hydrogen sulfide, ammonia, and phosphorus, and the release of reduced iron and manganese from the sediments. If entrained into the productive surface zone, phosphorus may stimulate algal growth, which exacerbates the problem because decaying algae ultimately fuel additional oxygen demand.”*

Thus, the bottom sediments of Huntsman Lake likely do not permanently sequester phosphorus associated with the settling out of organic and/or inorganic materials, leading to a potential positive feedback mechanism perhaps similar to the following:

•External loads of phosphorus and nitrogen to Huntsman Lake result in nutrient-rich waters;

•Nutrient rich waters result in elevated concentrations of phytoplankton;

•Elevated phytoplankton concentrations reduce water clarity;

•Based on predicted light attenuation coefficients, waters deeper than 1.5-2 m do not receive sufficient light for photosynthesis to exceed respiration, leading to a net decline in DO levels in deeper waters;

•Hypoxic to anoxic bottom waters result in a condition wherein phosphorus in bottom sediments is not effectively sequestered;

•Bottom sediments could “leak” phosphorus back into the water column, acting as an internal load to the water column.

Therefore, reductions in external nutrient loads alone may not be sufficient to improve water quality in Huntsman Lake until these internal nutrient pools or removed, isolated in some manner, or work their way out of the system over time.

4.4 Importance of Internal Nutrient Loads

Phosphorus flux from sediments is sometimes sufficient to maintain anthropogenic eutrophication of a lake even when external phosphorus loads have been reduced or eliminated [14]. As such, even after the implementation of large-scale projects designed to reduce external phosphorus loads (e.g., stormwater treatment or stream bank restoration) recovery to an improved condition can be delayed by many years if internal phosphorus loads are not included in restoration plans [14]. Fig. 12 illustrates the time course of events that can result in a delayed response of water quality to a cessation of external nutrient loads.

Consequently, the removal or inactivation of phosphorus-rich bottom sediments in Huntsman Lake could be a required activity, in addtion to reducing external nutrient loads, for the improvement of water quality within the lake.

In order to develop a scientifically valid water quality management plan, an extensive literature review was completed to predict water quality responses to the proposed restoration options of sediment removal, artificial circulation, and sediment inactivation. Adequate documentation was not available for lake or reservoir systems in Virginia, and so information from other locations was relied upon. Most of the available literature on water quality responses to implemented management techniques came from lakes and reservoirs in Florida, Denmark, and Poland.

4.5 Anticipated Benefits with Sediment Removal

The effectiveness of sediment removal as a potential water quality improvement tool had previously been summarized [9] by comparing water

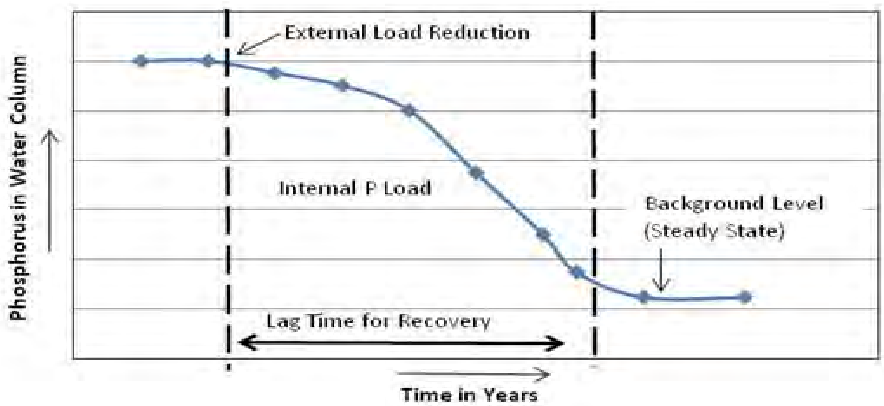

Fig. 12 Potential influence of internal phosphorus loads on phosphorus concentrations in the water column [14].

Table 7 Calculated percent of TN and TP concentration reductions in four Florida lakes after sediment removal.

Lake	Total phosphorus (mg/L)		% TP reduction	Total nitrogen (mg/L)		% TN reduction
	Pre-	Post-		Pre-	Post-	
Banana Lake	1.48	0.81	45	4.74	2.66	44
Lake Maggiore	0.08	0.08	2	2.07	1.80	13
Lake Hollingsworth	0.24	0.33	-38*	4.74	2.69	43
Lake Trafford	0.20	0.24	-20*	2.58	1.97	24
Mean			-3*			31
Median			-9*			33

*Negative percent concentration reduction indicates an increase in concentration.

Table 8 Calculated percent of TN and TP concentration reductions in three Florida lakes after sediment inactivation.

Lake	Total phosphorus (mg/L)		% TP reduction	Total nitrogen (mg/L)		% TN reduction
	Pre-	Post-		Pre-	Post-	
Lake Conine	0.22	0.10	52	2.14	1.49	30
Banana Lake	0.81	0.24	70	2.66	2.57	4
Lake Hollingsworth	0.33	0.13	61	2.69	2.02	25
Mean			61			19
Median			61			25

quality data from pre- and post-sediment removal periods from four Florida lakes (Banana Lake, Lake Hollingsworth, Lake Maggiore and Lake Trafford, Table 7). A substantial reduction in both TN and TP was observed in Banana Lake (44% and 45%, respectively). However, minimal reduction or an increase in TP was observed in Lakes Maggiore, Hollingsworth and Trafford. These results are from sub-tropical lake systems and may not be directly transferable to a temperate lake system (i.e., Huntsman Lake). At this time, data from sediment removal projects completed on temperate lake systems were not available for analysis. On average, a 3% increase in TP concentrations was observed after sediment removal was completed as a restoration technique. The only lake in which a substantial TP concentration reduction was observed was Banana Lake, where the initial TP concentrations (1.48 mg/L) were elevated far above the typical median concentration for Florida Lakes (0.05 mg/L). In contrast, a 31% average percent reduction in TN concentration was observed indicating that sediment removal was more efficient at reducing TN concentrations.

4.6 Anticipated Benefits of Sediment Inactivation

The effectiveness of sediment inactivation as a water quality improvement mechanism in Huntsman Lake was predicted by using a previous summary [9] of pre- and post-sediment inactivation water quality data from three Florida lakes (Lake Conine, Lake Hollingsworth, and Banana Lake, Table 8). A substantial reduction in TP concentrations was observed in all three lakes, ranging from 52% to 70%, with an average TP reduction of 61%. The percent change in TN concentrations was more variable, ranging from 4% to 30%, with an average TN reduction of 19%.

4.7 Anticipated Benefits of Artificial Circulation

Whole lake aeration was originally developed to prevent winter fish kills, but its primary application now is treating symptoms of eutrophication. Artificial circulation is designed to pump deeper water to the lake surface where the water becomes aerated [15, 16]. Artificial circulation also transports phytoplankton biomass to the deeper, light-limited portion of the water column, thereby reducing phytoplankton productivity [17]. The aeration of the sediment-water interface can result in phosphorus adsorption to ferric complexes, resulting in a decrease in phosphorus release to the overlying water column [18] and subsequent reductions in water column concentrations of TP.

There are a number of examples in the literature indicating an improvement in water quality following lake aeration [19-21] with results summarized in Table 9. Results of an artificial circulation system installed in the Lake Persimmon, a hypereutrophic lake in central Florida, were documented [21] and the authors concluded that "…several parameters have shown improvements including the reduction of ammonia and also chlorophyll". A 17% reduction in TP and 34% reduction in TN concentrations were observed in Lake Persimmon (Table 9). Substantial reductions in TP and TN concentrations (75% and 25%, respectively) were recorded in Lake Brooker, Florida [22] after whole-lake circulation was implemented. In Denmark, TP concentration reductions ranged from 38% to 88% in five lakes after implementing artificial circulation, with average TP and TN concentrations reductions of 52% and 30%, respectively [20]. In Poland, water quality improvements as a result of lake aeration in a eutrophic lake were also documented [22] although concentration data were not presented in that report.

4.8 Selection of Water Quality Improvement Projects for Huntsman Lake

Based on the results discussed above, three restoration options were suggested for further evaluation to address the internal nutrient loads of Huntsman Lake: sediment removal, whole-lake alum treatment and artificial circulation. Using the potential nutrient reduction estimates developed for each restoration option, the predicted TN and TP concentrations for Huntsman Lake were derived (Table 10).

The corresponding impact on chlorophyll-a concentrations due to projected nutrient reductions were calculated using the equations developed from linear regressions between TN or TP and chlorophyll-a. The predicted Secchi depth was calculated to demonstrate potential improvements in water clarity.

Based upon the predicted water quality response in Huntsman Lake, sediment removal was not chosen as the initial water quality restoration technique. However, sediment removal will be enacted in 2014, to increase the water storage capacity of the dam.

Table 9 Calculated percent of TN and TP reductions in two Florida and five Danish lakes after artificial circulation.

Lake	Total phosphorus (mg/L) Pre-	Total phosphorus (mg/L) Post-	% TP reduction	Total nitrogen (mg/L) Pre-	Total nitrogen (mg/L) Post-	% TN reduction
Lake Brooker, Florida	0.2	0.1	75	1.6	1.2	25
Lake Persimmon, Florida	0.036	0.030	17	3.4	2.2	34
Lake Hald, Denmark	-	-	88	-	-	-
Lake Vested, Denmark	-	-	49	-	-	-
Lake Viborg Norreso, Denmark	-	-	45	-	-	-
Lake Torup, Denmark	-	-	38	-	-	-
Lake Fure, Denmark	-	-	54	-	-	-
Mean			52			30
Median			49			30

- = data unavailable.

Table 10 Predicted chlorophyll-a and Secchi depth response to proposed restoration actions.

Restoration action	TP (mg/L)	Predicted chlorophyll-a (μg/L)	Predicted Secchi depth (m)	TN (mg/L)	Predicted chlorophyll-a (μg/L)	Predicted Secchi depth (m)
Existing conditions	0.109	80	0.2	1.1	76	0.2
Sediment removal	0.119	87	0.2	0.737	43	0.4
Sediment inactivation	0.043	6	3.1	0.825	51	0.3
Artificial circulation	0.056	19	0.9	0.77	46	0.4

4.9 Actual Water Quality Responses to Artificial Circulation

By alleviating anoxic conditions in the benthos via enhanced circulation, insoluble phosphorus should be more readily bound to sediments and less available in the water column [14, 15]. Furthermore, with a reduced internal phosphorus load, improved water clarity was expected due to expected reductions in the abundance of planktonic algae in the photic zone.

After installation of a whole-lake circulation device, the stratification of the water column in Huntsman Lake had diminished appreciably (Figs. 9 and 11) and substantial improvements were seen the levels of dissolved oxygen in bottom waters (Fig. 10). However, levels of chlorophyll-a and TN showed no significant changes post-circulation, and high amounts of variability in the data complicate the finding of the 52% decline in TP concentrations after implementing whole-lake circulation. Ongoing water quality monitoring is needed to determine if the better oxygenated bottom waters, a positive outcome, will also bring about the expectations of improvements in levels of TP and chlorophyll-a as well.

5. Conclusions

Significant time lags in water quality responses to nutrient load reductions have been well documented in Pohick Bay and Gunston Cove, downstream of Huntsman Lake [23]. Following major upgrades to the primary point source discharging to these two estuarine embayments, water quality improvements (including phytoplankton declines and increases in submerged aquatic vegetation) were not observed until 10 to 15 years after significant point source load reductions were implemented [23]. In locations where external loads have occurred for years to decades, the cessation of such loads will not likely bring about immediate improvements in water quality, as internal loads can strongly influence the length of time before water quality and ecosystem improvements are observed.

Dredging of Huntsman Lake sediments is scheduled for 2014. Following dredging, artificial circulation will be re-deployed and water quality monitoring will continue. The effects of artificial circulation alone will then be compared to the effects of re-initiated circulation on a newly-dredged Huntsman Lake. The authors hope that this will help to answer important questions regarding internal vs. external loads and the consequences of various management actions.

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Sustaining High Nature Value Farming Systems: lessons from the west of Ireland

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Sustaining High Nature Value Farming Systems:

Lessons from the West of Ireland

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High Nature Value (HNV) farming is low intensity farming that supports a high diversity of semi-natural habitats or species. A number of economic and social factors have led to significant declines in HNV farmland. Some policy initiatives to conserve HNV farming systems have been adopted at European and national scales, but it remains to be seen how they will be affected by the upcoming Common Agricultural Policy (CAP) post-2013. Two case studies of HNV farming systems (Smith *et al.* in press) were carried out in the west of Ireland to stimulate interest in this topic and to inform future policy development, and we summarise some of the main findings and recommendations of this study.

Traditional extensive farming practises have helped to create large parts of the European landscape as we know it today. The relationship between farming and natural heritage is one of mutual inter-dependence. Our landscape is the product of millennia of interaction of the natural environment with human land use, particularly farming. Much of Europe's natural heritage is influenced by or has evolved in response to these interactions. Maintaining that heritage relies on maintaining the more traditional, extensive type of farming – HNV farming – that influenced its development, and the farmers' role in this dynamic, is critical.

HNV farming systems are characterised by low intensity, low input management, often including livestock grazing of semi-natural vegetation (see Box). This low intensity management is beneficial for the habitats and species that have developed in tandem over centuries of traditional extensive farming. As only a small proportion of HNV farmland is formally designated for nature conservation, European biodiversity conservation goals cannot be met solely by designating sites. Sustaining the farming practices that maintain and enhance the richness and diversity of landscapes, habitats and species is also required.

Three Types of High Nature Value farmland

- Type I: Farms with a high proportion of semi-natural habitats used for extensive livestock grazing, *e.g.* Connemara, Aran Islands.
- Type II: Farms with smaller areas of semi-natural habitat occurring in mosaic with more intensive agriculture.
- Type III: Intensively managed farmland with little semi-natural habitat that nevertheless supports species of conservation concern, *e.g.* protected bird species.

HNV farmland is often economically marginal farmland. During the 20th century, modern intensive agriculture replaced HNV farming over much of Europe, with resulting biodiversity losses. The remaining area of HNV farmland is in decline, as low intensity farming typically generates poor income, leading to increases in part-time farming and a shrinking population of mainly older farmers. These socio-economic challenges often result in a reduction in farm management, partial or even complete abandonment leading to encroachment by scrub or bracken, with associated losses of biodiversity. Farmers under pressure may also respond by completely changing their land use, including planting forestry or selling their land to developers.

In response to the threats to HNV farming, EU Member States have committed themselves to identify all HNV farming areas in their territories and to put measures in place to protect a significant proportion of them. Currently, the main support mechanism for HNV farming is the European Agricultural Fund for Rural Development (EAFRD), which provides payments for farmers in Less Favoured Areas, compensatory payments for farmers whose lands are part of the Natura 2000 network, and Agri-Environmental Scheme (AES) payments. The European Environment Agency (2004) concludes, however, that 'these instruments… do not appear to be well targeted at high nature value farmland areas'. Thus, these measures are insufficient to stop the loss of biodiversity in Europe. The future shape and level of European funding for HNV farming is currently uncertain. Formal review of the CAP in preparation for the period post-2013 has commenced, and funding levels and baseline environmental standards above which payments will be permitted are emerging as crucial issues.

Case Studies

Until recently, Ireland has made little progress in achieving the aims of identifying and supporting HNV farming, in part due to a lack of information on what HNV farming is in this country. Thus, the Heritage Council commissioned case studies of HNV farming in the west of Ireland to better understand the relationships between biodiversity and associated farming practices and to develop recommendations for measures to conserve HNV farming. The project was overseen by a steering group drawn from the nature conservation and agricultural sectors.

The objectives of the case studies were to collect information on current and past farming practices, the biodiversity of the case study areas, and threats to HNV farming. These data were then used to develop recommendations on measures to sustain HNV farming in the case study areas and elsewhere.

The case study areas selected were the Aran Islands and north Connemara in County Galway. These were chosen as they are of national importance for natural heritage, they are characterised by low intensity farming, and it was considered likely that changes in farming are occurring that could impact on biodiversity. Furthermore, the study areas present contrasts in farming systems and ecology (discussed below) that would allow more generally applicable conclusions to be drawn.

Case study areas

A critical part of the case study methodology was direct engagement with farmers. A series of public meetings was held to gather information on current and past farming practices and the challenges they face in maintaining HNV farming. Their views on nature conservation, agri-environmental policy and potential solutions to HNV farming threats were canvassed. At the meetings, volunteers were sought for a series of more detailed farm-level ecological surveys and interviews.

Thirteen individual farm surveys were carried out in each of the case study areas. The ecological component involved field and desk-based studies of the habitats, flora and fauna on each of the farms. Information on farm management was collected during field surveys and during an interview using a detailed questionnaire. On the Aran Islands, some of the interviews and public meetings were carried out through the Irish language.

Results

Aran Islands

Along with the Burren, the Aran Islands are perhaps the best example remaining in Ireland of a traditional agricultural landscape. The islands are characterised by a multitude of small pastures separated by a network of dry stone walls. The pastures themselves are mosaics of orchid-rich calcareous grassland, limestone pavement and calcareous heath, all EU Habitats Directive Annex I habitats. Dune systems, machair and lowland hay meadows can be found in the lower-lying northern parts of the islands. Raising beef cattle for export and finishing on the Irish mainland is the chief agricultural enterprise. Cattle are wintered in the rocky, species-rich pastures in the south of the islands before being brought down to more improved grasslands around the settlements in the northern half. It is this system of extensive winter grazing that maintains botanically diverse calcareous grassland and heath. In the absence of grazing, these habitats would become invaded by bracken, bramble and other scrub or become rank, low-diversity grassland, thus losing most of their conservation value. In fact, this has already occurred in small unused lanes between fields and in some of the less-managed fields.

Until recently, farming on the Aran Islands was much more self sufficient, with significant amounts of potatoes, vegetables and cereals grown, some for export to the mainland. These enterprises have declined as they have become less profitable and cheap produce from the Irish mainland and further abroad has become more readily available. Potato and vegetable growing for home use is still widespread, but the traditional practice of growing rye for thatching is in severe decline. Rye cultivation has allowed the persistence of a number of rare arable weed species, such as cornflower *Centaurea cyanus*, on the Aran Islands through the 1980s at least; however, their current conservation status is unknown.

North Connemara

North Connemara is a mountainous, agriculturally marginal region with many habitats of conservation importance. The uplands are occupied by upland blanket bog, dry and wet heath and acid grasslands. The lowlands support semi-improved, often wet grassland, lowland blanket bog and wet heath, coastal habitats, conifer plantations and small pockets of semi-natural woodland. A typical north Connemara farm is composed of an area of lowland grassland and a larger area of upland commonage. Lowland grasslands are used for cattle and sheep grazing, with hill sheep pastured in commonage for part of the year. In the recent past, CAP subsidies led to drastic overstocking of commonages with severe overgrazing and erosion of upland blanket bog and wet heath. Mandatory destocking programmes have been put in place to reverse these impacts, but monitoring has found mixed ecological responses to decreased grazing pressure. Many farmers have ceased using their commonage altogether, and there is virtually no cattle grazing in the uplands of north Connemara. There is some indication that localised undergrazing has resulted in scrub encroachment or dominance by purple moor-grass *Molinia caerulea*. However, the situation is complex, and some places may have suffered a long-term decrease in ecological value and grazing potential where overgrazing and trampling has been particularly severe.

As with the Aran Islands, there has been a trend towards simplification of farming practices in north Connemara. Suckler cattle and sheep are the main enterprises, with little tillage or hay-making carried out. The latter has resulted in decreases in the diversity of lowland flora and birds. In addition, mixed cattle and sheep enterprises are declining with many farmers choosing to focus on one or the other. Mixed cattle and sheep grazing can benefit biodiversity of grassland swards due to their different grazing behaviours. Limited poaching by cattle can further diversify swards that under sheep only often develop into close-cropped putting greens.

Threats to Farming

Despite the very different farming systems and ecosystems in the two case study areas, there are many similarities in the challenges faced by HNV farming, suggesting the potential for unified solutions. The main challenges to HNV farming include changes to the agricultural economy and policies that limit farm enterprises and lack of access to markets. These changes include decreases in the price of farm produce, limited markets for many traditional products, and increasing costs in complying with environmental and food safety regulations. In both case study areas, there are no local butchers/abattoirs, largely due to the cost of compliance with strict food safety regulations and

North Connemara HNV farming landscape
Photo: Eamonn Delaney

Aran Islands HNV farming landscape
Photo: Eamonn Delaney

economies of scale. Produce must therefore be sold outside of the region to a limited number of suppliers, and this makes marketing local, conservation-grade produce difficult.

Over the long-term, the numbers of full-time farmers have significantly declined due to economic pressures. Together with greater availability of part-time and full-time off-farm employment opportunities (at least until the recent economic downturn), the labour-intensive farming lifestyle has become less attractive to much of the younger generation. Poor incomes and dissatisfaction with the current state of farming is leading to an ageing and shrinking population of largely part-time farmers. Due to farm consolidation, the average farm in both case study areas is becoming increasingly larger and more fragmented. Larger, more fragmented farms and lack of labour lead to reduced levels of agricultural management. One manifestation of this is the simplification of farm enterprises, with a resulting decrease in habitats and species, particularly those associated with hay meadows or cereal tillage. Partial or complete land abandonment and reversion to scrub is a potential threat. This was not observed on any of the case study farms; however, only actively managed farms were selected for detailed survey, and scrub invasion was observed on other unsurveyed farms in the case study areas.

An additional factor that farmers believe is discouraging younger generations entering farming is the manner in which agri-environmental and nature conservation policies have been implemented. These have been largely implemented in a top-down fashion, with little consultation or discussion with farmers. This has alienated farmers, who state that they do not fully understand the objectives of many of the restrictions and who feel they are unfairly bearing the costs of providing public goods and ecosystem services.

Some positive factors did emerge from the case studies, however. The current generation of farmers love and take pride in farming, and most have no desire to leave farming. They recognise the importance of their role in managing landscapes and ecosystems and consider themselves lucky to live and work in HNV farming landscapes. There is the potential to harness these positive views of their environment and also a desire for greater self-sufficiency to benefit habitats and species dependent on HNV farming systems.

Policy Lessons

The case studies, in addition to other HNV farming projects such as the BurrenLIFE Project (see Jackson 2010 for a summary), generated several policy lessons which are outlined below. They highlight the need for measures to support HNV farming in a more targeted way and to accommodate regional differences in farming practices, habitats and species. The case studies also point towards ways in which links can be forged between vibrant rural communities and nature conservation.

Fostering Better Relationships

Increased communication between local farmers and policy-makers is necessary for farmers to buy into the objectives of agri-environmental and nature conservation policy. Farmers should have greater participation in developing policy measures, including agri-environmental scheme design and commonage destocking, without sacrificing conservation objectives. Greater farmer ownership of policy measures will improve their success as their participation would be more proactive and willing, rather than grudging and reactive.

Identifying HNV Farmland

Supporting HNV farming and associated biodiversity in Ireland requires adequate information. Farming systems and agricultural landscapes that can be considered HNV farmland in Ireland requires clearer definition. Indicators should be used to identify and monitor potential HNV farmland at the national and European scales. Caution should be exercised in mapping HNV farmland to avoid confusing the HNV farming concept with another set of designated areas. Designation of broad ‘HNV farmland areas’ for support runs the risk of providing benefits to intensive farmland within these areas while denying benefits to HNV farmland within an otherwise intensively farmed landscape.

Agri-Environmental Schemes

HNV farmland should be specifically targeted for support under AES, whether as separate schemes or as an element of a wider scheme. Such a scheme should focus on threatened, rare and declining habitats and farmland species and should ensure that the maximum payments made under the scheme are targeted to the farmers whose land is of greatest heritage value. As noted above, scheme entry criteria should be evaluated at the individual farm level. Scheme requirements should be simple and flexible and focus on conservation results rather than strict management methods or prescriptions. This will permit adaptive management by farmers faced with changing weather and economic conditions. A focus on results rather than management prescriptions will require training to improve the conservation management skills of farmers and scheme assessors. In the current economic climate, this presents funding challenges.

As the case studies demonstrate, there are significant differences in farming and ecosystems among different HNV farming areas. Thus, different versions of the scheme should be tailored for areas with differing farming systems and ecosystems in conjunction with local farmers. Objectives and

Surveyed farm in North Connemara
Photo: Eamonn Delaney

Scrub encroachment in calcareous grassland on the Aran Islands
Photo: The Heritage Council

criteria will need to be tailored to take into account differences in the habitats and species present and their abundances in different biogeographical regions. Payment levels will also need to reflect differences in the ecosystems and farming enterprises that occur within HNV areas. It is imperative that an ecological monitoring and evaluation programme be built into an HNV farming AES at the earliest stages to ensure the scheme meets its objectives.

Although an AES or element of an AES targeting HNV farming should be available to farmland irrespective of nature conservation designations, taking designations into account when designing scheme objectives can provide added conservation benefits. Appropriate management of HNV farmland adjacent to a designated site may be of critical importance to its favourable conservation status, *e.g.* farmland adjacent to an estuary designated for wintering wildfowl that provides a critical feeding ground. Thus, designated sites should be considered when the options for each applicant farm are being agreed. This issue of connectivity will also be particularly important given the potential impact of climate change on the distribution of habitats and species and the need to cater for these changes in conservation planning.

Spring gentian* Gentiana verna *in calcareous grassland
Photo: Eamonn Delaney

Marketing HNV Produce

Proactive marketing of produce from HNV farmland is required to improve financial viability. Conservation-grade branding should be employed to obtain a premium price reflecting the biodiversity benefits and food quality of these farming systems. In many cases, advertising campaigns will be required to raise consumer awareness of conservation-grade produce, and producer groups should be formed to ensure consistent supply of quality produce. Local butchers should be established to facilitate this process, and State aid will probably be required to enable compliance with food safety regulations. The potential for diversifying farm enterprises to add value to conservation-grade brands and to support biodiversity should be investigated.

Next Steps

The case study results will be used to inform the development of policy advice by the Heritage Council. They are also the subject of ongoing work on HNV farming in Ireland by the European Forum for Nature Conservation and Pastoralism, with particular reference to the Aran Islands, north Connemara and the Iveragh Peninsula, supported by the Heritage Council. The operation of the recently implemented Burren Farming for Conservation Programme will also provide useful lessons in how such schemes can be designed and implemented in other HNV farming areas. This programme targets the delivery of a range of environmental benefits, in particular production of species-rich limestone grasslands and improvement of water quality, and aims to support HNV farming in the Burren, continuing and mainstreaming the findings of the BurrenLIFE Project.

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Kat Stanhope and John Box

Post-industrial and brownfield habitats

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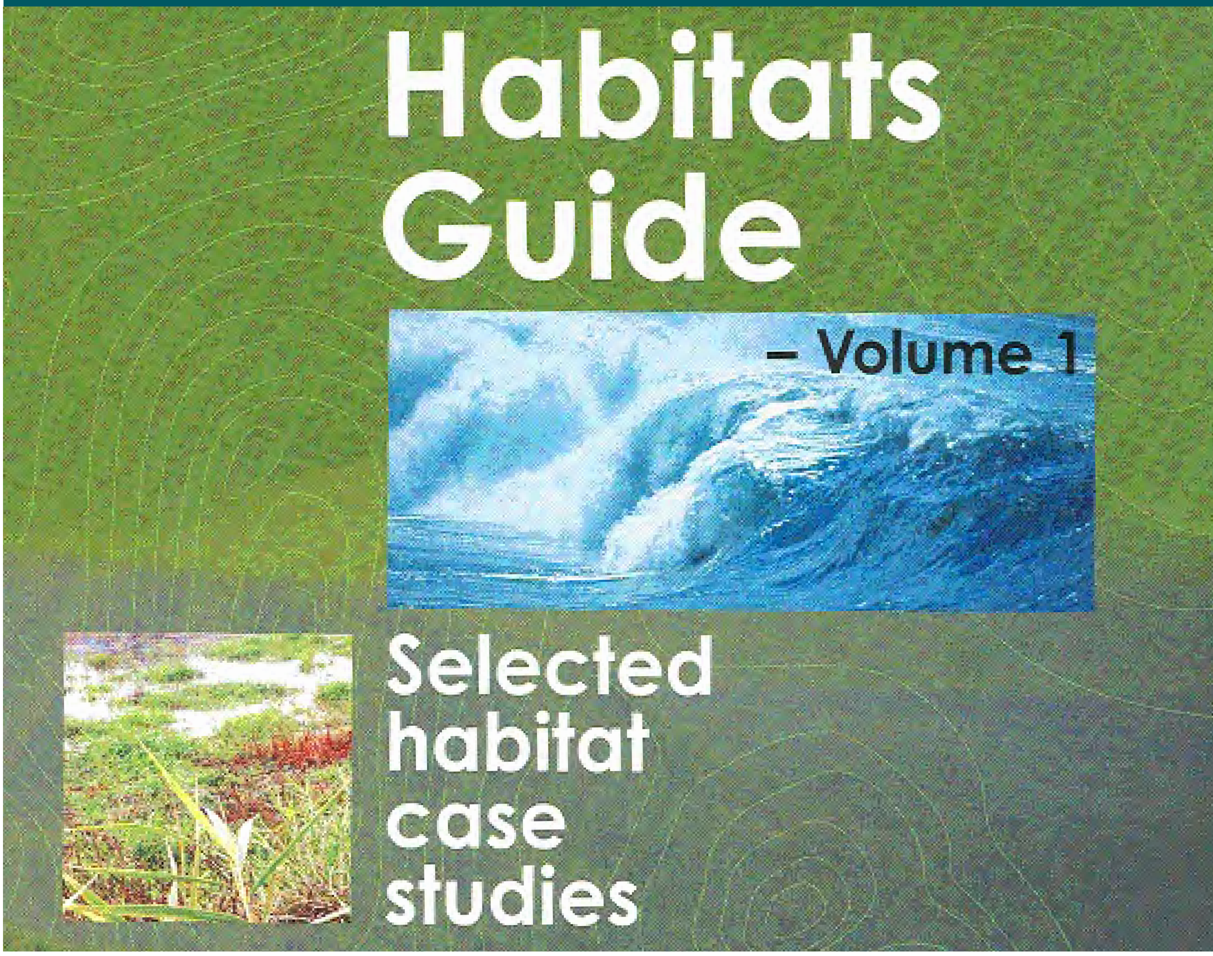

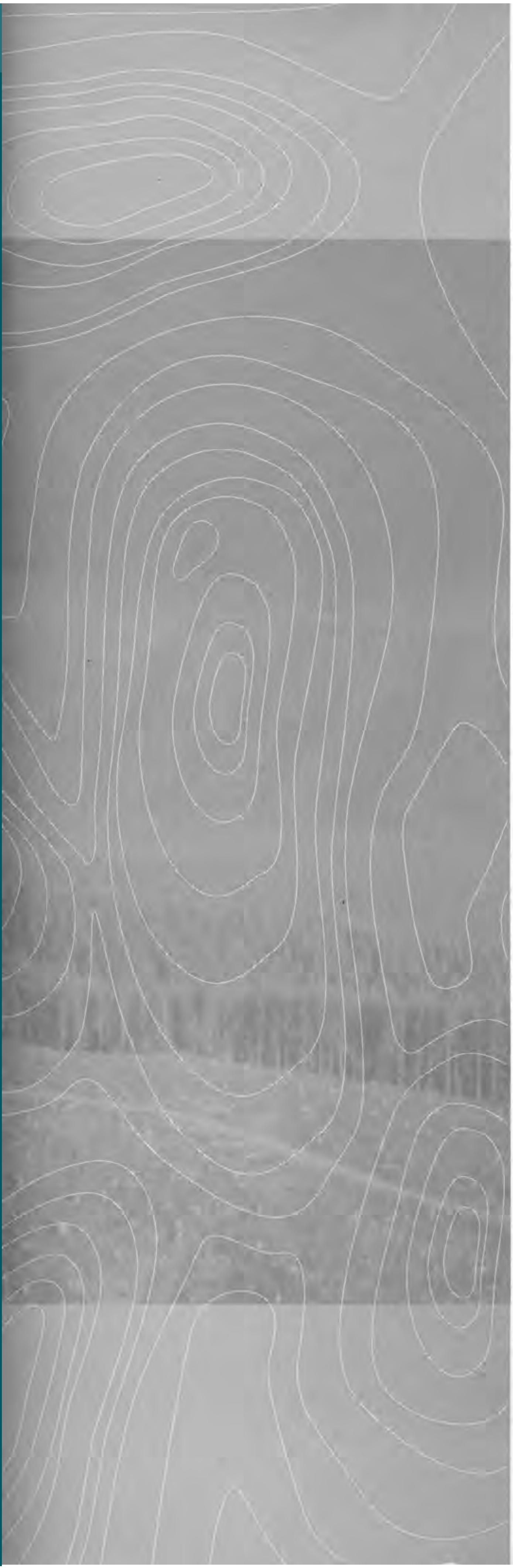

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CHAPTER FIVE – POST-INDUSTRIAL AND BROWNFIELD HABITATS

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INTRODUCTION

Abandoned quarries, pitmounds, spoilheaps, industrial sites and railway lines provide a multitude of potential wildlife habitats created by variations in substrate, slope and aspect. Environmental stresses, such as lack of water and nutrients and extremes of acidity/alkalinity, favour the development of species-rich communities of plants and animals because no one species is able to dominate. Nature can colonise these habitats in unpredictable ways and such sites may contain locally rare species. These colourful and valuable wildlife habitats can help to maintain biodiversity in a local area. Linear features such as disused railways and canals provide green corridors for both people and wildlife.

Natural recolonisation of abandoned workings is not a modern phenomenon. The Neolithic flintmines at Grimes Graves in Norfolk are noted for the species-rich grassland that has developed on the spoil from the shafts excavated down through the chalk, and the waterbodies that make up the Broads are flooded medieval peat-workings. Some post-industrial sites such as spoil heaps may need restoration, as conditions are so harsh that revegetation is patchy or does not occur. However, this paper does not deal specifically with the practical methods of reclamation of post-industrial sites, as the problems encountered on sites are numerous and measures need to be site specific.

Urban habitats, which include post-industrial habitats, are listed on the UK Biodiversity Action Plan (UKBAP) as well as many local BAPs for areas such as Nottinghamshire, which has a legacy of industry including coal mining and the resultant spoil tips. The biodiversity value of these sites has, until recently, been underestimated. Features of these sites that result in a diverse flora and fauna include the lack of intensive management, variations in topography, soil type and drainage, and freedom from disturbance.

THE WILDLIFE OF BROWNFIELD SITES

Brownfield sites are often associated with severe disturbance to the soil structure and extremes of environmental conditions, such as: high or low pH, varying soil particle size (bricks down to fines), extreme nutrient levels, poor drainage leading to flooding or free drainage with no fines to hold water leading to very dry conditions, and toxicity (Bradshaw & Chadwick, 1980). The development of vegetation on such sites is partly influenced by these factors and partly depends on which plant species colonizes first and gains a local dominance (Gilbert, 1989).

The combination of all these factors operating over time can result in a wide range of flora and fauna on post-industrial sites, which often comprise communities that occur early in the process of natural colonisation and succession and may well include locally uncommon and rare species. Indeed, a survey of six counties in England showed that mineral workings are associated with some 13% of their biological Sites of Special Scientific Interest; the proportion ranged from 4% in Devon to 25% in Durham, where the grasslands that develop on magnesian limestone workings are very important for nature conservation (Coppin & Box, 1998).

Brownfield sites can provide habitats for plants and animals that can extend their range. The classic example is the spread of Oxford ragwort along the railway system. Other examples are: the extension of the ranges of little ringed plover and great crested grebe through colonisation of old gravel workings; the discovery of alpine clubmoss — normally found in mountainous areas in Wales, the Pennines and Scotland - on Coal Measures spoil in lowland Shropshire in the 1980s; and the colonies of two orchids (green-flowered helleborine and narrow-lipped helleborine) growing on spoil from abandoned lead mines in Northumberland, well to the north of their usual locations in Britain.

The range of microhabitats and early successional stages on brownfield sites can give rise to a diverse invertebrate fauna where the typical coloniser is likely to be on the edge of its global range (restricted to the south), naturally occurs in more lowland habitats and is nationally scarce rather than rare (Gibson, 1998). Brownfield sites can also be valuable where a species has lost or is losing its natural habitat and can act as a refuge, especially for invertertebrates such as the dingy skipper that inhabits post-industrial sites in Nottinghamshire.

NON-RECREATABILITY

Unusual biological communities can be so intimately linked with the particular industrial process that created them that they can be considered to be unique and non-recreatable. For example, strongly alkaline waste from the obsolete Leblanc process for manufacturing washing soda (sodium carbonate) was tipped at Nob End on the outskirts of Bolton in Greater Manchester between 1835 and 1885. The caustic surface layers (originally with a pH of 12 or more) have slowly weathered to give an infertile calcium-rich substrate with a pH of around 8.5. Natural colonisation has resulted in a species-rich vegetation noted for the abundance and variety of its orchids (particularly the masses of fragrant orchids) as well as a wide range of other lime-loving plants (Shaw & Halton, 1998).

Another example is furnace clinker and slag wastes from the early blast furnaces. Furnace slag was formed when limestone was added to the blast furnace to act as flux and combine with impurities in the molten iron. The slag was a versatile waste material that was crushed and used in road construction and in agriculture (the high calcium content could be used to neutralize acidic marginal soils). Large boulders of clinker and slag waste, which may be up to 100 years old, are only now found at a few sites in Britain (for example, Blists Hill in Telford and Moorcroft Wood Local Nature Reserve in Walsall). These boulders have become colonised by a distinctive flora that includes hare s-foot trefoil, purging flax, wall pepper and ivy-leaved toadflax.

Pulverised fuel ash (PFA) is produced by coal-fired power stations and is associated with vegetation which demonstrates the fascinating processes of natural colonisation and natural succession that reflect the changing environmental conditions of the waste as it weathers (Shaw 1994). Initially very alkaline with high boron and salt concentrations, leaching by rainwater reduces the boron and salt levels in the upper layers. The first plants to colonise are salt-tolerant species such as spear-leaved orache, a saltmarsh plant. Subsequently, the plant species that colonise tend to be those associated with calcareous and base-rich habitats, such as orchids, yellow-wort and centaury. Eventually, woodland species start to colonise and the notable orchid-rich wild flowers are progressively lost as the shade increases due to the closure of the woodland canopy. PFA is not currently a unique non-recreatable waste, as it is still being produced in significant volumes by power stations, but the industrial process is likely to be limited as it is replaced by other technologies in due course. In the future, PFA tips will be a significant source of interesting plant communities demonstrating stages in the natural succession on this particular waste from bare ground to woodland.

ARCHAEOLOGY

Brownfield sites may retain architectural and technological monuments from industrial processes which have ceased. These can include blast furnaces and furnace walls, boulders of furnace slag, pumping houses, pit-head and winding gear, pitshafts and tunnels, bridges, mineral railway lines, canal basins and wharves. Industrial archaeologists value these redundant structures and there are museums where these form valued exhibits, such as the Black Country Museum in Dudley and the Ironbridge Gorge Museum in Shropshire.

Wildlife will also seek opportunities to colonise these features. For example, bats and badgers may live in old structures, great crested newts can breed in canal basins, birds of prey can nest on furnace walls and chimneys and plants growing on walls and mineral railway lines can be spectacular as well as unusual. Such industrial monuments can be of great value to local people who see them both in their present condition and as a link with the past. It is vital that proposals for reclaiming post-industrial landscapes involve ecologists and industrial archaeologists (Box, 1999).

THE SOCIAL VALUE OF RECOLONISED BROWNFIELD SITES

The modification by nature of brownfield sites often leads to the use of these sites for a range of recreational purposes, which are almost invariably informal. For instance, in a study of colliery waste heaps around Wigan in Greater Manchester, Molyneux (1963) refers to one known as Industrious Bee (South) as an undulating site which is very popular as a children s playground . Brownfield and post-industrial sites provide opportunities to experience wildlife and even wilderness, not only in urban and urban fringe areas, but also where these sites occur in the countryside (because changes in agriculture and forestry over the past 40 years have led to a reduction in the diversity and abundance of wildlife habitats). Some very informal recreational uses can be destructive, such as the prevalence of dirt bikers using coal tips and spoil heaps, although the bikes can produce an earlier successional stage on the disturbed areas of spoil.

Studies of the nature of the relationship between local people and the natural world have indicated that the presence of wild areas as an integral part of the urban fabric is very important to people (Rohde & Kendle, 1994). Wildlife should be accessible to people on an everyday basis and the conservation of wildlife in ordinary environments is likely to be just as important to most people as the conservation of wildlife in outstanding and exceptional environments in the deepest parts of a shire county. However, the reclamation of abandoned industrial sites to an open-space or soft end-use, as opposed to a commercial or hard end-use, often involves the creation of a landscape where standard trees march across acres of mown grass covering well-graded features with smooth contours. This reduces the potential for a diversity of habitat and flora and fauna.

THE INTEGRATION OF DEVELOPMENT WITH THE ENVIRONMENT

Industrialization has left a legacy in terms of dereliction and pollution that is providing a challenge for reclamation and redevelopment. Since April 2000, Part IIA of the Environmental Protection Act 1990 has provided a new regulatory regime for identifying and remediating contaminated land, with particular duties for local authorities and the Environment Agency. Problems range from the pollution of watercourses by discharges from abandoned mines to the ingestion of heavy metals in dusts from mine dumps by inhalation and on foodstuffs. Solutions to these challenges, such as removal to landfill or deep burial of contaminated materials, may not sit easily with maintaining wildlife habitats that have developed over the years.

Nevertheless, the quality of the urban landscape, including the contribution made by open spaces of all kinds including brownfield sites to local biodiversity, is a decisive factor for new development and investment (Box & Shirley, 1999). Successful economic development needs to make allowances for the retention of key post-industrial features for educational, recreational and inspirational purposes and for nature conservation. The environmental services and community benefits provided by open spaces and brownfield sites need to be appropriately valued and owners and managers have a duty of care in respect of wildlife and natural features.

Research by the Department of the Environment (1989) into the management of reclaimed derelict sites looked at the economics and performance of various soft end-uses: public open space, agriculture, formal recreation, woodland and nature conservation. The most cost-effective sites in terms of capital costs were those restored to a nature conservation end-use because of low capital costs resulting from the use of existing vegetation. Large public open space schemes (>10 ha) and woodlands were cost-effective, the latter being due to the use of *in situ* soils. In addition, model maintenance costs for these sites were derived from an assessment of the maintenance required using standard costs: nature conservation, established woodland and large open space schemes had the lowest maintenance costs.

This study provides good economic reasons for seeking a nature conservation end-use, where appropriate, as well as the incorporation of an ecological rationale into other restoration objectives (such as improving the design of woodland projects and sympathetic and naturalistic landscaping). A land suitability approach to after-use options for land reclamation has been set out by Coppin & Box (1998) that takes account of key environmental issues, including the steepness of the slopes and the soils, and pays particular attention to nature conservation as the end-use. In addition, there is an excellent handbook covering the assessment, protection, creation and management of nature conservation features in the context of reclamation (Land Use Consultants/Wardell Armstrong, 1996).

RECLAMATION

A useful distinction can be made between passive and active reclamation and habitat creation. Passive reclamation allows natural colonisation and natural succession to produce vegetation communities that reflect the underlying soil conditions and the age of the site; the succession may pass quickly to woodland or it may be checked by severe environmental factors and produce early successional communities. Active reclamation involves the use of soil amelioration and selected species or strains of plants to produce a quick revegetation for amenity, aesthetic or practical reasons (such as slope stabilisation). Any reclamation project, whether active or passive, needs to take account of the process of natural colonization on derelict sites so that there is recognition of the potential for interesting habitats to develop that are species-rich and may contain notable species of flora and fauna.

Variety in the landscape is valued by people whether in a rural or an urban setting. Moreover, the greater the variety in the landscape, the greater the biodiversity. For example fields and woods woven together with hedges, streams and stone walls will be associated with more wildlife than expanses of intensive agriculture. The creation of landscapes needs to use variations in topography, soils and drainage, as well as habitat creation techniques which take account of the local landscape and its wildlife habitats. Environmental stresses can be minimised by using the substrates present on a site without the import of topsoil and fertilisers. One key test of the value of such artificial landscapes might be whether a group of children could play an exhilarating game of hide and seek there.

HABITAT MANAGEMENT

The multiple values to society inherent in post-industrial landscapes demand the use of a range of techniques to ensure the management and enhancement of existing features as well as the development of new features (Shaw, 1998). Soil mapping and classification is fundamental to habitat creation, restoration and management on urban and brownfield sites. Existing soil classification schemes were not completely satisfactory for practical application in urban environments and a new framework for classification, description and mapping of urban soils has been developed by Hollis (1992) which can be readily applied to brownfield sites and their surrounds.

The determination of the key aims - the ideal management objectives - may need great vision on some post-industrial sites as the direction of the natural succession is often not clear and interesting species may not colonize until conditions are favourable. Moreover, the location of the site in the context of other open spaces needs to be established as does the contribution of the site to green networks of public and private open spaces (Barker, 1997). The development of management objectives within the format of a management plan provides a basis for deriving management costs and priorities. The objectives, once agreed by all interested parties, are relatively independent of changes in staff or expert advisers.

There is not yet a body of management experience of post-industrial sites as there is with other semi-natural habitats, such as woodlands or meadows, where the desired vegetation type can usually be agreed upon. Climax vegetation such as woodlands can be managed by controlled neglect or planned non-intervention. Heathlands and grasslands need intervention which stops the natural succession to woodlands. The early successional communities typical of many brownfield and post-industrial sites may need considerable intervention if the appropriate conditions for continuous colonization and succession are to be maintained.

There are often practical problems in managing the communities which develop on post-industrial sites. Grazing may not be possible due to the expense of fencing, the lack of water supply, or the availability of suitable stock, and mowing may not be feasible due to the uneven nature of the terrain and the need to remove cut vegetation afterwards. Local people become attached to and proud of their sites and may not take kindly to efforts to recreate early successional stages by bringing the bulldozers in again.

THE WAY FORWARD

There is considerable pressure nowadays to restore post-industrial features rapidly and, in particular, to green them as quickly as possible for the sake of appearance. Natural regeneration should be incorporated into reclamation schemes so that there will be a range of interesting post-industrial landscapes in the future. For example, today it would be hard to see the waste from the Leblanc process at Nob End in Bolton (discussed earlier in the chapter) being allowed to remain unreclaimed for the time needed for the processes of weathering and natural succession to produce the unusual communities for which the site is now valued. We value the semi-natural habitats of the agricultural age (for example, hay meadows and heathlands); we must learn to value the semi-natural habitats of the industrial age.

Novel habitats such as alkaline waste, pulverized fuel ash and metal-rich mine wastes should be allowed to express themselves in terms of the communities which become established, but not at the expense of other environmental issues such as pollution due to spoil. A commitment to natural revegetation in such circumstances will mean compromises by both planners and landscape architects (many of whom would like to green the sites immediately for the purpose of visual screening) as well as conservationists, who will have to decide which part of an unusual, and probably unknown, natural succession should be maintained.

The reclamation of derelict and degraded land to open spaces and landscapes over the past decades has been a significant achievement. It is regrettable that, in some cases, wildlife habitats and geological features have been reclaimed to open spaces with mown grass and planted trees. The few studies of what people want from local open spaces show that diversity of opportunities is important. The apparent strong desire of many professionals for wooded landscapes needs to be tempered by an appreciation of the value of existing wildlife habitats and geological features. Diversity is what people want from landscapes and greenspace. The reclamation and restoration of brownfield and post-industrial sites should involve an appreciation of the services provided by the environmental resources and a philosophy where caring for land is important (Leopold, 1949; Darling, 1970; Bradshaw & Chadwick, 1980).

CASE STUDIES

Case Study 1 - The Regeneration of Gibfield Park, Greater Manchester

WS Atkins Consultants Ltd was commissioned by Black Country Properties Ltd to prepare a comprehensive masterplan, undertake an Environmental Impact Assessment and prepare and submit a detailed planning application for a proposed mixed development scheme in Greater Manchester. The proposed development site comprised a former opencast coal site, colliery spoil heaps and a former brickworks (see Figure 1, page 53).

The key objectives of the proposals were the comprehensive remediation and restoration of derelict and despoiled areas followed by redevelopment to release areas for residential and business development and community use. Problems with the site included the presence of contaminated spoil, steep slopes and erosion, and ochreous run off causing pollution of a stream. Visual intrusion of the spoil heaps was also a problem.

Ecology surveys identified existing habitats of value including local Biodiversity Action Plan (BAP) habitats, a Site of Biological Importance, acid grasslands, wetlands, species rich hedgerows and heath growing on acidic colliery spoil. There was also a population of great crested newt, a legally protected species, using former balancing ponds and other smaller water bodies.

A comprehensive masterplan was prepared for the whole development, including a landscape framework for the residential area and the design of 60 ha of public open space, which incorporated a community woodland and other habitats of ecological value (see Figure 2, page 53). There was a significant ecological input to the masterplan design, using the Greater Manchester BAP as a guide. This included the creation of new water bodies and the retention and enhancement of balancing ponds for the existing great crested newt population. Where possible, the new landform was designed to retain and enhance existing valuable wildlife habitats.

Planning permission was granted in August 2002 and the restoration of Gibfield Park is on-going at the time of writing. Contaminated spoil present on the site has been re-used on-site to fill the void created by open cast coal extraction, and then buried under uncontaminated spoil acting as a cap .

The next stage is to redistribute the mounded overburden, sub-soil and top soil around the rest of the site to give a more natural and unobtrusive profile. The design of the landform ensured that no material would be transported off-site. Topsoil spreading will be largely restricted to the main tree planting areas, with overburden and sub-soil forming the substrate for habitats such as acid grassland and scrub habitats. An ecological management plan was produced for on-going maintenance, management and monitoring, which will be carried out by specialist sub-contractors.

Figure 1 - Aerial photo of Gibfield Regeneration site showing extent of land to be restored/developed

Figure 2 - Masterplan for Gibfield Regeneration showing the main ecological design features

Case Study 2: Reclamation and Redevelopment of Shirebrook Colliery, East Midlands

WS Atkins Consultants Ltd has recently produced the landscape design for Shirebrook Colliery Reclamation. The site, which is on the border of Derbyshire and Nottinghamshire, is owned by East Midlands Development Agency (EMDA) and is part of EMDA s overall South Shirebrook Regeneration Project. The colliery site (shown in Figure 3, page 55) comprises acidic coal spoil, although the underlying geology is magnesian limestone. The site has planning permission for a mixed end-use, including industrial units, public recreation and wildlife habitat, and covers 400,000m^2.

Problems encountered include steepness of the existing spoil, acidity and large particle size of the spoil, spontaneous combustion of spoil at depth and generally poor vegetation colonisation. To solve these problems prior to redevelopment, the tip is being reprofiled (overturning and cooling combusting spoil in the process) and soil ameliorants are being added to neutralise acidity. To create a high quality landscape, existing sub-soil and top-soil will be reused to form the planting medium. The eradication of Japanese knotweed is also being implemented.

The landscaping proposals were developed by an Atkins landscape architect, working closely with Atkins ecologists, local council ecologists and landscape architects, and incorporating suggestions from the local wildlife trusts. The scheme includes the creation of:

- Species-rich calcareous wildflower areas seeded on crushed natural magnesian limestone;
- The reconstructed drainage system incorporating the Shire Brook, part of which will be de-culverted;
- Reed beds, open ponded water and limestone cascades, along with varying bank profiles from shallows to almost vertical limestone banks and various water depths to increase the diversity of aquatic habitat;
- Woodland areas planted with fast growing nurse species tolerant of conditions found on coal spoil, such as birch for screening purposes mixed with a range of broad-leaved native tree and shrub climax species;
- Grassland areas seeded with a fescue grass bridleways mix;
- Public areas including footpaths and cycle routes, a grassed amphitheatre seating area, and interpretation boards giving information on available routes and the history and ecology of the site.

Five year ecological management plans have been produced for the landscaped areas, to encourage and maintain the proposed vegetation types.

The completion of the landscape design is due in late 2003. Figure 4 (on page 55) shows how the colliery will look following the implementation of the landscape scheme.

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Figure 3 - Photomantage of existing site conditions, with the railway line separating the Derbyshire and Nottinghamshire sides of the Colliery

Figure 4 - Photomontage of how the colliery will look following the implementation of the landscape scheme

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Translocating wildlife habitats: a guide for civil engineers

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Translocating wildlife habitats: a guide for civil engineers

There is increasing public and regulatory pressure on civil engineers to retain or replace established wildlife habitats during project design and delivery. However, they can also be moved. Habitat translocation is an effective and long-standing technique that can be used to rescue or salvage homes for wildlife which would otherwise be lost. This paper presents three case studies which demonstrate how civil engineers can successfully translocate and retain habitats on site, albiet in different locations, resulting in wide-ranging project benefits.

Wildlife in the UK is subject to continuing pressures due to the loss of and damage to habitats. Valuable habitats for wildlife, such as hedges, species-rich grassland, ponds and mature trees, should be retained in situ in the planning and design of infrastructure, built development and mineral-extraction projects. However, this may not be possible because of physical constraints to site design and layout, access to the site, and commercial and financial reasons.

Figure 1. Mature hedges, which provide landscape structure, visual screening and wildlife corridors, can be excavated and moved to a new location rather than being cut down and chipped

The 2010 international year of biodiversity, and the growing awareness of wildlife issues throughout the development and construction industry, means there is an increasing challenge for civil engineering project managers to retain wildlife habitats within a site. Biodiversity and wildlife can result in good media stories and provide useful publicity about a project.

Habitat translocation is a simple tool that can be used to rescue or salvage wildlife habitats which would otherwise be lost and to incorporate these into the design and layout of new infrastructure, built development or mineral-extraction projects. Mature hedges that provide landscape structure, visual screening and wildlife corridors can be excavated and moved to a new location rather than being cut down and chipped during site clearance operations (Figure 1). Species-rich vegetation and its associated animals can be moved to new green areas and open spaces rather than bulldozed into topsoil mounds. Wetland vegetation and invertebrate-rich sediment can be taken from an existing pond that needs to be filled for development and moved to a new pond.

Established hedges, species-rich vegetation and ponds are valuable wildlife habitats and it can take years for newly created, planted and sown habitats to attain the same degree of maturity and complexity. Translocation ensures that native species of local provenance are used rather than imported plants.

New legislation driving change

Legislation is driving change through new legal duties placed on public authorities and public bodies in Britain to conserve biodiversity in carrying out their functions (Nature Conservation (Scotland) Act 2004, 2004; Natural Environment and Rural Communities Act 2006, 2006).

Public authorities – including government ministers, government departments, local authorities and statutory undertakers – in England and Wales must have regard to the purpose of conserving biodiversity in carrying out their functions, and in Scotland public bodies and office-holders must further the conservation of biodiversity (a tougher form of words). This is often achieved through the land-use planning system.

Lists of habitats and species of principal importance for the conservation of biodiversity have been published for England (BARS, 2008), Scotland (Biodiversity Scotland, 2005) and Wales (Wales Biodiversity Partnership, 2009). Such habitats may support species given legal protection by the Wildlife and Countryside Act 1981 (1981) (as amended) or the Conservation (Natural Habitats, &c.) Regulations 1994 (HMG, 1994) (as amended and as consolidated for England and Wales by the Conservation of Habitats and SpeciesRegulations 2010 (HMG, 2010)) (see also Abbatt, 2007). These habitats and species are a material consideration when assessing a development proposal that, if carried out, is likely to result in harm to the species or its habitat.

Dealing positively with wildlife issues can help to demonstrate to regulators and key voluntary bodies that a project has good environmental credentials.

Well-known projects which have involved major habitat translocation include

- Manchester airport second runway
- Stansted airport
- Channel Tunnel rail link
- Heathrow Terminal 5
- M4 extension at Twyford Down
- M6 toll road

as well as mineral extraction at Durnford quarry and Thrislington plantation (Anderson and Groutage, 2003; Box, 2003; Bullock, 1998; Palmer and Wilbraham, 2008; Roberts, 2000; Trueman *et al.*, 2007). These projects featured large-scale habitat translocations that were specific to each scheme and which were carefully designed and planned into the overall programme.

Translocation methods and options

Smaller-scale rescue or salvage translocation can be used to move ecologically important habitats, such as hedges, small trees, pond vegetation and sediments, and areas of species-rich grasslands, wetlands or heathlands. The likelihood of a successful outcome and the risk of failure are significantly influenced by the translocation methodology.

Engineering and ecological skills are required in the selection and preparation of the receptor site, such that its landform and environmental characteristics match those of the donor site in terms of aspect, slope, soil characteristics (especially pH and nutrients) and hydrology.

The plant and machinery needs to be appropriate for the habitats being moved. For example, using low-ground-pressure tyres or tracked machinery to avoid soil compaction and using large buckets to maximise the length, width and thickness of turfs so that disruption to vegetation is minimised.

Habitats are best translocated in the autumn when the soils are warm and moist and new root growth is possible before winter. Translocation in spring has a greater risk of failure as the roots may not develop before the stresses of summer; while translocation in summer is very risky because the vegetation will have the greatest demand for water at a time when the supply of rainwater is lowest and the root system has been disrupted.

Following translocation, the habitats will require appropriate after-care similar to that required for newly created habitats and landscapes (e.g. cutting grasslands, trimming hedges, watering in dry weather) and monitoring to assess success and determine what, if any, remedial treatment may be required.

Translocating mature and complex habitats provides landscape structure, visual screening and habitat diversity more quickly than habitat creation using seeds or nursery-grown materials. Retaining features within a site, even in a different location, keeps their ecological functions, such as corridors for wildlife to move along and to provide connections between habitats.

In carbon terms, translocating features such as hedges within a site may be more environmentally sustainable and have a smaller overall carbon footprint than planting a replacement hedge using nursery-grown trees and shrubs together with protective rabbit guards and stock-proof fencing.

A further driver for salvage translocation is the ecosystem services provided by wildlife habitats – such as flood mitigation, noise reduction, air-quality improvement and visual screening. There is new official guidance in the UK on ecosystem services

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which will be used to support the UK cross-government public services agreement no. 28 on the natural environment. This explicitly calls for the value of the services provided by the natural environment to be reflected in decision-making (HMG, 2007; Defra, 2007). The benefits of ecosystem services, and any losses as a result of development projects, need to be included in project cost–benefit models. Retaining wildlife habitats on a site – albeit translocated to a different part of the site – can maintain these valuable services.

This paper sets out three case studies involving ground-engineering and construction projects where salvage translocation was used to relocate important habitats within each site where these habitats could not be retained in situ. Each case study includes background information about the project, the method used to translocate the habitat and the results of each translocation derived from post-translocation monitoring.

Figure 2. Moving a 1 m section of hedgerow at Lightmoor in September 2007

Case study 1: Lightmoor Urban Village, Telford

Background

Approximately 100 m of hedgerow, within the footprint of the proposed development for Homes & Communities Agency and Bournville Village Trust, was assessed as being of considerable age and species richness. The evidence for this assessment included signs of past hedge-laying (e.g. large horizontal stems) and very thick woody stems (e.g. hawthorn stems up to 150 mm in diameter at base), the presence of a 'bank and ditch' feature, and a diversity of woody species including hazel, ash, holly, field maple, common hawthorn and blackthorn with a ground flora containing bluebells.

Hazel, field maple and bluebells are associated with ancient woodlands and their presence indicates that the hedge was a remnant of an old hedgerow. This section of hedge could not be retained in situ and was translocated as part of the earthworks programme to form one side of an area of open space thus creating a wildlife corridor with a small woodland at one end and an established hedge at the other.

Pragmatic judgement was required in deciding that sections of recently planted hedge within the site should not be translocated because their value was not considered to merit the extra costs involved in translocation compared to merely clearing them from the site.

Method

Approximately 100 m of hedgerow was cut to a height of 300–500 mm at the start of 2007 to prevent birds nesting. Ash and field maple trees up to 225 mm in diameter were reduced to about 1 m in height. The translocation was undertaken in late September 2007 at the start of the earthworks programme.

A trench was dug at the receptor area immediately prior to the hedge translocation to prevent the receptor trench drying out. The base of the receptor trench was scarified and slow-release fertiliser (20:4:10 N:P:K with mycorrhizal additive) and water-retaining gel was spread along the trench.

The hedgerow was dug out in sections (approximately 1·5 m width × 1 m length) across the line of the hedge to a depth of at least 1 m using a tracked 360° excavator with the largest ditching bucket available (Figure 2). During the excavation, a chainsaw was used to free roots and branches where necessary to prevent them being torn.

Sections of hedge with thick horizontal

Figure 3. Placing a hedge section in the receptor trench, which had a slow-release fertiliser and water-retaining gel

stems were moved without severing the stems and were transported immediately to the receptor trench before the next section of hedge was excavated. These hedge sections were placed in the receptor trench in the order in which they were removed and soil used to backfill any voids and gaps (Figure 3). Subsequent watering during the autumn was undertaken in dry conditions.


Figure 4. New growth from translocated hedge and ash stump at Lightmoor in May 2008


Figure 5. Translocated hedge with protective fence and showing new growth in July 2008

Monitoring

Monitoring in May and July 2008 found abundant new growth within the translocated hedgerow and the ash tree stump which was translocated as part of the hedgerow (Figures 4 and 5), although there was evidence of die-back in holly which can be hard to establish once moved. In August 2009, there was extensive new growth on nearly all of the woody plants within the hedge (species include hazel, common hawthorn, holly, blackthorn, ash, field maple, dog rose).

The holly showed die-back in 2008 but in 2009 there was healthy re-growth with an average annual growth of 200 mm in 2009. The old hawthorns with large horizontal stems were showing severe die-back in 2009, but the younger hawthorns show healthy growth. Further monitoring will be undertaken to see if the older hawthorns show re-growth in future.

The success of the 2007 translocation resulted in another hedge translocation in late 2009 using the same methodology; this demonstrates that that the earlier translocation brought tangible benefits to the overall development.

Case study 2: British Library book depository, Boston Spa

Background

The British Library receives around three million new items every year, requiring 12 km of new shelving and the site of the British Library book depository in Boston Spa, Yorkshire required an extension to cope with future storage. An existing earth bank supporting species-rich grassland with abundant orchids (pyramidal orchid, common spotted orchid, bee orchid) was within the footprint of the proposed extension and the planning authority, Leeds City Council, required the retention of this valuable ecological feature.

The solution was to translocate the species-rich grassland to two new landscape bunds constructed using limestone spoil excavated from the foundations of the new building. The bunds are located close to other areas of species-rich grassland that are being retained on the site that are part of the Thorp Park trading estate site of ecological or geological interest, a non-statutory site of importance for nature conservation.

The retention of approximately 8000 m^3 of spoil on site saved some £250 000 in landfill costs, contributed to a reduction in lorry movements through the nearby villages and reduced the overall carbon footprint of the development.

Method

The species-rich grasslands covered an area approximately 130 m in length by 10 m wide on a steep north-east-facing slope. There were two distinct grassland communities – short open grassland covering around 900 m^2 that was typical of calcareous soils and taller grassland covering around 400 m^2 that was more characteristic of neutral soils. The receptor sites were the north-east face of the main landscape bund (the same aspect as the donor site) and the south-east face of the adjoining bund (as an additional site). The landscape bunds were designed with a surface layer of at least 1 m of limestone over the materials used to construct the bund in order to mimic ground conditions at the donor site. The bunds were graded to give 1:2 slopes and were not covered with topsoil or treated in any other way.

The translocation involved carefully excavating turfs that were 1 × 0·5 m and 300 mm deep using a tracked 360° excavator with a modified bale-cutting bucket (turf box cutter) (Figure 6). The turfs were either placed directly by the excavator onto the toe of the south-east face of the bund that was very close to the donor site (Figure 7) or placed on a flat-bed trailer for transport to the other bund where they were placed at the base of the north-east face by a telehandler fitted with a wide bucket (Figure 8).

Each turf was carefully placed to ensure a tight fit with the adjacent turfs and was pressed down by the bucket to expel air from between the turfs and the underlying substrate. Turf off-cuts and soils from the donor site were used to fill any gaps between turfs and along the four external sides of the translocated turfs. Voids between or under the turfs were not permitted because the air spaces would cause drying out of the fragile grassland root system.

Rain during the latter part of the translocation operation caused some problems with vehicle movements on site but meant that watering of the turfs immediately after translocation was not required. The translocation works took about 3 weeks to complete in late February to March 2006. Initial inspections of the translocated turfs in May and June 2006 found that key species such as cowslips and pyramidal orchids were already flowering.

The tops and slopes of the bunds that did not have translocated turfs were not topsoiled or seeded. Once the surface soils have weathered, natural colonisation and natural succession will produce orchid-rich and diverse grasslands of high nature-conservation value because of the proximity of a good seed source in the translocated turfs and the other adjacent species-rich grasslands that were retained in situ.

Monitoring

The species-rich grasslands on the site, both the translocated grasslands and other areas of species-rich grasslands are subject to a 10 year monitoring and management schedule with reports submitted to Leeds City Council as required by planning conditions. The success of the translocation and the continuing habitat management is measured against targets such as the presence and abundance of key plant species, the ratio of herb species to grass species, the sward structure and height, and the extent of bare ground.

Monitoring the translocated grasslands in June 2008 found that many of the targets had already been reached just 2 years

Figure 6. Removing 1 × 0·5 × 0·3 m turfs from the Boston Spa site with a modified bale-cutting bucket in spring 2006

Figure 7. Placing turfs on the new landscape bund at Boston Spa

Figure 8. Turfs were also transported by a flat-bed trailer and placed by telehandler

after translocation. There was very little slippage of turfs or gaps between the turfs and the species-rich grassland was flowering well and contained pyramidal orchids, common spotted orchids and cowslips (Figures 9 and 10).

Bee orchids were not recorded in the translocated turfs in June 2008; however, only one bee orchid was found in all of the species-rich grasslands elsewhere on the British Library site and their absence in the translocated turfs in 2008 may have been due to weather conditions rather than the translocation process.

The initial monitoring results provide a basis for cautious optimism about the final outcome although it is too soon to say that the vegetation on the receptor site is the same as the original vegetation of the donor site

Figure 9. Translocated grassland on the main Boston Spa landscape bund in June 2008

Figure 10. Wild flowers flourishing in the translocated grassland in June 2008

Case study 3: i54 strategic employment site, Wolverhampton

Background

i54 Wolverhampton is a 90 ha site to the south of junction 2 on the M54. Since 2002, the site has been subject to an extensive programme of preparation works by Advantage West Midlands and its joint venture partner Wolverhampton City Council, including removal of contaminated soil, earthworks to form development platforms, the construction of a site spine road and footpaths/cycleways, and the provision of drainage and landscaping. A new access from the M54 is proposed.

Engineering consultant Mouchel is working with Atkins (responsible for ecology) and Potterton Associates (responsible for landscape design) to deliver the project. Implementation of the landscape and nature conservation management plan prepared by Atkins and Potterton Associates in 2007 has resulted in ecological enhancement and mitigation involving valuable habitats and legally protected species – water voles, badgers, newts and nesting birds.

The principal green infrastructure elements include the retention of boundary hedgerows and ancient woodland together with the enhancement of existing copses, ponds and watercourses and the creation of footpaths. Habitat translocation of fen/swamp vegetation, as well as sections of hedgerow and a young oak tree, has enabled these features to be retained within the site albeit in different locations.

Translocation of fen/swamp vegetation

The fen/swamp vegetation had developed on heavy clay soils and was dominated by reed sweet grass with great willowherb and tufted hair grass, as well as soft rush, meadowsweet, lesser pond sedge and brooklime. The translocation was undertaken in September 2007 using a tracked 360° excavator with a digger bucket to take approximately 500 m^2 as turfs from the wetter areas of the fen/swamp vegetation as these had the greatest ecological value (Figure 11).

The turfs were placed at four locations within the receptor site to ‘seed’ it with wetland plants. The receptor area was a large expanse of low-lying land which had been previously shaped and compacted as a surface water attenuation area. This area receives surface water drainage from the i54 site as the final stage in a sustainable drainage system involving a series of newly created swales and ponds along a watercourse that runs through the site and discharges into the adjacent brook.

The whole of the i54 site is subject to the local landscape and nature conservation management plan, which includes annual monitoring in spring and summer with an annual report being submitted to the planning authority and other interested parties as part of planning conditions. Monitoring of the fen/swamp vegetation in 2008 and 2009 showed very successful regrowth of aquatic plants. Much aquatic vegetation appears to have developed of its own accord from the existing seed bank and plant roots in this area. However, the translocated turfs clearly stand out as areas of more established vegetation and provide structural diversity within the new wetland habitat.

The vegetation was originally classified as reed sweet grass swamp before translocation and the vegetation can still be classified as this community in 2009. This surface water attenuation area is not yet receiving surface water runoff as the development plots are yet to be constructed. As a result, a mosaic of habitats has developed, with wetter areas indicated by brooklime and drier areas with meadowsweet and great willowherb (Figure 12). Nettle, broad-leaved dock and spear thistle are also present but these species are expected to decrease as the area starts to receive surface water runoff and becomes wetter.

Mallard and lapwing have been recorded and, in 2009, three pairs of snipe and two pairs of little ringed plover nested in this area.

Translocation of hedge and oak tree

Two 120 m long sections of hedgerow were translocated in late October/early November 2006 together with a small


Figure 11. Digging up selected wetland plants at the Wolverhampton site in September 2007


Figure 12. Fen and swamp vegetation in the surface water attenuation area at Wolverhampton in 2009

oak tree about 10 m high and around 40 years old. These were moved to the western boundary of the site, where they linked to existing hedges that were strengthened with new hedge planting to create continuous wildlife corridors around the i54 site boundaries.

Both hedgerow sections were heavily coppiced before translocation. A receptor trench was excavated by a 360° tracked excavator which then moved the hedge in 2 m long sections using a toothed bucket; a power saw was used to cut roots and branches where necessary. The excavator also moved the oak tree by easing it slowly from the soil and moving the whole tree together with its root ball. The hedgerow sections and the tree were placed in the receptor trench and immediately backfilled (Figure 13). After-care comprised watering in dry periods.

The monitoring of the hedges is by one annual visit in summer using fixed-point photography and measuring growth rates. The translocated hedges showed no evidence of die-back but had abundant new growth of up to 400 mm in April 2007, some 6 months after translocation (Figure 14), and both the translocated hedges and the oak tree showed healthy new growth in 2008 and 2009.

Conclusion

Translocation is not a new technique. Individual trees have been moved since at least 1700 by wealthy landowners. Techniques were devised by landscape designers such as Capability Brown to


Figure 13. Hedge and 10 m tall oak tree immediately after translocation at the Wolverhampton site in October 2006


Figure 14. Hedge at Wolverhampton in April 2007, some 6 months after translocation

dig up mature trees while maintaining the root system and move them on specially designed machines (Mabey, 2007).

Habitat translocation can be used in a planned and designed way through the application of guidance such as the UK Highways Agency *Design Manual for Roads and* Bridges (Highways Agency, 2001) and the Ciria best-practice guide to habitat translocation (Anderson and Groutage, 2003).

The three case studies presented in this paper demonstrate that important ecological habitats can be retained during the development of a site, even if rearranged and in different locations. Habitat translocation is an effective technique that enables mature and complex ecological resources to be retained on a site or in the vicinity of a site. This maturity provides landscape structure, visual screening and habitat diversity more quickly than habitat creation using seeds or nursery materials.

The retention of a habitat within a site allows ecological functions associated with the habitat to be retained within a site – for example, the habitat connectivity and wildlife corridor provided by a hedgerow. Translocation can generate ecological resources for new habitat creation schemes – such as moving wetland vegetation from an existing pond to a new one – and ensures that native species of local provenance are used rather than imported plants.

The success or failure of habitat translocation depends on four critical factors

- matching the environmental context of the receptor site to that of the donor site
- using appropriate plant and machinery for the habitats being moved
- translocating habitats at the right time of year
- after-care and monitoring as with any newly created habitat.

There is a growing evidence base for both success and failure in habitat translocation which underpins the application of these critical factors to the particular set of circumstances on any given site. Habitat translocation has as much chance of success as habitat creation. The probability of a successful outcome can be established by reference to experience and to published case studies so that the reasons for success or failure can be identified (Anderson and Groutage, 2003; Box, 2003; Bullock, 1998).

Monitoring of habitat translocations over the long term is very important in identifying the success of both the translocation technique and subsequent management of the habitats, thus allowing remedial actions to be implemented. Furthermore, the data from such monitoring will result in greater understanding of the ecological and engineering limitations associated with habitat translocation, improved and cheaper habitat translocation methodologies, and an increase in the likelihood of success.

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Coed Darcy urban village - delivering biodiversity during redevelopment of a former oil refinery adjacent to a wetland of international importance

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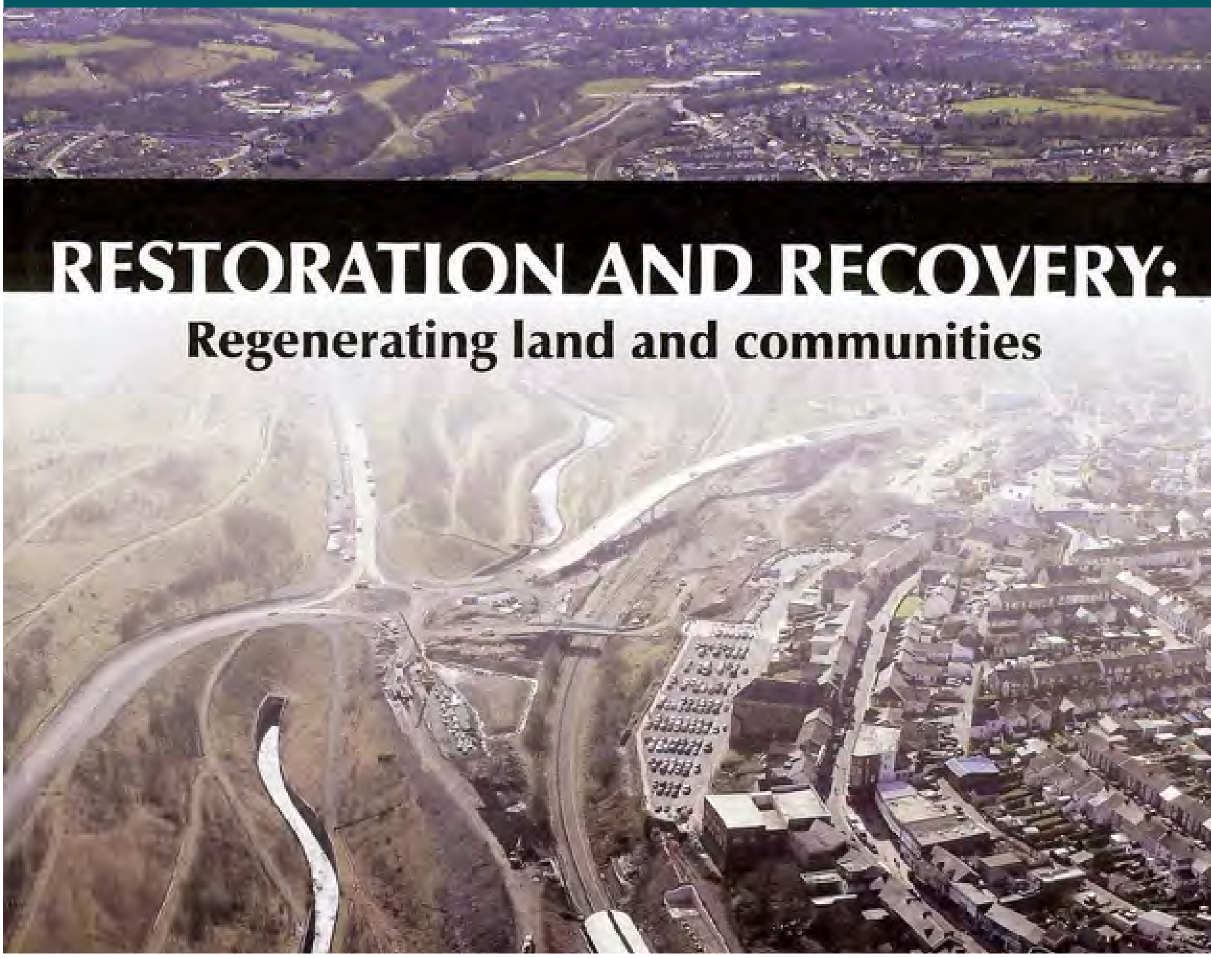

RESTORATION AND RECOVERY:
Regenerating land and communities

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INTRODUCTION

St. Modwen Developments Ltd (St Modwen) is transforming the former BP Llandarcy Oil Refinery, between Swansea and Neath in south Wales, into the Coed Darcy Urban Village. The former oil refinery opened in 1922 to process crude oil which was piped from Swansea Docks where the oil tankers unloaded. From the 1960s, the oil was piped across south west Wales from a new terminal used by the larger tankers that required the deeper waters of Milford Haven in Pembrokeshire. Peak annual production was 340,000 tonnes of petrol, diesel, kerosene and other products such as lubricating oil and wax. The oil refinery closed in 1985 and the site was formally closed in 1997 leaving the largest single brownfield development site in Wales.

Figure 1 Urban Village site from J43 on M40 with Crymlyn Bog in the distance.

Outline planning permission for the Urban Village and detailed planning permission for the main access road to the south (the Southern Access Road) were granted in February 2008. St Modwen purchased the land from BP with the aim of delivering the planned development. A Management Company was also set up comprising St. Modwen, The Prince’s Foundation for the Built Environment and the local authority, Neath Port Talbot County Borough Council.

Crymlyn Bog (Cors Crymlyn), an internationally important wetland with multiple nature conservation designations, part of which is owned by St Modwen, is immediately adjacent to the site of the Urban Village and its proximity is a prime asset to the development. The juxtaposition of the Urban Village and Crymlyn Bog provides challenges for the remediation of the refinery site and for the planning, design, construction and long-term operation of the Urban Village. Key issues for the Urban Village development are maintaining the quality and the quantity of the inflows to the Bog from the site.

REMEDIATION AND DEVELOPMENT

Remediation of the industrial residues on the 250 ha site will be followed by a phased development programme to provide 4,000 homes for around 10,000 residents with three primary schools, a secondary school and community halls, library, youth centre and doctor's surgery. In addition, there will be around 50,000 m^2 of office space and commercial development. Major infrastructure works include improvements to Junction 43 of the M4 to provide access from the north and construction of the Southern Access Road to provide a link to the south over the Tennant Canal and the railway to the existing A483 Fabian Way from Neath to Swansea. Local residents will be encouraged to enjoy the wildlife in and around the Urban Village through, for example, information packs for house buyers and an ecological liaison officer to promote the benefits of regular contact with wildlife (Rohde & Kendle, 1997; Douglas, 2008).

The remediation of the site is anticipated to take seven years to complete, with twenty years of infrastructure and building construction to follow in a phased approach. The Remedial Action Plan (RAP) works address potential contamination around the boundary of the site. The RAP works must be undertaken to the satisfaction of the Environment Agency (the statutory agency dealing with rivers, flooding and pollution). The RAP works generally relate to a series of pathway interruption measures (e.g. interception trenches). The need for reedbeds or other engineered wetlands is also to be considered. Development remediation will involve the excavation and on-site bioremediation of hydrocarbon-contaminated materials using accelerated biological techniques, and possibly more specialist groundwater techniques in localised areas of the site.

Economics and sustainability have ensured that the demolition of buildings and associated infrastructure and the removal of pipelines have produced 8,500 tonnes of steel and 220 tonnes of cables for recycling, 12,000 m^3 of concrete crushed and ready for re-use, and some 500,000 litres of oil have been recovered for recycling since the RAP programme started.

MASTERPLAN

The original Masterplan that accompanied the outline planning application submitted by BP has been revised and updated to reflect the site conditions and planning obligations. Key structural elements within the site include the sandstone rock exposures that are covered with heathland vegetation and form strategic landscape features. Woodlands give a definite edge and a buffer zone to the Urban Village where it meets the Crymlyn Bog along the western boundary. The Masterplan recognises that the open spaces and woodland in the buffer zone between the Urban Village and Crymlyn Bog must be treated with great care and foresight.

The Masterplan has been revised by Alan Baxter Associates and includes the strengthening and extension of the major open space corridor running through the length of the site. This feature will provide a crucial wildlife corridor through the Urban Village, a landscape buffer between residential areas and a woodland for residents to enjoy daily contact with nature.

The Masterplan includes the creation of wetland habitats adjacent to the Crymlyn Bog which will contain waterbodies and surrounding wet grassland habitats designed for the great crested newt population on the site that is believed to be the largest in south Wales.

CRYMLYN BOG (CORS CRYMLYN)

Crymlyn Bog is internationally important and has overlapping designations as a Special Area for Conservation (SAC), a Wetland of International Importance (a Ramsar site), a nationally important National Nature Reserve (NNR) and Site of Special Scientific Interest (SSSI) [see www.jncc.gov.uk/ProtectedSites/SACselection/sac.asp?EUCode=UK0012885; www.jncc.gov.uk/pdf/RIS/UK14006.pdf and the references given; Robertson (2000)]. Crymlyn Bog comprises a floodplain-valley mire located within a lowland coastal context and is the most extensive wetland of its type in Wales. The mire features a complex mosaic of vegetation types, supporting transition mires and quaking bogs, swamp, tall herb fen, fen meadow and wet woodland (carr) communities.

Figure 2 Crymlyn Bog.

The key issues in relation to the Crymlyn Bog are the quality and the quantity of the water inputs. The principal regulators, CCW and the Environment Agency, worked closely with BP following the formal closure of the refinery in 1997 regarding the implementation of a comprehensive risk-based strategy to the long-term amelioration of the refinery site and its redevelopment. A review of the hydrological processes at Crymlyn Bog was completed in March 1998. CCW and the Environment Agency worked jointly in 2004 on a detailed hydrological and nutrient assessment of the site to ascertain the overall nutrient loading on the mire communities (Headley, 2004). The results of these studies were used in the development of the remediation programme (particularly the RAP), the Environment Agency surface water discharge consents from the site into Crymlyn Bog, and the formal Appropriate Assessment of the implications of impacts on the SAC and the Ramsar site from the proposed Urban Village required by the Conservation (Natural Habitats, &c.) Regulations, 1994.

Surface water drainage from the Urban Village is expected to provide a high quality input to this internationally important wetland. The lagoons associated with the four surface water discharge points to Crymlyn Bog will be enhanced. A surface water drainage strategy is being developed and discharges to Crymlyn Bog will be attenuated and controlled to mimic as closely as possible the flow conditions existing when the refinery was operational. With closure of the refinery, the source of hydrocarbons has been removed and remediation of the Urban Village site will deal with contamination in the ground, thus improving the quality of the discharges to the Bog.

DELIVERY OF ECOLOGY INPUTS

As the refinery site was progressively abandoned, buddleia scrub began to colonise the concrete and brickwork areas and scrub, young birch woodland and gorse were regenerating elsewhere across the site. Clearance of these habitats from the majority of the site started in 2008 because of the need to understand the site, to undertake demolition, to locate and remove redundant pipework and buildings, to identify site hazards such as manholes and areas contaminated by hydrocarbon residues, to manage the fire risk, to start the remediation works in certain areas and to undertake topographical surveys. Habitats of importance such as the woodlands along the western boundary of the site with Crymlyn Bog, the woodland in the south east part of the site, and the heathlands on the rock exposures have been retained.

The demolition of redundant buildings required bat surveys to bring the existing survey data up to date. Demolition of the Bath House and the adjacent Medical Building in 2008, each of which contained an occasionally used roost by common pipstrelle bats, was undertaken under a Welsh Assembly Government (WAG) licence because bat species are European Protected Species. The mitigation for the loss of this roost involved the erection of four Schwegler bat boxes on nearby buildings and three Schwegler bat boxes on suitable trees subject to annual monitoring over 5 years. The first year of monitoring in 2009 showed no use of the boxes. A maternity colony of brown long-eared bats in another building will require a purpose built building in a part of the site that is both suitable for bats and which will not be disturbed.

The key ecological constraint to the remediation and construction works is the presence of a population of great crested newts, a European Protected Species, in a group of five waterbodies in the southern part of the site. An exclusion zone was established around these waterbodies at a distance of 250 m from the waterbodies in agreement with CCW whilst a full newt trapping and clearance exercise is being undertaken over 3 years from 2009 under a WAG licence. The exclusion area was divided into 27 compartments with plastic fences to prevent newts moving around (some 7 km of newt fences were erected in 2009) and newts are trapped in plastic pitfall traps or located under the squares of carpet laid as artificial refuges in each compartment.

Bottle trapping and torch surveys of these five waterbodies in 2008 identified a peak count of 15 great crested newts in the Triangular Pond and a peak count of up to 10 newts in each of the other four ponds. These five ponds are close together and there is likely to be a regular interchange of animals between them. The great crested newt metapopulation was estimated on the basis of a maximum total count of all five ponds during one visit to be at the lower end of the ‘medium’ size class (range 11–100 animals, English Nature 2001, p. 28). A population estimated from six visits using torches at night and overnight bottle traps can range between 2% and 33% of the actual great crested newt population, although it is recognised there is much uncertainty

Figure 3 Plastic exclusion fences for great crested newt trapping programme.

to these figures (English Nature, 2001, p.27). The mitigation for the loss of the five breeding ponds and associated terrestrial habitats is based on a theoretical population of around 750 great crested newts. The trapping programme in 2009 found a large number of great crested newts in the terrestrial habitats around the waterbodies.

The existing great crested newt mitigation strategy prepared in 2006 envisaged the retention of the Triangular Pond and the creation of new waterbodies. In practice, the presence of contaminated residues in the waterbodies and the land around them has precluded this approach. New waterbodies (the Coed Darcy Wetlands) will be created following remediation of land which currently contains sludge lagoons and is adjacent to Crymlyn Bog. In the interim, a dedicated newt receptor area has been created. This will be linked to the Coed Darcy Wetlands through existing woodland with additional ponds created to provide a 'newt corridor' with both breeding sites and resting places. The newt receptor area covers 13 ha and contains good quality habitats with rough grassland, scrub, woodland, bracken and bramble, ditches and waterbodies. This area will be used for the translocation of great crested newts trapped on the main site. It is estimated that the carrying capacity of this receptor area is around 2,000 great crested newts based on data given in Oldham (1994) and Redgrave (2009).

The reptile conservation and mitigation strategy that has been established for the whole of the Urban Village site is designed to be applied to each phase of development – or to an area of land prior to the start of site works. The strategy assigns areas of land with habitats that might support reptiles (common lizards, slow worms and grass snakes) into four categories of potential for reptiles: negligible (the majority of the site), low, moderate, high (e.g. heathlands, wet grassland). A specific set of mitigation measures will be followed for land within each of these categories ranging from simple habitat modification to displace reptiles on land of low reptile potential to searching, trapping and translocating reptiles on land of high reptile potential.

The clearance of much of the recently recolonised scrub and young woodland from the site resulted in bare and open ground that was potentially attractive to ground nesting birds (lapwings, skylarks). Measures to deter these birds from nesting are introduced to those areas of the site that require works. In 2009, these measures involved a network of fence posts and plastic tapes with CDs hung from the tapes. Cetti's warbler is a notable bird that breeds in the scrubby margins of reedbeds and overgrown ditches and is known to breed on Crymlyn Bog. It receives special legal protection in the UK from disturbance whilst on the nest and with dependent young. Surveys for this warbler in 2009 noted its distinctive explosive bursts of song calls but found no evidence of breeding.

The Southern Access Road (SAR) will follow the route of former pipelines that ran southwards from the refinery and will use the existing 12 m wide corridor over part of Crymlyn Bog that is not subject to nature conservation designations. The SAR will cross the Crymlyn Bog on a viaduct whose design and construction planning is being undertaken in consultation with CCW. The SAR construction will take into account protected species including bats, water voles, otters, breeding birds and badgers.

CONCLUSION

Innovation and a pragmatic approach are required to deliver biodiversity enhancements and mitigation for the Urban Village. Ecologists have been working directly on project delivery rather than their more usual role in ecological consultancy of providing advice and recommendations as part of project planning and obtaining planning consents. This means that the ecologists have to frame their advice in terms of how it can be directly implemented by their engineering colleagues and by the contractors on the site involved in remediation, infrastructure design and construction. This focus on implementation drives innovation and an approach that thrives on finding effective and economic solutions. Delivering biodiversity in such a large scale project with its demanding and phase-driven timetable is about conserving biodiversity in terms of local populations and local habitats. This means moving animals (for example, great crested newts and reptiles) to new locations, enhancing existing habitats for wildlife, translocating habitats and creating new habitats.

The key to delivering biodiversity in such a challenging environment requires ecologists to work in multi-disciplinary teams with engineers, planners and landscape architects. Just as crucial are the excellent working relationships that have been developed with the local ecologists from the Countryside Council for Wales and Neath Port Talbot County Borough Council, and the consents officers from the Welsh Assembly Government, who have provided effective and timely advice and a sound process for the necessary consents and licences.

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Liam Atherton, Stuart Smith and Brian Cox

How much water do rivers need? Hydroecology and environmental flows

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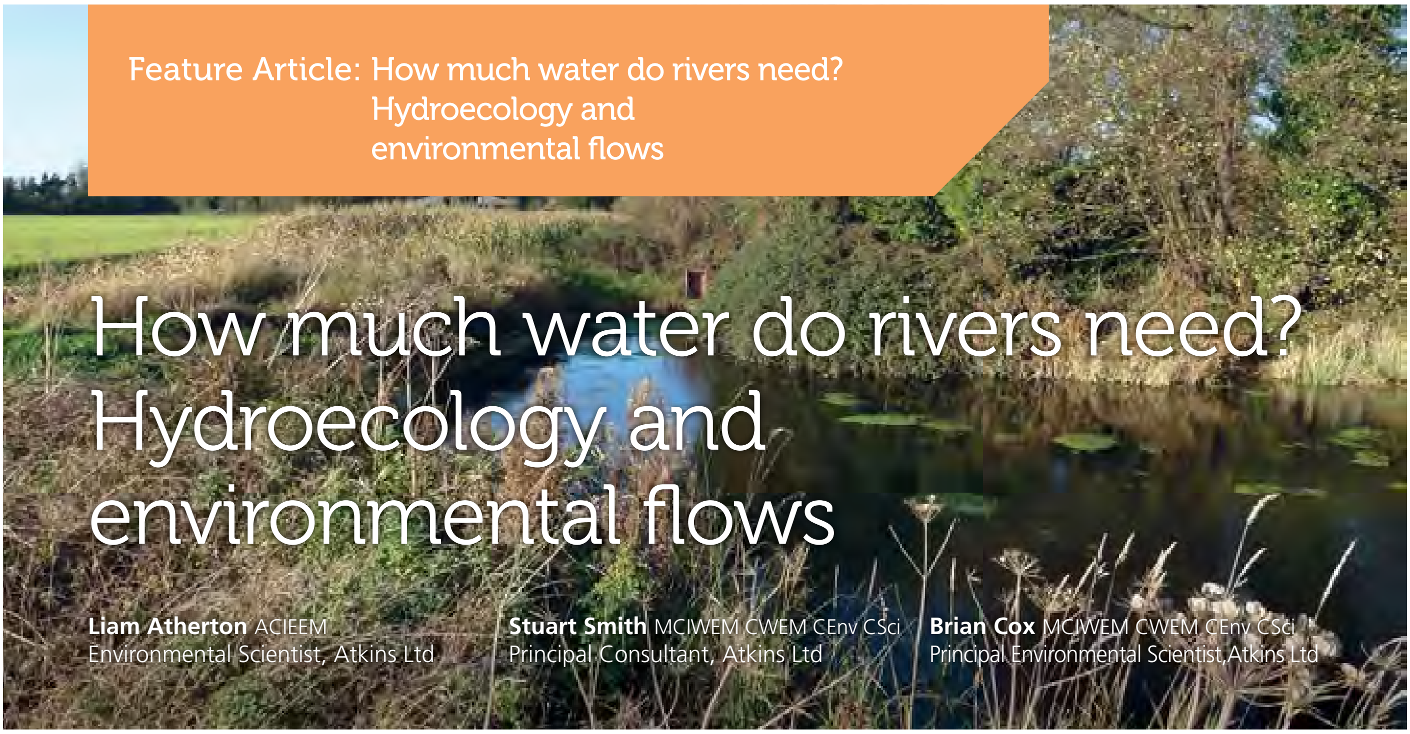

How much water do rivers need? Hydroecology and environmental flows

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Current public perception of water availability in the UK has been heavily influenced by recent media coverage of winter flooding. However the fact remains that there is great spatial and temporal variation in water availability. Water stress in the Thames Valley for example is comparable to that of Syria, Spain and Australia (Gassert *et al*. 2013). The effects of climate change and population growth are predicted to place even greater demand on water resources, setting our future requirements on a collision course with ecosystem services and biodiversity conservation. Ecologists and environmental managers need to be capable of supporting water resource management by ensuring activities such as public water supply are not environmentally damaging or ecologically degrading. Environmental flows has become a widely-used term to describe the quantity, timing, duration, frequency and quality of flow to sustain freshwater, estuarine and near-shore ecosystems; also the human livelihoods and wellbeing that depend on them (Acreman and Ferguson 2010). The concept of environmental flows provides a structure to improve the protection of river ecosystems whilst allowing more sustainable development of water resources. Here we introduce the concept of environmental flows and present a recent case study for hydroecological management in practice.

Introduction

Rapid industrialisation, population expansion and urbanisation in the 18th and 19th Centuries turned many European rivers into open sewers. By the early 20th Century, the freshwater arteries of our landscape had become severely degraded, with iconic British species such as Atlantic salmon *Salmo salar* functionally extinct from many of their natural catchments in the UK.

Not surprisingly, river ecosystem management in the UK historically sought to address water quality issues first and foremost, with early drivers being largely anthropogenic (i.e. public health related). Over many years, UK research, legislation and river monitoring has been shaped towards tackling water quality pressures, with great success. This has resulted in widespread improvements in water quality and the return of a host of freshwater biota to some of our most impoverished riverine habitats (Vaughan and Ormerod 2012). Yet, whilst UK river water quality has been improving, the inaugural 2013 State of Nature Report estimated that 57% of freshwater and wetland species in the UK have declined over the last 50 years (SON 2013). If river water quality is improving, why are the species that depend on rivers in decline?

The reality is that rivers are subject to a complex mix of pressures; in the broadest terms these include physical habitat, flow regime and water quality modification (Dunbar and Acreman 2001), as well as invasive species. Recent improvements in water quality mean the limiting role of physical structure and hydrological regime has become increasingly apparent (Vaughan *et al*. 2009).

With the Office for National Statistics predicting UK population growth of up to 35% by 2050 (pop. 85 million) (ONS 2013), the demand for water resources will continue to grow. Current projections of UK climate change are also predicting regional reductions in summer precipitation of up to 28%, and winter increases of up to 23% (as central probability estimates) (UKCP 2012). Against this backdrop it is essential that we answer a seemingly simple question: how much water do rivers need?

Ecological impacts of artificial flow regulation

Abstractions and discharges are necessary to sustain vital services such as Public Water Supply (PWS), agricultural irrigation, (Left) Figure 2. River Lark, Suffolk. © Atkins Ltd.

waste removal and hydro-electric power generation. However, alteration of the natural flow regime can elicit various physical, physicochemical, quality and thermal property impacts within the riverine environment. There is significant potential for these impacts to cause undesirable effects within aquatic (and associated) ecological communities:

- reduced flows and levels exacerbate the impact of barriers such as weirs to the passage of migratory species;
- increased sedimentation rates associated with lower water velocities affect sediment-sensitive communities and fish spawning;
- reduced connectivity with natural floodplains can result in the loss of fish nursery areas and wetland communities;
- changes in erosion and deposition patterns affect morphological diversity;
- reduced aeration through mixing lowers dissolved oxygen levels;
- reduced dilution exacerbates water quality issues;
- changes in thermal gradients can increase physiological stress, and increase susceptibility to disease.

Legislative responsibility

The European Water Framework Directive (WFD; 2000/60/EC) (European Commission 2000) has established objectives for the 'ecological status' of all waterbodies across EU Member States which need to be achieved by 2027. In England and Wales only 26% of river waterbodies currently achieve these objectives (EA 2013). In Scotland, only 56% of river waterbodies are meeting their objectives (SEPA 2013).

Three elements define the ecological status of river waterbodies (see UKTAG 2009), all of which may be affected by alteration of the natural flow regime:

- biological quality elements (benthic invertebrates, fish, phytobenthos and macrophyte communities);
- general chemical and physicochemical quality elements (e.g. dissolved oxygen, ammonia and phosphate); and
- hydromorphological quality elements (e.g. morphology and hydrology).

Hydrology is assessed for all river waterbodies and therefore underpins

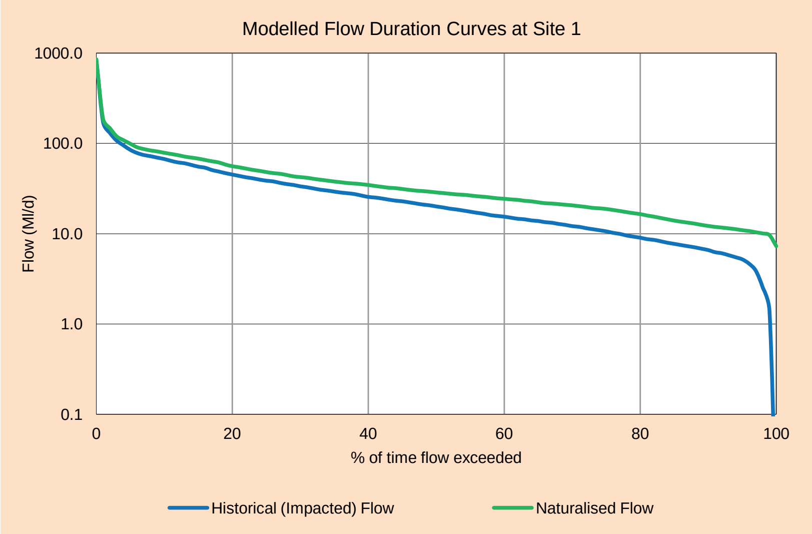


Figure 1. Example Flow Duration Curves showing the Naturalised flow regime for the site, and the Historical flow regime impacted by abstraction activities.

ecological status. Recently, over 650 river waterbodies in England and Wales were reported as having hydrology that did not support the minimum legislative requirement of 'good' ecological status (EA 2012). Taking account of the role of hydrology for 'high' ecological status (near-pristine status), it represents a limiting factor in approximately 60% of all river waterbodies in England and Wales (EA 2012).

Although the WFD provides the main structure for water management in the UK, there are further legislative considerations. The Natural Environment and Rural Communities Act (2006) requires public bodies to have regard for biodiversity conservation when carrying out their functions. The UK is also committed to halting overall loss of biodiversity by 2020; commitments intrinsically linked to the appropriate management of our freshwater habitats and, by extension, their hydrological regimes.

Current applications: the Environment Agency In-River Needs Programme

Atkins is supporting the Environment Agency in its Anglian Region to assess river environmental flow requirements through the In-River Needs Programme. The programme investigates river waterbodies where water resource activities (such as public water abstraction and discharges) have altered the natural flow regime to such a degree that it no longer supports a healthy riverine ecosystem (i.e. good ecological status under the WFD).

The initial flow impact screening test for river waterbodies is undertaken using Environmental Flow Indicators (EFIs). The EFIs represent generic river flows required to support good ecological status, expressed as a percentage of allowable flow reduction (i.e. net water abstraction) from the 'naturalised' scenario for the river under investigation. They have been developed through expert consensus (Acreman *et al*. 2008) and vary according to the abstraction sensitivity of the ecological communities present. The allowable impact is expressed at key points on a Flow Duration Curve (FDC) as presented in Table 1. The FDC is a plot used by hydrologists to summarise the hydrological characteristics of a river, showing the percentage of time that a given flow is exceeded (see Figure 1).

By way of example, at Q_{95} (the flow exceeded 95% of the time) on the 'most sensitive' rivers, the permissible abstraction impact is up to 10% (i.e. flows must be maintained at 90% of the natural flow). If the abstraction impact is greater than this, the waterbody is prioritised for further investigation under the In-River Needs Programme. To date, 77 waterbodies have been investigated.

Feature Article: How much water do rivers need? Hydroecology and environmental flows (contd)

Table 1.Environmental Flow Indicator (EFI) thresholds for permissible abstraction

Abstraction Sensitivity	Flow Exceedance Value (Point on Flow Duration curve)			
	Q_{30}	Q_{50}	Q_{70}	Q_{95}
	Permissible Impact on Naturalised Flow (%)			
Most sensitive	24	20	15	10
Moderate	26	24	20	15
Least sensitive	30	26	24	20

Case Study: River Lark Catchment

The Environment Agency identified the River Lark and a number of its tributaries as failing to achieve the EFIs. The River Lark is a chalk stream rising 1.5 km to the south of the village of Whepstead in Suffolk (Figure 2). It flows for approximately 47 km, through Bury St Edmunds before discharging to the River Great Ouse. The upper reaches are naturally ephemeral, with lower reaches underlain by unconfined chalk which supports river flows. Over 7,000,000 litres (7 Ml) of water are abstracted every day through groundwater and surface water abstractions on the Lark catchment alone, mainly for public water supply. The challenge is to define acceptable abstraction levels that sustain a healthy ecosystem. The steps taken are outlined below.

Step 1: Confirm hydrological impact on ecology

The Environment Agency's Regional Groundwater model was validated by reviewing available hydrometric (gauged) flow and level data within the catchment, ensuring it was accurately representing impacts across the catchment.

Fish and aquatic macroinvertebrate monitoring data were analysed to ascertain whether hydrological stress was apparent. At this stage, detailed water quality and morphological characterisation were integrated in the assessment, ensuring that habitat limitations and water quality impacts were not falsely attributed to hydrological stress. The flow-sensitive biotic metric, Lotic-invertebrate Index for Flow Evaluation (LIFE; Extence *et al*. 1999) and the sediment-sensitive metric, Proportion of Sediment-sensitive Invertebrates (PSI; Extence *et al*. 2013), were central to this assessment and water quality-sensitive metrics were also considered.

Step 2: Define bespoke ecological targets (environmental flows)

Where riverine communities were identified as suffering from hydrological stress due to water abstraction, the next step was to derive more ecologically meaningful targets. Three distinct methods were employed, making the overall assessment more robust:

- **Macroinvertebrate Flow Regression** (MFR) regresses the LIFE biotic metric against a range of antecedent flows (e.g. the annual Q_{95}) prior to each macroinvertebrate sample over the monitoring period (Figure 3). If a strong statistical response is established between antecedent flow and the resultant LIFE score, the trend is used to identify flows that should sustain the target community.
- **Rapid Hydro-Ecological Flow Thresholds** (R-HEFT, an example of hydraulic habitat modelling) involves topographic and hydrometric surveys at flow-responsive locations suitable for sensitive species such as brown trout *Salmo trutta* (Figure 4). This establishes the relationship between river flow, and the provision of depth and velocity targets for key species given the channel characteristics.
- **Ephemeral analysis** takes flow time-series data from the regional groundwater model along the waterbody length, defining the natural extent of ephemeral reaches. By developing a baseline 'heat map' of ephemeral distribution, it provides a functional target in terms of restoring the natural extent of ephemeral and perennial reaches.

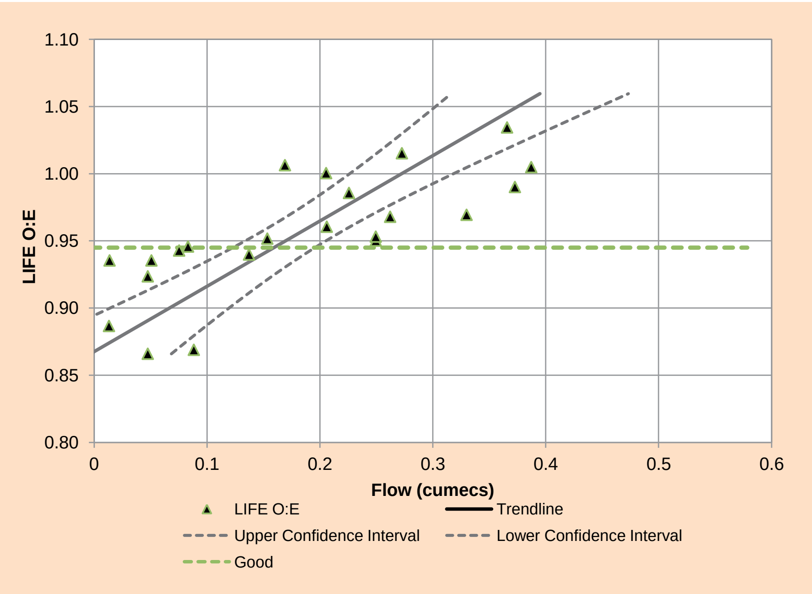
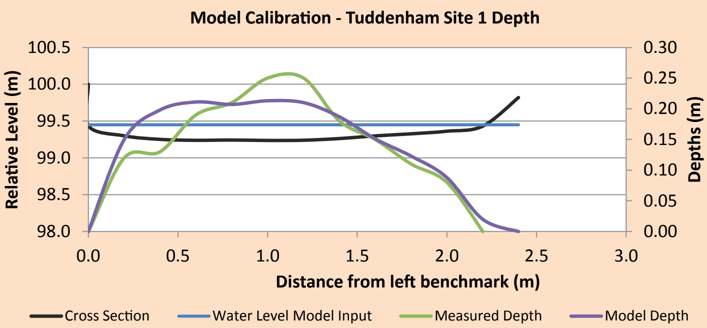
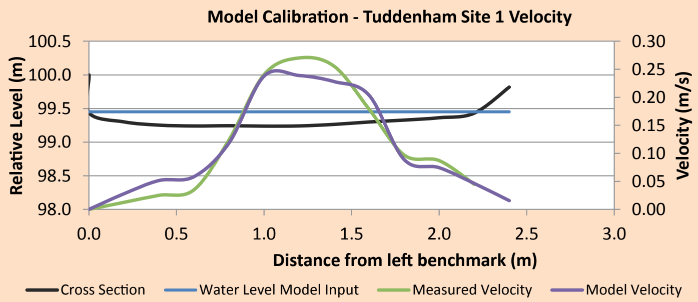


Figure 3. Example of MFR analysis output
Note: the intersect of the target and trendline help define the flows required to sustain the target macroinvertebrate community. In this example tributary site, a minimum flow of approximately 0.15 cumecs is likely to support the target macroinvertebrate community.





Flow Data Calibration

Statistic	Measured	Simulated	Difference
Flow (cumecs)	0.05	0.05	-5%
Mean Depth (m)	0.14	0.15	5%
Median Depth (m)	0.14	0.17	17%
Max Depth (m)	0.25	0.21	-17%
Mean Velocity (m/s)	0.11	0.12	1%
Median Velocity (m/s)	0.09	0.08	-6%
Max Velocity (m/s)	0.28	0.24	-16%
Wetted Width (m)	2.20	2.2	0%
Wetted Perimeter (m)	2.28	2.47	8%
Flow Area (m^2)	0.33	0.35	5%
Hydraulic Radius	0.15	0.14	-2%
25th Percentile Depth (m)	0.20	0.21	6%
25th Percentile Velocity (m/s)	0.21	0.21	-2%
Overall Difference	-	-	**-1%**

Comments

Historically re-sectioned, embanked and straightened reach. However, straightened reach has led to a steep gradient with high velocities and showing signs of natural recovery (narrowing).

Figure 4. Example of R-HEFT model calibration for a tributary in the Lark catchment
Note: once calibrated (shown above), R-HEFT models changes in habitat provision (e.g. depth and velocity requirements for sensitive species), based on predicted flows under different water resources options (e.g. varying abstraction through the catchment) to identify optimal provision.

Step 3: Appraise options for water resource management

Derived environmental flow targets were set as benchmarks, against which catchment options for water resource management were assessed. Options included changing the abstraction regime (e.g. relocating abstraction points or reducing abstraction volumes) and providing river flow support (e.g. supply from groundwater). By producing a range of bespoke flow targets across the catchment, the relative advantages and disadvantages of the options could be assessed with confidence. A key observation was that sites within the Lark catchment with very similar natural hydrological regimes exhibited significantly different flow requirements to restore target ecological communities. This provides an excellent example of the limiting role of other factors (in this case, habitat structure), and again highlights the importance of an integrated approach to water resource assessments. Consequently, the investigation was able to identify the benefits of holistic solutions for the Lark catchment including hydrological mitigation, but also revised maintenance and river restoration.

Discussion

Our knowledge of the environmental requirements of rivers will most likely remain incomplete for the foreseeable future (Dunbar and Acreman 2001). Objectives may be set too low, resulting in damage to a river, or too high, resulting in potential waste of resources, or the exploitation of other sensitive water resources. Nonetheless, we must meet legislative requirements and, more fundamentally, ensure water resource management supports functioning aquatic ecosystems. To achieve this we are obliged to apply the best and most proportionate methods available to define environmental flows.

Bespoke hydroecological monitoring programmes and detailed hydraulic modelling are costly and time-consuming; reliance on new programmes is not feasible at a national level. The In-River Needs Programme in Anglian Region provides a blueprint for utilising available monitoring and modelling data, augmented with focussed site-specific data collection. However, there remains an urgent need for more formal integration of multiple pressures (morphology, water quality, etc.) in hydroecological models that utilise existing monitoring data.

Pressures on the aquatic environment are interdependent (for example, flow reduction can lead to deterioration in water quality, resulting in the misdiagnosis of a water quality pressure). Considering any single pressure (e.g. flow) in isolation from the wider factors (e.g. channel dimensions, sediment loading, water quality, etc.) will inevitably be misleading.

Atkins is actively developing multiple linear regression approaches to help overcome some of this uncertainty. Together with models under development by the Environment Agency, such as Dried-Up With Incremental Droughts (DRUWID; M. Dunbar, pers. comm., February 2014), we can be optimistic that hydroecological assessments will continue to improve.

However there remain some important underlying issues, such as questions over the appropriateness of WFD objectives (cf. Moss 2008) and also our concept of "natural". Where we have fundamentally altered the physical character of a river and its floodplains, the flow regime will never be truly natural, even in the absence of abstractions and discharges. What regime should we seek to restore? Should we make sacrifices in some areas

Feature Article: How much water do rivers need? Hydroecology and environmental flows (contd)

for the concentration of resources in others? How much weight should be given to environmental protection in the face of water shortage? The science of environmental flows goes some way to telling us how much water riverine ecology needs, but the drivers behind their implementation remain rooted in what society values. In the end, environmental flows are socio-economic tools as much as they are science-based. It may be that the optimisation of anthropogenic benefits through river ecosystem protection (or ecosystem services) is ultimately where the balance is struck.

Acknowledgements

The Anglian In-River Needs Programme is financed by the Environment Agency. The approach has been shaped by numerous staff from the Environment Agency, Atkins and Amec. The authors would like to thank Ian Morrissey and Andy Gill (both Atkins Ltd) for their comments on a draft of this paper.

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The Olympic Park - A Biodiversity Action Plan in Action

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The Olympic Park – A Biodiversity Action Plan in Action

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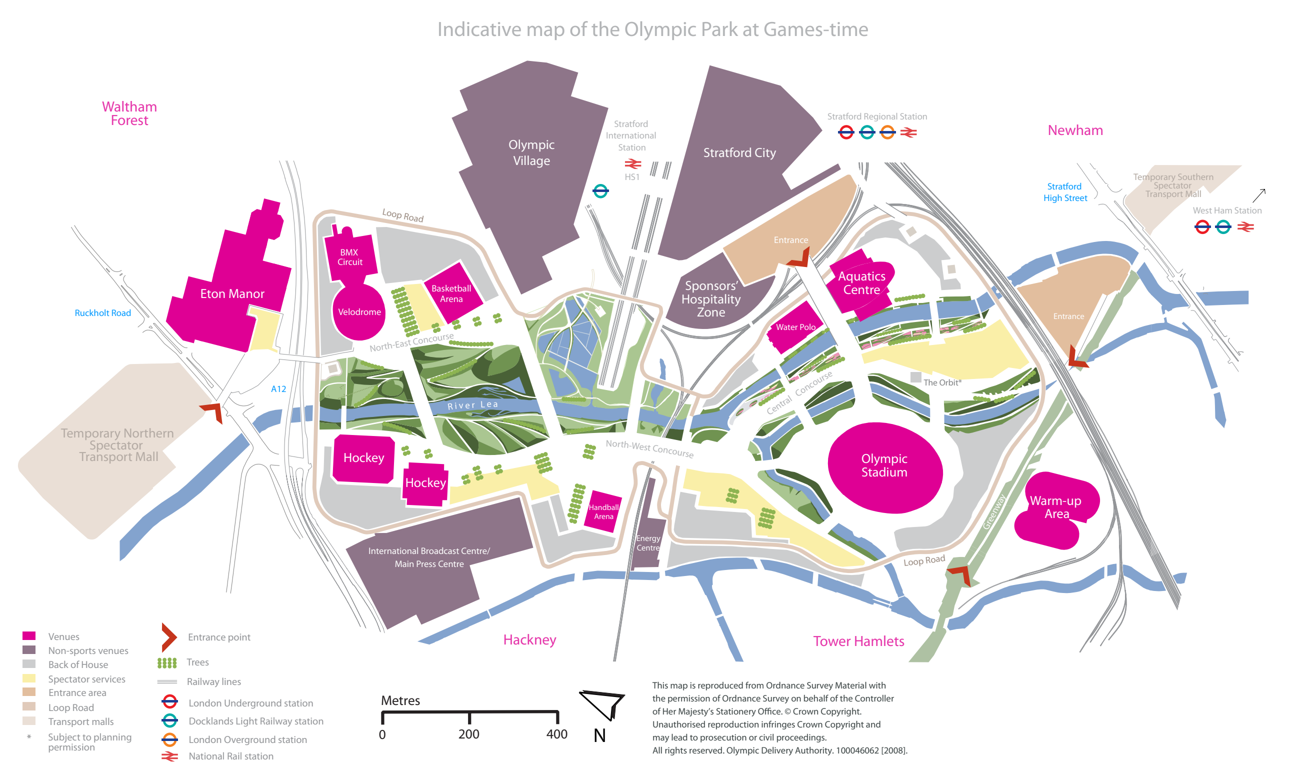
The London 2012 Games are styled as 'the green games', aiming to be the most sustainable Games ever held. The preparation of the Olympic Park site included clearance of large areas of industrial, and often contaminated, land across 246 hectares in East London. When working on such a vast project, with a non-negotiable deadline, there could always have been a risk that the promises made when planning permission was granted could have gradually fallen by the wayside as site clearance and construction progressed. This article explores the history of the project to date, and shows how the production of a scheme-specific Biodiversity Action Plan (BAP) has helped ensure that the opportunities provided by the transformation of the site were not missed.

Before site clearance began, the area that is being transformed into the Olympic Park was dominated by industrial development. Despite this, it was not an area devoid of biodiversity; a series of inter-linked waterways flow through the Park, eventually joining the Thames to the south, while patches of wetland, scrub, trees and brownfield habitat formed a series of wildlife 'islands' surrounded by the urban landscape. Whilst the intrinsic biodiversity of each patch may not have been of particular note in a different context, their urban setting increased their effective value. Some of the areas were covered by non-statutory designations, including water courses within two Sites of Metropolitan Importance, and wetland, woodland and wasteland within four Sites of Borough Importance (Grade I) and one Site of Borough Importance (Grade II).

The site spreads across the boundaries of four London Boroughs: Hackney, Newham, Tower Hamlets and Waltham Forest. In 2006 a new area-specific planning authority was created, the Olympic Delivery Authority (ODA). This body was created under the London Olympic Games and Paralympic Games Act 2006 and works in close consultation with the Local Planning Authorities and statutory and non-statutory consultees. The outline planning application for the Olympic Park was submitted in 2007.

It is difficult to envisage the Environmental Statement that would be required to assess the impacts of redevelopment across almost 250 hectares of land. In order to provide a meaningful assessment, the Park was divided into a series of fifteen 'Delivery Zones' so that baselines and impacts could be described for a series of individual areas, and then viewed cumulatively. Permission was granted in 2008. However, at that stage the detailed design of individual buildings, bridges, roads and parkland areas had not been prepared. The planning permission was therefore granted subject to numerous conditions, and the planning decision notice



Olympic Park Map ***Image: London 2012***

itself totals 168 pages in length. The conditions included one specifically for biodiversity:

Biodiversity Action Plan	
OD.0.11	Before 30 September 2008, the Biodiversity Action Plan, which shall be based on the Biodiversity Action Plan Framework submitted with the application, shall be submitted to the Local Planning Authority for approval. This shall clearly identify the areas of recognised wildlife habitat to be provided and the means by which these will be maintained.
Reason:	To help achieve biodiversity objectives and protect habitats and species.

The production of a scheme-specific BAP may not be justified in many small development projects, but for the Olympic Park the BAP was critical to ensure commitments made in the Environmental Statement are delivered on the ground. As detailed design progressed, the BAP, and supplementary guidance, provided clear targets, such as a specific number of bird and bat boxes to be incorporated into the design of individual bridges.

During site clearance, the requirements of the BAP meant that measures to protect wildlife were not limited to the familiar 'headline' legally protected species. For example:

- Four thousand smooth newts were translocated to new and existing ponds off site.
- Where existing habitat could be retained, clearly labelled fencing was used to demarcate protected areas. These protected areas were identified on a map as a requirement of a planning condition, so works could not impinge on them unless the ODA had given prior approval.
- At the start of site clearance a log wall was created in the north of the Park, helping to provide some temporal connectivity for the invertebrates displaced during the subsequent habitat clearance elsewhere across the site. These invertebrates should be able to spread again to colonise newly created habitat as the site matures.
- Experimental translocation of brownfield habitat was also undertaken at Thornton's Field railway sidings. Pre-clearance site surveys revealed a variety of invertebrates, such as the toadflax brocade moth. Ballast, soil and timber sleepers were carefully moved from this area to a part of the Park where they would not be disturbed.
- Across the site, a 'permit to clear' system was used, so that no area of habitat would be removed until site ecologists had confirmed in writing that any issues had been resolved.

The BAP set out commitments that had to be met as the detailed design of parkland areas evolved. It also ensured that key works were programmed in correctly, so that opportunities were not missed. One example is provided by the planting trials on the

New brownfield habitat provided by log wall
Photo: Atkins Ltd

The Olympic Park ***Photo: London 2012***

challenging riverside areas, described by Ian Morrissey of Atkins, London 2012's official engineering design provider, in another article. Among the terrestrial habitats, seed collection and sapling translocation were undertaken. The areas of wetland being created should be suitable for reintroduction of water voles once the habitats are well established; to prepare for this, mink monitoring is already being undertaken. As the range of otters continues to expand, occasional records are made in the London area. In order to help ensure that they can colonise the Lower Lea valley if and when they spread that far, two artificial holts are being installed in the north of the Park.

Overall, an impressive list of habitats will be created. The Park is one of the largest to be created in Europe for over a century with large areas, totalling 45 hectares, planned to support BAP priority habitats. The south park will be dominated by more ornamental planting, while the north park will have more 'natural' native planting, incorporating habitats that would once have been widespread in the Lower Thames Basin. The Park will include 10 hectares of native trees and shrubs, nearly two hectares of reedbeds and ponds and over 20 hectares of species-rich grassland. Room has been found for brownfield habitats, such as the extensive log walls created at East Marsh, resisting the temptation to only create 'tidy-looking' habitats. This habitat creation will be supplemented by other activities, such as the creation of living roofs on some buildings and the installation of 675 bird and bat boxes to provide roosting and nesting opportunities as the vegetation matures.

The success of these measures is not taken for granted. A BAP monitoring programme is ongoing during construction and the Games time phase of the site. After the London 2012 Olympic and Paralympic Games, the 'transformation' phase will convert the Park into the 'Legacy' site. Monitoring will then continue under a 10-year Maintenance and Management Plan for the Parkland and Public Realm. The overall target of the BAP for habitat creation is 45 hectares of BAP priority habitat. This will be monitored, with management adapted if needed. The aim is to reach a point where the habitats' quality is such that the 45 hectares is worthy of designation as at least a site of Borough Importance (Grade I) within the Greater London designation system.

In conclusion, while the client's and contractors' commitment to following biodiversity guidance is essential to the success of any project, another important factor is the mundane process of getting the paperwork right. On a large and complex scheme like the Olympic Park, a site-specific Biodiversity Action Plan can provide a consistent point of reference to guide design and site works. In this case, the production of the BAP has helped to ensure that the project will create an end product that provides genuine biodiversity benefits.

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Ian Morrissey and Mike Vaughan

Delivering Wetland Biodiversity in the London 2012 Olympic Park

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Delivering Wetland Biodiversity in the London 2012 Olympic Park

Ian Morrissey MIEEM and Mike Vaughan
Atkins Ltd

Appointed as the river edge engineers for the London 2012 Olympic and Paralympic Games, Atkins is responsible for the design of the wetland and river edge habitat enhancement works in the Olympic Park. Working alongside the Park's landscape architects (LDA – Hargreaves), and with the Olympic Park Biodiversity Action Plan (BAP) (ODA 2008) document providing the reference for biodiversity requirements, 2.3 ha of wetland habitat have been designed to provide ecological gains inside the Park's waterways. Although the wetland habitats equate to less than five percent of the overall 45 ha of BAP habitat to be created in the Park's landscape, when completed, they will form an area the size of three football pitches containing 40 wetland plant species and a total of approximately 300,000 wetland plants, making it one of Europe's largest urban river edge habitat enhancement programmes to date.

Located in East London's Lower Lea Valley, the River Lee and its associated waterways form an arterial route through the Park and as such will provide a significant focus during both the Games and following transformation into the legacy period. The landscape architect's vision to bring the once neglected river systems into the Park Landscape by opening up the river corridor and reducing bank slope angles will significantly improve access to the public, connecting visitors with the various wetland elements including reedbeds, wet woodlands, amphibian ponds and ecologically enhanced river edges.

Extents

From a wetland habitat diversity perspective the Olympic Park can be broadly divided in two. To the north lies the ecological park, containing the majority of the wetland features alongside the River Lee, and in the south, around the stadium island, there are smaller ecological gains for the Waterworks River and Bow Back canal systems. The focus in the North Park will be the wetland bowl area containing two online reedbeds in direct hydrological connectivity to the river. These will total over 5,000 m^2 and be composed mainly of common reed *Phragmites australis*. An additional reedbed to the north of the bowl will provide an extra 550 m^2 of this, a UK BAP priority habitat known to support an abundance of insect, amphibian and bird life. Within the reedbeds, wetland channels have been designed to increase habitat complexity, maximise reed edge extent and provide refuge for a range of fish species including eel *Anguilla anguilla*, a London 2012 BAP priority species.

'Soft' bio-engineered banks (2.5 km in total), planted with wetland species will maintain longitudinal ecological connectivity through the North Park, connecting the larger wetland features including two wet woodland habitats and a number of amphibian ponds. Two new wet woodlands, totalling 0.4 ha will provide off-river habitat, with excavated channels to maintain hydrological and ecological connectivity with the River Lee. These areas were designed to retain waters from the Lee during periods of impoundment as a result of the new water level control structures at Three Mills Island. The wet woodland habitats are being planted with a mix of shade tolerant sedge species and trees, with birch, alder and black poplar on raised features within the landform. Shallow depressions were also designed into the wet woodlands to hold standing water and provide a range of moisture gradients across the habitat. Marginal wetland flowering plants are being used in such areas adding species richness to the habitats.

Wetland planting is also being provided for three new amphibian ponds, summing 0.2 ha. All of the ponds are fed by drainage waters from the Park's concourse, with the largest having been designed with an adjustable feed from the River Lee to allow maintenance of a permanent water level. The maintained water body will be planted with a range of plants including oxygenating submerged aquatics *e.g.* rigid hornwort *Ceratophyllum demersum* and species such as water forget-me-not *Myosotis scorpioides* to provide suitable egg laying sites for newts. A series of log walls installed alongside the ponds will increase ecological value through the provision of habitat for invertebrates and hibernation sites for amphibians.

Planting Trial

In order to inform the wetland designs and reduce the risk of wetland plant failures in the final design an on-site riverside

Figure 1. View of planting trial following installation in 2008 **Photo: Atkins Ltd**

planting trial was undertaken to investigate and advise on wetland plant species selection, plant installation techniques, species specific planting elevations and potential environmental constraints to wetland plant establishment and performance. Of particular concern to the designers were the effects of changing river levels arising from the impoundment of the River Lee, to the south of the Park. These flow control structures have changed the river regime from an inter-tidal habitat to a freshwater fluvial system experiencing a twice daily rise and fall in river level of typically 400 mm as the Lee becomes tide-locked at Three Mills Island. This was identified as presenting a potential constraint to the successful establishment of the wetland plants along the river edge as well as having implications for the range of species that were suitable for installation and hence overall wetland habitat biodiversity.

The planting trial was established in the north of the Olympic Park in September 2008 along a 50 m length of river bank extending from mean low water level (2.4 m AOD) to a top bank level of approximately 4.2 m AOD, an elevation of 1.8 m. The trial platform was divided into eight separate treatment areas across which a total of 15 species of wetland plant were trialled over a period of 12 months (see Figure 1). Of these, seven species were key to our conceptual designs for the river edges and large wetland features: common reed *Phragmites australis*, purple loosestrife *Lythrum salicaria*, reed canary-grass *Phalaris arundinacea*, lesser pond sedge *Carex acutiformis*, common club-rush *Schoenoplectus (Scirpus) lacustris*, reed sweet-grass *Glyceria maxima* and yellow iris *Iris pseudacorus*.

The relative success of the different planting techniques and approaches was investigated through the assessment of the performance of wetland plants installed on the river bank as either pre-established coir pallets, (plants established in a 2x1x0.1 m coir coconut fibre mattress pegged on to the planting platform, see Figure 2) or as plug plants (plants grown in root trainers and plugged directly into the river bank substrate). Plants were installed at a standard density of 20 m^{-2} across the trial area to ensure that comparative analysis between treatments could be undertaken. The influence of bank form was also qualified with the plants being installed on both sloped (1 in 2.5) and terraced platforms. But perhaps most importantly, the influence of the frequency and duration of inundation by the river was examined. This was achieved by running the planting trial in conjunction with a water level monitoring programme which, when combined with planting trial area elevations, allowed the calculation of plant inundation depths and frequency of inundation in response to fluvial flow variability and the effects of tidal fluctuations. The performance of the plants was assessed through a combination of biometric monitoring undertaken seasonally and by monthly repeat fixed-point photography to assess plant establishment progress. Biometric monitoring included the calculation of percentage cover and the measurement of average height achieved by the installed species at different elevations to determine establishment success.

Figure 2. Pre-established coir pallet ***Photo: Atkins Ltd***

Moving Forward

In response to the trial findings, poor performing and failing species were removed from the final planting arrangements and replaced with those plants shown to have established successfully over the year-long monitoring period. The trial results were used to make informed decisions on the appropriate topographies across the larger wetland features and the need for fine adjustments to be made to the final planting arrangements. This took the form of relocating certain species to more appropriate river bank levels *e.g.* movement of reed canary grass to higher bank elevations.

The trial successfully validated our approach of using pre-established coir pallets as the preferred planting installation technique to be adopted along the river banks and larger wetland features. This was demonstrated by the greater success of plants when installed as pre-established coir pallets compared to the same species trialled as plug plants. For example, percentage cover and height achieved over the trial period by reed sweet-grass installed as pre-establish coir pallets was 75% and 425 mm respectively, compared to 2.5% and 250 mm for the same species when installed as a plug plant. The trial also provided some very useful pointers as to the potential effects that land contamination could have on the final planting arrangements with leachate causing plant failures in certain trial areas. This highlighted the value of the remediation programmed to be undertaken after the trial by the Enabling Works contractor along the river banks. The negative effects of wildfowl grazing on plant establishment were also observed at trial resulting in the incorporation of temporary wildfowl fencing into the landscaping programme to exclude bird grazing pressure up to the Games period.

The planting trial proved a vital tool in allowing us, as designers, to evolve and finalise the approach in the delivery of wetland biodiversity in the Olympic Park. As described in my colleague's (Claire Wansbury) article, the eventual aim is for the quality of the individual wetland habitat elements to meet appropriate criteria for their designation as a site of Borough Importance. The ethos of maximising opportunities and designing for wider ecology, coupled with the commitment to monitoring and undertaking appropriate management, will go a long way to achieving this goal. The environments created will provide a significant betterment to the habitat diversity and ecological connectivity of the waterways in the Lower Lea Valley, transforming a once neglected river corridor characterised by high levels of contamination and a prevalence of invasive species, into an area containing 2.3 ha of new wetland Olympic Park BAP habitat.

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Riverfront Landscape Design for London 2012 Olympic Park

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图01 奥林匹克公园和湿地鸟瞰图
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伦敦2012奥林匹克公园滨水景观设计与营造
Riverfront Landscape Design for London 2012 Olympic Park

如果在不久的未来，将有数百万人途经这块滨水区域，沿河该如何种植？如果了解到自然界中人造湿林地已十分罕见，该如何设计、创造并维护周边这种湿地环境？该如何长期保持栖息地的活力和可持续性？在伦敦2012奥林匹克公园项目中，来自阿特金斯的工程师们受托负责湿地和河滨地区设计及建设，将会找到所有这些问题的答案。

为2012年伦敦奥运会和残奥会而新建的奥林匹克公园是由一片废弃工业用地改造而成，占地超过246hm²，是欧洲最大的城市景观项目之一。东伦敦下利亚山谷的河流和湿地形成新的风景将是整个新公园的视觉中心，并为2012年伦敦奥运会平添景色。

阿特金斯负责河岸和湿地的软化设计，以及河岸修复和生物工程。从项目的概念设计到施工建成，阿特金斯的工程师们发挥了重要作用。该项目不仅仅修复了超过8km长的河岸，同时还修建了2hm²的芦苇地和池塘和9000m²的珍稀湿林地。这样的河岸改造力度可谓史无前例（图01）。

项目的挑战在于要使人们从观、行两方面都能有更好的亲水体验，因此设计要让河流更具可达性和开放性，并由此成为公园的焦点景观。

迈克·沃恩领导了阿特金斯的多学科设计团队，其中包括河流工程师、地貌学家和生态学家。“我们希望通过调整陡峭的斜坡来打开河流廊道，”迈克解释，“坡度降低后，河水被引入公园，也更具可达性。人们可以近距离靠近河边，观察这里到底发生了什么。”

精确的河堤几何处理是一门微妙的平衡艺术。河堤若太陡，河岸的人工加固成本将高居不下；河堤若太浅，河流会开始侵蚀场地上宝贵的空间。最终设计采用的坡度为1:2.5，约为22°。河堤的占地空间则受限于疏导洪水、陆地景观和基础设施的需要。因此，对于过度陡峭的河堤采用了两种办法：第一，尽可能地用椰皮纤维卷和木桩沿斜坡堆垒河提；对于一些坡度达70°的河堤部分则采用加固手段，设计了多层土工格栅和钢丝网笼，表面覆以草皮。

如今，新的景观正在迅速成型，利亚山谷的旧貌已难觅踪迹。在2006年伦敦奥运交付

管理局拿到这块土地之前，这里的河道纵横交错，被入侵的杂草所阻塞，还有旧购物车和汽车轮胎等城市废弃物被随处丢弃。

利亚山谷中没有被好好照管的河道网络不仅仅是城市的“眼中钉”，更成为了“绊脚石”——它将西面的哈克尼区（Hackney）和陶尔哈姆莱茨区（Tower Hamlets）与东面的沃尔瑟姆福雷斯特区（Waltham Forest）和纽汉区（Newham）分而隔之。

现在，重新焕发生命的水路以及横跨其间的新交叉点，不论在赛期还是2012年之后都将十分重要。它们是遗赛后规划方案的一个主要部分，将新公园及其水网与东伦敦大范围内的规划布局连接起来。

1 栖息地再焕生机

伦敦奥运交付管理局对奥林匹克公园寄予厚望，希望场地中大部分河流和自然特征都能创造可持续的栖息地。但是要将杂草和垃圾填塞的河谷转变为其最初的原始状态，是一个漫长而艰难的过程。

阿特金斯的工程师们先深入了解场地上盘枝错节的河道水路，并在一定时间内对不同地点的水流量和水速进行监测，将收集的数据构建了一个精细的水力模型，用以预测洪水风险。这点极其重要。因为阿特金斯承担着场地上平均海面（海平面）4m以内的所有责任。

在环境影响评估阶段，洪水风险的充分评估就已展开。分析不仅包括了由于频繁降雨造成的雨洪，还考虑了由于潮汐变化和蓄水所造成的水位自动调节。2008年期间，河道系统进行了蓄水，使得建模过程相当复杂。

实际上，尽管蓄水削弱了潮汐对泰晤士河的直接影响，但间接影响仍然存在。“当潮汐来临，它会阻碍河水排出李河。”迈克·沃恩解释，“因此，水位一天的变化能达到平均400mm。”

阿特金斯的模型计算准确地预测到了这一现象，以及由此增加的洪水风险。“这些发现带来了景观轮廓上的改变。”迈克说，“沿河道路被抬高了1m，同时对湿地的轮廓也做了抬升。对于它们的生存而言，保持正确的水位十分关键。”

公园大量运用了可持续的排水技术。在景观领域，广场采用了渗水性材料，水可流入与河边池塘相连的生态洼地。此外，还采用了地面运输、地下管道和存贮等多种手段（图02）。

由于原有的河堤许多部分非常陡峭，因此河道修复的第一步就是让河堤“躺下来”。经过几个世纪不断地堆积，有的地方地平面高达10m，堆在河堤上的垃圾有碎石、玻璃、动物骨头和更近一些时期从伦敦最东端运来的拆除的战时物品。

项目面临的另一个挑战是遍地的入侵植物，如喜玛拉雅凤仙、日本紫菀和巨猪草。这些都是19世纪由于园艺好奇心而引进英国的速生植物，却成失误之举。

入侵植物对于河堤是个坏消息。它们疯狂地繁殖、生长，驱逐本地植物，并且非常顽固。紫菀能在水泥中扎根；巨猪草含有呋喃香豆素和见阳光激活的毒素，会导致皮肤溃疡。因此清理成了优先考虑的工作——整个场地的土壤都做了处理，河堤上的所有植物被尽数铲除。

同时，阿特金斯还要保护地块上的动植物种群。2006年环境影响评估中即开展了第一阶段的栖息地调查工作，包括鸟类和鱼类的调查。同时，已着手进行了一项主要的物种迁移工作，规划了一批适宜的迁移地，包括临近公园营建的1hm²栖息地。阿特金斯迁移了330种普通蜥蜴、100种蟾蜍和4 000种欧洲滑螈。为了保护场地上的植物群，阿特金斯延续了“清除许可制”系统的做法，并详细规划了包括梧桐树林在内的多处免干扰栖息地地区。

2 选择栽种植物

阿特金斯承担着营造河堤和湿地最终景观的任务，因此需要决定场地种植的植物种类。面对清理后赤露的河堤，新栽种的植物不仅要满足生态和审美要求，还需要在赛期开花——但要符合工程规律。

团队选择了生态工程技术，即用植被和自然材料而非水泥来保护和巩固河堤。那些有着良好根系的合适的植物能有效保护河堤不被侵蚀。

但河网的半潮汐状态又提出了新的挑战。水位一天之内约400mm的升降会给新种植物带来很大的潜在威胁，河流的高载沙量也会让种下的植物面临缺氧危险。“我们确实没有一个自然的河道系统，”迈克说，“植物无法适应这种生存条件。”

为了找到成活率高的植物，并且建立最有效的种植方法，阿特金斯在奥林匹克公园内李河的一段50m长的河堤进行了特殊的种植试点。

“我们尝试了不同海拔高度的植物和不同种植技术，监测观察了1年。”阿特金斯高

图02

图03

级环境科学家伊恩·莫里西（Ian Morrissey）说，“这些工作对于我们选哪些植物和在哪里种植很有帮助。”

试点显示直接插种的植物太易受到伤害，但预种在椰皮纤维垫上能避免被冲走或淹没。椰皮纤维还有一个好处——能方便又快捷（图03）。

“垫子本身可用作覆盖物，因此能阻止可能混在筑堤土料中的杂草从中生长出来。但更重要的是，当河堤泛洪被淹没时，椰皮纤维具有很好的沉积物收集作用，帮助植物根部固定和汲取营养。”伊恩说。

3 寄望明日幼苗

创造可持续性的河堤生态系统意味着采用本地物种。因此，在河堤重整以前，已经收集了一批本地水生物种的种子——这一过程由阿特金斯进行管理——并且储存在一个种子银行。部分种子由生态工程和苗圃专家组萨利克斯团队(Salix)进行研究。他们受伦敦奥运交付管理局委托离线培育植物，这也被认为是英国有史以来最大的苗圃合同之一。

这项工作的工作量非常巨大，2009年6月就开始了播种。因为在面对河堤恶劣环境之前，植物必须经过一年生长并且妥善种在椰皮纤维托盘中。

萨利克斯团队在斯温西附近的半岛苗圃，培育了7 000多盆莎草等用于湿林地的植物。他们还为这个2012项目专门在诺福克新辟了一个面积约1hm²的苗圃（图04），种植了30多万株植物，大约28个不同品种，包括莎草、芦苇和驴蹄草等。它们种在上千个椰皮纤维托盘中，等待在稍后几个月里陆续运往伦敦。

2010夏季，18 000m²的植物聚集在一起，宛若一幅巨大的拼图。为了方便物流和排序，每个托盘和卷垫都做了标记。此外，在运输过程中还需要保证它们之间留有恰当的空间，防止切割到植物的根系和根茎。工程队将它们整批提取，确保不发生上述意外。

4 白手起家的池塘和湿林地

尽管阿特金斯在“旧利亚河”的河堤上花费了大量精力，他们仍然需要考虑全新的水体问题。公园北面滨水区域的生物多样性的基础部分有赖于远离东岸的3处新三角形池塘。其中两个池塘设计在夏季干涸，形成潮湿的草洼，第三处池塘则保持水体，以便诸如睡莲、驴蹄草等植物能够繁衍（图05）。

既要避免第三处池塘干涸，又要确保它不与利亚河一起泛洪，无疑是一个难题。阿特金斯在池塘和河道之间设计了一个连接渠，便于水量的补给和排出。池塘的水量变得可控：当水位太高，水便排出至河里；当池塘快干涸时，阀门打开，河水又可回到池塘。虽然这听起来简单，但被认为是首个案例在大规模栖息地采取如此做法。

新的湿林地与提升后的河道和河堤将一起成为奥林匹克公园的亮点。它们是现在英国境内罕见的栖息地，而公园内的这些栖息地更是白手起家建设出来的。

“这些任务非常新颖，”伊恩·莫瑞思回忆道，“挑战来源于我们要确保湿地区域的正确水位。阿特金斯负责解决地势和渠道的问题，以及它们和河流如何相互作用。”

湿地有逐渐萎缩并至最后干涸的趋势，但通过选择正确的物种、精细的水位管理和维护，这个过程可以得到延长。

“我们推荐的莎草科品种最终入选了，因为它们十分旺盛，并能与陆生物种竞争。”伊恩说。

湿地的树种包括柳树、桤木、桦树和现在稀有的黑杨，迈克沃恩指出：“它们很受野生动物的欢迎。有大量的无脊椎动物栖息这里，还有许多鸟类在此筑巢。”不过，鸟儿们会给河堤上新种下的植物带来挑战。

“野禽飞到场地上，可能会抓伤植物，”迈克说。为了防止这类事情，阿特金斯沿新植被区设计了几百米的防护栅栏，并已于2012年春季将防护栅栏拆除（图06）。

5 终点线之外

下利亚山谷地区的转变和已近完工的新公园，无论用什么标准来衡量，它们都是卓越非凡的。奥林匹克公园的参观者——在赛时高峰期将高达每天25万人——将看到一个最绿色、最环境友好型的奥林匹克公园。

公园在2012年之后仍将发挥效益。“我们真正的困难在于把力量投入到了基础设施上，这不仅有利于赛事，更作用于将来”伦敦奥运交付管理局的约翰·霍普金斯说，“这里的中心有河流、有公园，将成为一个生活、工作的理想之地。不论从社会、经济还是环境上来讲，都是很棒的遗产——新的景观驱动了新的城区。”


图04

图04 湿地植物进行了1年的离线培植

Fig.04 The wetland plants were grown for a year off site

图05 3处新修建的三角形池塘有利于生物多样性。

Fig.05 Three new triangular ponds were created to encourage biodiversity.


图05

How do you plant along a river's edge, knowing that millions of people could be passing through the site in the near future? How do you design, create and maintain the surrounding wetlands, knowing that man-made wet woodland is very rare and transitional by nature? How do you ensure that the habitat being created remains viable and sustainable in the long-term? Atkins' engineers of the wetlands and river edges on the London 2012 Olympic Park were tasked with finding answers to all of these questions.

Covering more than 246 hectares of formerly derelict industrial land, London's new Olympic Park for the London 2012 Olympic and Paralympic Games is one of Europe's biggest-ever urban greening projects. Rivers and wetlands are at the heart of the vision for the new park, which lies in east London's Lower Lee Valley. The landscape that's now emerging will provide a backdrop for the main action of the London 2012 Games.

As river edge and wetland engineers for the project, Atkins has played a critical role in turning the vision into reality. Atkins' remit includes design of the soft river edges and wetlands, including riverbank restoration and bioengineering.

The transformation is unprecedented. More than 8km of riverbanks have been restored as part of the project; in tandem with this, 2 hectares of reed beds and ponds have been created, along with 9,000 square meters of rare wet woodland(Fig.01).

The challenge was about getting people both visual and physical access down to the river-to actually make the rivers more accessible and more open, and therefore the centerpiece of the Park.

Mike Vaughan heads up Atkins' multidisciplinary design team, which includes river engineers, geomorphologists and ecologists. "The idea was to open up the river corridor by making the steep slopes that line the river flatter," explains Mike. "By dropping the slopes, we've brought the river into the park and made it much more accessible-people can get close to the river and see what's going on there."

Getting the riverbank geometry just right was a delicate balancing act. Too steep, and the banks would need costly artificial reinforcement; too shallow, and they would start to eat into valuable space on the site. An optimum slope of 1 in 2.5-about 22 degrees-was chosen. The space occupied by river bank was restricted by the need to convey floodwater and the location of terrestrial landscape and infrastructure. As such, the banks were over-steepened using two approaches. Firstly, where possible, the riverbanks were terraced using coir rolls and timber stakes. In other locations, where only a 70 degree bank was possible, a reinforced detail was used, providing layers of geo-grid and steel mesh cages, faced with a riverside turf.

Today, with the new landscape rapidly taking shape, it's easy to forget how the Lee Valley used to look. Until the Olympic Delivery Authority (ODA) took possession of the site in 2006, many of the river channels that criss-cross the site were clogged with invasive weeds, along with the predictable detritus of urban decay: abandoned shopping trolleys and car tires.

The Lee Valley's neglected river network wasn't only an eyesore, but also an obstacle-a gulf separating Hackney and Tower Hamlets in the west from Waltham Forest and Newham in the east.

Now, the revitalized waterways-and the new crossings spanning them-will be vital not only during the Games, but also after 2012. They are an integral part of the legacy solution, stitching the new Park and its waterways into the wider fabric of east London.

1 Bringing Habitats back to Life

Making the most of the site's rivers and natural features to create sustainable habitats is a key part of the Olympic Delivery Authority's vision for the Olympic Park. But the process of transforming the park's rivers from weed and rubbish-infested gulches into pristine watercourses has been long and tough.

For Atkins, that process started with developing an intimate understanding of the labyrinth of waterways and channels that wind their way through the site. Flows and velocities were measured at different points over a period of time, with data used to construct a detailed hydraulic model to predict flood risk. That's of critical importance, because Atkins had responsibility for everything up to a contour of 4 meters above ordnance datum (sea level) on the site.

A full flood risk assessment was undertaken at environmental impact assessment stage. Atkins undertook analyses of the risk of flooding caused by frequent rainfall, taking into account the automated regulation of water levels in the impounded reaches and the impact of tidal lockout. The modeling exercise was made considerably more complicated by the impoundment of the river system during the course of 2008; in effect, this eliminated the direct tidal influence of the Thames. But its indirect influence is still felt. "When the tide comes in on the Thames, it stops water flowing out of the River Lee," explains Mike Vaughan. "So the river levels fluctuate by an average of 400mm a day."

Atkins' modeling calculations correctly predicted this phenomenon, and also the increased risk of flooding. "These discoveries led to some changes in the landscaping profile," says Mike. "The

图06 几百米的防护栅栏用于保护新生的植物。
Fig.06 Hundreds of meters of deterrent fencing protected the new vegetation

图06

riverside paths have been raised by up to a meter and the profile of the wetlands was also raised, as maintaining correct water levels is critical to their survival."

Sustainable drainage techniques have also been used across the Park. In the landscape areas, porous strips have been used in the concourse, feeding into bioswales which drain down into the riverside ponds. Surface conveyance, underground pipes and storage features have also been utilized(Fig.02).

The first step in the river restoration process was to "lay back" the banks, many of which were precipitously steep. This re-profiling was necessary because much of the surrounding land was "made" ground, the result of centuries of tipping that had raised the ground level by as much as 10 meters in places. The cocktail of materials on the banks included rubble, glass, animal bones and, more recently, wartime demolition materials from London's east end.

Another challenge facing the Atkins team was the prevalence of invasive weeds. These included Himalayan balsam, Japanese knotweed and giant hogweed. All are fast-growing non-native plants introduced to Britain in the 19th century as garden curiosities; all have prospered on the wrong side of the garden wall.

Invasive species are bad news for riverbanks. They reproduce and grow with prodigious speed, driving out native plant species. And they're highly resilient. Knotweed can force its way through solid concrete, while giant hogweed contains furocoumarins, sun-activated toxins that can cause skin ulceration. Elimination was a priority – soil was treated throughout the site and the banks stripped of all remaining vegetation.

In addition, Atkins was responsible for ensuring the protection of the existing flora and fauna on the site. Phase one habitat surveys were undertaken as part of the environmental impact assessment in 2006, including bird and fish surveys. A major translocation of species was undertaken to suitable receptor sites including a specially-created 1 hectare site just outside the Park. Atkins translocated 330 common lizards, 100 toads and 4,000 smooth newts. In order to protect the flora on the site, Atkins maintained a 'permit to clear' system for contractors, and specified safeguarded habitat areas that were not to be touched including areas of sycamore trees.

2 Choosing Plants to Plant

Atkins is responsible for the final look of the riverbanks and wetlands-and deciding what to re-plant presented a challenge. With banks now bare, new planting would have to fulfill not only ecological and aesthetic demands-they'd be expected to be in bloom for the Olympic Games-but engineering imperatives too.

The Atkins design team chose bioengineering techniques, rather than culverting and hard engineering, for the project. That means protecting and consolidating riverbanks by using vegetation and natural products instead of concrete. Choosing the right species with the right root systems would be critical to protect the banks from erosion.

An added challenge was that the river network is semi-tidal. The twice-daily rise and fall of around 400mm had the potential to play havoc with new planting, and the river's high sediment loads threatened to smother anything planted from seed or plugs. "We don't actually have a natural river system," notes Mike. "Plants don't cope well in those conditions."

To find out which plants would fare best-and to establish the most effective planting methods -Atkins conducted a unique riverbank planting trial along a 50-metre stretch of the Lee in the Olympic Park.

"We trialled plants of different elevations and different installation techniques. These were monitored over a year," says Ian Morrissey, senior environmental scientist with Atkins. "That's really helped to inform exactly what species we should plant and where."

The trial revealed that plug plants would be just too vulnerable. But plants pre-grown in coir -coconut fibre matting-resisted being washed away or swamped. Coir has other benefits too-it's easy and quick to install in rolls and pallets two meters long and a meter wide(Fig.03).

"The mat itself acts like a mulch, so you prevent any weeds growing up through it that might already be within the bank material. But more importantly, when the banks become inundated, you get fine sediment trapped within the coir. That helps to bind the roots and feed the plants," says Ian.

3 Banking on Tomorrow's Seedlings

Creating a sustainable riverbank ecosystem means using native species. So before the banks were scraped back, seed was collected from suitable native aquatic species-a process managed by Atkins -and stored in a seed bank. Some of this seed was then used by bioengineering and nursery specialists, Salix, who were appointed by the Olympic Delivery Authority to cultivate plants offsite in what's believed to be one of Britain's biggest-ever nursery contracts.

The offsite growing operation was huge and sowing for the project commenced in June 2009, as plants must be a year old and well established in their coir pallets before encountering the tough riverbank environment.

Plants for the wet woodlands, including sedges, were raised in more than 7,000 pots at Salix's nursery on the Gower peninsula, near Swansea. And in Norfolk, the company created a new 16-acre nursery dedicated to the 2012 project(Fig.04). Here, more than 300,000 plants representing some 28 different species, including sedges, common reed, marsh marigolds and yellow flag irises, were grown on more than a thousand coir pallets, ready to be transported to London in the following months.

During the summer of 2010, the 18,000 square metres of planting were then pieced together like a giant jigsaw. This was a massive logistical challenge. To make it easier, each of the pallets and rolls was tagged. It was vitally important that each one went in exactly the right space so as to avoid cutting and trimming the roots and rhizomes of the plants. The team laid them out in blocks, to a plan, to make sure this didn't happen.

4 Ponds and Wet Woodlands from Scratch

While the riverbanks of the "Old River Lee" occupied much of the attention of the Atkins team, there were also entirely new bodies of water to consider. A fundamental part of the biodiversity of the river edges in the north of the Park lies in three new triangular ponds, off the east bank. Two of these were designed to dry up in the summer, forming moist grassy hollows. The third pond was created to retain water, enabling species such as water lilies and marsh marigold to thrive(Fig.05).

Preventing that third pond from drying out -while also ensuring that it did not flood along with the River Lee-was a conundrum. Atkins responded by designing a connection between the pond and the river to act as both overflow and feed. Flows could be regulated: when the pond level rose too high, water could be drained back into the river; when it started to dry out, a valve could be opened to release river water back into the pond. It sounds simple, but it is believed to be the first of its kind for a habitat feature of this scale.

As well as the improved waterways and riverbanks, new wet woodlands will be a notable feature of the Olympic Park. They're now a rare habitat in the UK, and the ones in the Park are being created from scratch.

"It was quite a novel thing to be asked to do," recalls Atkins' Ian Morrissey. "The challenge was to make sure we had the right water levels within the wet woodland areas. Atkins was responsible for working out the topographies and the channels, and how they would interact with the river."

Wetlands have a tendency to become dry land eventually, a process that can be slowed down through selecting the right vegetation, careful water level management and maintenance.

"The sedge species we selected were chosen because they are quite vigorous so are able to compete well with terrestrial species," says Ian.

Tree species for the wet woodland include willow, alder, birch and the now rare black poplar, points out Atkins' Mike Vaughan: "It's fantastic for wildlife. You get a lot of invertebrates in there, as well as nesting birds."

Birds, though, can present a challenge, particularly on the freshly planted riverbanks.

"There's a risk of wildfowl grazing our plants when they get on site," says Mike. To prevent that happening, hundreds of meters of deterrent fencing were erected around new vegetation. That stayed there until spring 2012(Fig.06).

5 Beyond the Finishing Line

The transformation of the lower Lee Valley and the creation of the new park, now nearing completion, is remarkable by any standards. Visitors to the Olympic Park – up to 250,000 every day at the peak of the Games – will encounter one of the greenest and most environmentally friendly parks ever to be created for the Olympics.

And the benefits will be felt long after 2012. "We're pulling that really difficult trick of putting in infrastructure that's good for the Games, but will work in legacy," said the ODA's John Hopkins. "This will be a great place to live and work, with rivers and parklands at the heart. Socially, economically and environmentally, there will be a terrific legacy-it's a new landscape powering a new piece of city."

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The Blackwater Valley Road: using green infrastructure for ecological mitigation

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The Blackwater Valley Road: using green infrastructure for ecological mitigation

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Summary

The Blackwater Valley Road comprises 17km of dual carriageway on the Hampshire/Surrey borders. The road was planned and built long before the term 'Green Infrastructure' became widely used, but it provides an excellent practical example of the approach. The success of ecological mitigation was first assessed in 2004 and reviewed again in 2011.

The cost of ecological mitigation is usually just a small part of the budget for a road scheme. Nevertheless, a cost is involved. Despite this, it is very rare for schemes to be reviewed to see whether the mitigation provided actually gave the desired results and to allow others to learn from the experience. One exception was the Blackwater Valley Road (A331) on the Hampshire/Surrey borders. In 2004 the success of ecological mitigation on this road scheme was assessed (Atkins *et al*. 2004). In 2011, 15 years after the road was completed, the authors review the results of monitoring studies. This paper is presented to allow others to share the experience, and also to ask whether the recommendations originally made in 2004 are now being put into practice more widely.

The Blackwater Valley Road comprises 17km of high speed dual carriageway linking the A30 with the A31 and M3 (Figure 1). It was conceived in the 1960s and then built in four stages between 1985 and 1996 by Hampshire and Surrey County Councils. Each of the four stages was the subject of a separate Environmental Impact Assessment (EIA); at the time these schemes were some of the earliest road schemes to be subject to EIA in the UK. The scheme was planned and built long before the term 'Green Infrastructure' became widely used (CIRIA 2011), yet the Blackwater Valley Road provides an excellent example of the approach.

The Blackwater Valley Countryside Project (BVCP) was formed in the 1960s to tackle the neglect and pollution in the Blackwater Valley and now works to increase the valley's importance for biodiversity and as a recreational resource for local people. The BVCP was involved in the scheme throughout its construction and remains responsible for management of the green infrastructure that was retained and created as part of the scheme.

The Blackwater Valley is a wedge of open space separating major urban areas on the Surrey/Hampshire/Berkshire borders. The landscape is dominated by a chain of lakes formed by sand and gravel extraction. Rapid urban expansion led to the degradation of the local landscape and resulted in traffic congestion in the urban areas along the valley. The scheme to build the Blackwater Valley Road was constrained by significant engineering challenges including the narrowness of the valley, the presence of lakes and rivers and by the need to construct an aqueduct for the Basingstoke Canal Site of Special Scientific Interest (SSSI).

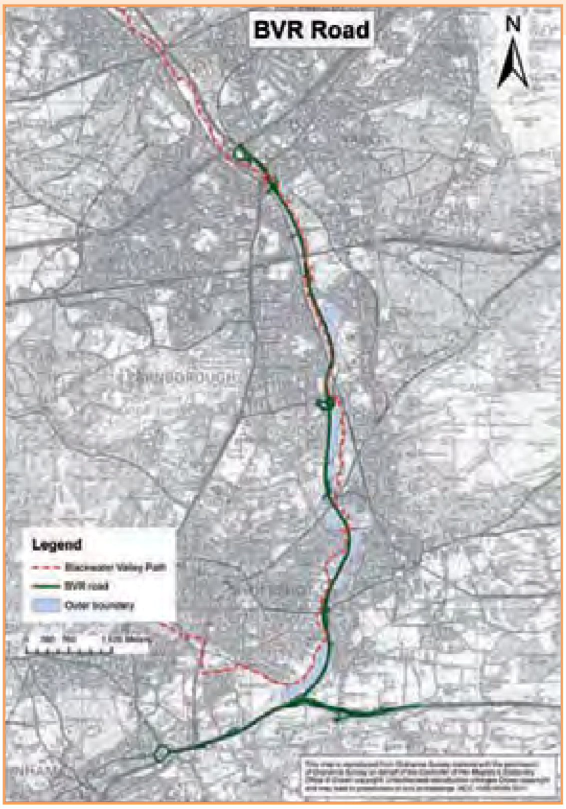

Figure 1. The Blackwater Valley Road (map provided by Hampshire County Council).

The EIAs for the four schemes recognised that the road would have significant effects on ecology as a result of:

- habitat loss – a net loss of 38ha;
- habitat fragmentation reducing the effectiveness of the valley as a wildlife corridor;
- noise disturbance;
- potential pollution of the River Blackwater from surface water run-off;
- effects on legally protected species due to habitat loss and severance; and
- loss of recreational facilities.

In order to mitigate significant effects, the two County Councils purchased substantial areas of land alongside the scheme. This enabled a comprehensive package of green infrastructure measures to be designed and constructed. The overall package of ecological measures included:

- avoidance of existing sensitive areas wherever possible;
- temporary fencing to prevent damage to adjacent areas during construction;
- design of river diversions to improve riparian habitat;
- habitat creation resulting in an increase in water bodies and woodland of 90ha;
- habitat management, such as tree removal from grassland and swamp areas;
- translocation of heathland, aquatic and marginal vegetation and individual rare plants;
- natural regeneration of chalk grassland communities (Figure 2);
- capture and translocation of reptiles, amphibians and fish;
- design of drainage ponds to provide wildlife habitat;
- construction of a tunnel for roosting bats and erection of bat and bird boxes;
- measures to protect the water quality of the river; and
- provision of a public footpath, doubling the area of open access land and improving the quality of an angling lake.

The results of monitoring exercises undertaken in 2004 and 2011 indicated that the habitat creation schemes were largely successful, although some of the new habitats, such as woodland, will still take many years to be of equal quality to those lost. Wildfowl populations have largely benefited from the borrow pits, which provided new water bodies, and woodland bird populations use the extensive new tree belts. Translocation of aquatic plant species was successful, whereas few of the translocated grassland plants survived. Populations of legally protected species have been retained. The habitat changes brought about by the road scheme also benefited many species not targeted by the mitigation work, such as the wildfowl that have benefitted from the new water bodies within the valley; for example, attempts to translocate common spotted orchid were unsuccessful.

Figure 2. A verge that was not sown with a seed mix and was allowed to regenerate naturally supports chalk grassland plant species. Photo by Tony Anderson, BVCP

Lessons for Other Schemes

The BVR was one of the earliest schemes EIA and ecological mitigation. Major issues and ideas identified in 2004 as a result of post-construction monitoring are set out below, with comments on whether they appear to have been taken on board more widely based on the hindsight available

No.	2004 recommendation	2011 update
1	Provide sufficient area for the essential environmental works. In this study as much as four times the area of land required for the road was required for the mitigation measures.	There is now a recognition that plans for mitigation should be prepared early, so that land genuinely required for mitigation can be included in any Compulsory Purchase Order. Nevertheless, the area affected by habitat creation and management for the BVR remains unusually large, although this reflects the sensitivity of the location.
2	Prior to preparing a landscape plan assess what important habitats and species will be affected and in what quantity. The plan can then establish a correct balance of habitats to meet the needs of the target species.	EIA procedure ensures that baseline conditions are identified. Policy measures such as national and local Biodiversity Action Plans help ecologists identify the important habitats and species with clear justification.
3	Avoid sensitive areas rather than relying on translocation. Translocation often fails and should be regarded as a valid method of salvage when loss is unavoidable, but essentially a means of enriching a newly created area and not a way of saving existing habitat.	The value of translocation as a salvage operation is increasingly recognised, and some improvements have been made in methods. The message that it is a method of last resort if in situ protection is not viable remains important.
4	Ensure works are correctly timed. Habitat creation work in advance of the habitat destruction during road construction allows time for establishment of habitat, and increases chances of success of translocated material.	Careful programme planning can ensure that time is allowed for mitigation measures. Commitments can be made to such programming can be set out in planning conditions or equivalents. Nevertheless, early consultation with ecologists is essential for project managers to take this into account.
5	Habitat creation and protection should concentrate on large scale and permanent features. Ephemeral features and small areas require intensive management to maintain in the long term.	Large scale permanent features are generally a better use of resources. However, there can be benefits in providing some ephemeral features, such as 'brownfield' bare ground, without entailing excessive costs if they are recognised as ephemeral and secondary vegetation is allowed to colonise naturally. As brownfield sites have been lost to development, even temporary provision of features that can be colonised by mobile species such as some specialist bare ground invertebrates contributes to their habitat (Box & Stanhope, 2004).
6	Address habitat fragmentation. If underpasses are not possible consideration can be given to installing 'green' bridges, specifically for wildlife; a reduction in road kills will also benefit road safety.	Understanding of successful design for wildlife underpasses, either purely for wildlife or combined with human access routes, has improved. There have also been a few examples in the UK of green bridges. However, the expense of green bridges means that they are rarely built, particularly if they cannot be incorporated into bridges required for people. For example, a green bridge was built over the Lamberhurst by-pass on the A21 in Kent, but this structure also provides the main access to the National Trust's Scotney Castle estate.
7	Do not over landscape. Current landscaping practice appears to be measured by the number of trees planted. This approach leads to tree planting on inappropriate areas, blocked footpaths and planting too dense to allow natural woodland flora to develop.	Landscaping understanding has improved, although tree planting schemes can still appear overly influenced by the desire to supress other plants at all costs and if dense planting is not followed by intense thinning, as would happen in planting for timber production, problems can arise. Establishment of scrub and woodland habitats through natural regeneration alone can be successful but is dependent on the distance and suitability of a suitable seed source and intensity of deer grazing. The authors feel that landscape schemes can also seem to sometimes focus on tree planting at the expense of good quality grassland creation.

Feature Article: The Blackwater Valley Road: using green infrastructure for ecological mitigation (contd)

No.	2004 recommendation	2011 update
8	Use native provenance vegetation. The long period of advance planning required for roads allows plenty of time to source and propagate all plant and seed requirements from local sources.	Use of native and, where justified, local provenance plants has increased. The problems that can arise if non-native stock is used are better understood. However, it can still be difficult to get funding for advance seed collection and propagation due to uncertainty of project approval and timescales. In addition, new questions will appear in the near future about the appropriateness of using plant stock more adapted to a drier and warmer climate than native plants. Such stock is already in use for some forestry schemes, and careful consideration needs to be given to the issues in ecological mitigation schemes. The authors' personal feelings are that such an approach underestimates the adaptability present in plants from the UK itself and this complex area needs more research. (In 2013 the potential implications of 'native' stock being collected in the UK but grown overseas for cost savings are also being highlighted as the scale of such operations is viewed in the light of risk of spreading plant diseases.)
9	Establish good working relationships between highway engineers and ecologists at an early stage. Communication between the different professions can be difficult but is essential to meet common goals. A dedicated Landscape Clerk of Works, involvement of local expertise, and a working group, are all ways of tackling the problem.	Use of an Ecological Clerk of Works is increasingly common on large schemes affecting sensitive habitats and species. Communication between disciplines at the design stage remains critical, and project managers should take control of this process and ensure that the specialists are talking to each other.
10	Monitoring should be put in place from the very beginning. Be prepared to alter designs and management to reflect monitoring results.	Some degree of monitoring is a common requirement prescribed in Environmental Statements. However, the feedback loop that allows this to influence designs and management and share results with others needs improvement. A particular challenge is provided where works are subject to licence due to effects on European Protected Species, as changes to the design would often require the submission and approval of a new Method Statement under the licence. This requirement could increase resistance to change.
11	Permitted development ancillary to construction needs to be strictly controlled with restoration conditions imposed. Site compounds, access roads, storage and processing areas etc can be highly damaging to the environment yet can be outside normal planning restrictions.	This recommendation is a reminder to ecologists, project managers and planning authorities to investigate requirements for ancillary development at an early stage.
12	Ensure continuing management so that beneficial impacts of mitigation measures are not lost. Mitigation measures should remain effective for the life time of the road. Funding to support mitigation measures should be an integral part of the long term highway maintenance budgets.	Funding in perpetuity cannot really be guaranteed. However, a commitment to a long term management plan helps ensure that this is taken into account in the highways maintenance programme.

Three key factors were identified as being instrumental in the success of the mitigation for this scheme:

- Ecologists worked closely with the highway engineers during design and construction of mitigation;
- The Blackwater Valley Countryside Partnership works closely with local authorities, private landowners and local community groups such as the Tongham Woodland Improvement Group (Figure 3) to manage the green infrastructure;
- Maintaining a flexible approach to management, based on monitoring of habitats and species, helped to direct and reshape mitigation measures, while continuing to focus on the original overall aims. For example, trees were removed from a number of plantations to allow naturally regenerating grassland flora to flourish.

The review of the Blackwater Valley Road recommendations shows that some aspects that were quite novel at the time have now become common practice, such as the use of an Ecological Clerk of Works. However, some lessons still need more consistent application. These include the need to mitigate impacts of ancillary permitted development and a tendency among some designers to plant more trees than necessary. Ecologists and land use planners are now experimenting with concepts of 'biodiversity offsetting'. Within such schemes, habitat translocation can add value to newly created habitat. However, this must not diminish the message that translocation is a method of last resort if *in situ* protection is not viable.

In summary, long before the phrase 'Green Infrastructure' came into common use, the Blackwater Valley Road scheme retained, created and managed 117 ha of land to provide multi-purpose benefits for people and wildlife. This green infrastructure was placed in local authority ownership and is being sympathetically managed to ensure that it provides green space for local people, habitat for wildlife and mitigation against the impacts of the road. At a time when budget constraints are likely to have an increasing influence on road and other major infrastructure schemes, it is essential that mitigation is as effective as possible.

Figure 3. Members of TWIG – Tongham Woodland Improvement Group – in their 'wood henge' seating area. Photo by Tony Anderson, BVCP

Green infrastructure with its multi-purpose approach to realising benefits is an excellent way of ensuring this. However, the exercise of monitoring and of reporting the results of mitigation is also important, to build a stronger evidence base to help demonstrate the which techniques work and allow others to learn from experience.

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Potential impacts of sea level rise on Sarasota Bay seagrasses

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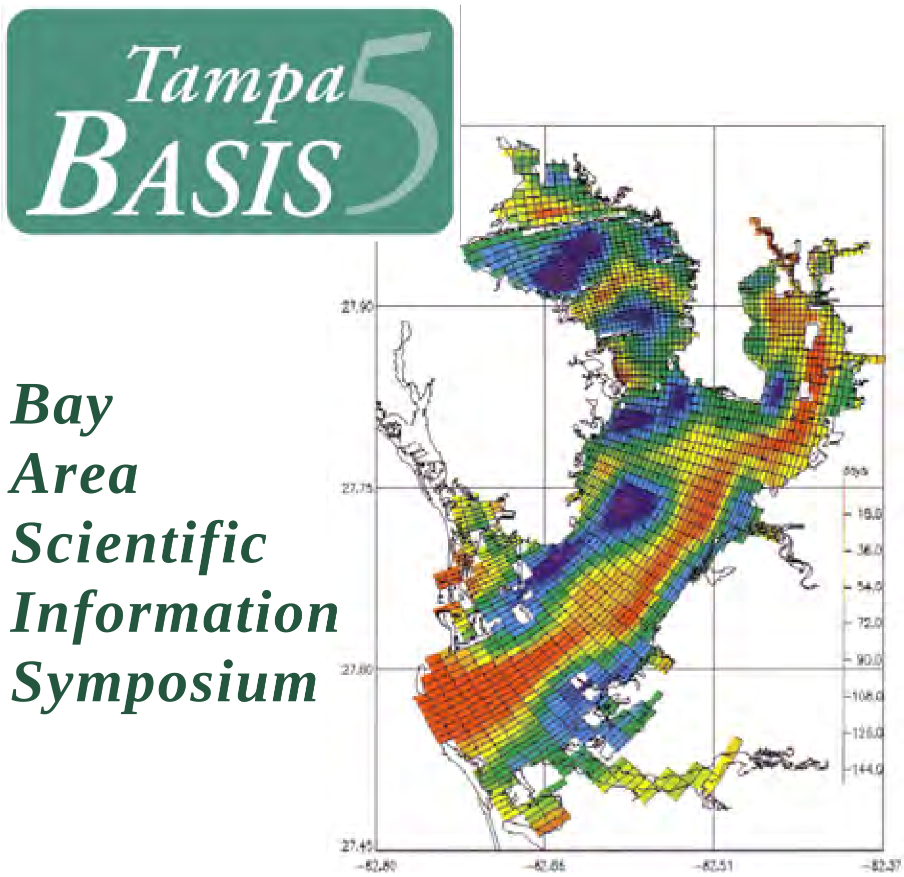

POTENTIAL IMPACTS OF SEA LEVEL RISE ON SARASOTA BAY SEAGRASSES

D.A. Tomasko & E.H. Keenan

ABSTRACT

In Sarasota Bay, correlations between levels of chlorophyll-a and water clarity allow for the development of chlorophyll-a targets to protect seagrass as a bio-indicator of system health. However, questions have arisen as to the appropriateness of seagrass maps in determining the depth to which seagrass beds can penetrate. Based on a comparison of techniques, it was found that surveyed offshore edges of seagrass meadows are typically 17 cm deeper than offshore edges delimited from seagrass maps produced via aerial photography and photointerpretation. These surveyed edges typically extend 30 to 40 meters farther offshore than edges delimited from seagrass maps. Based on a comparison between predicted depths from a 1950s-era bathymetric map and direct depth measurements, it appears that the waters of Sarasota Bay are approximately 17 cm deeper in 2007 than was the case in the 1950s, which is consistent with estimates of the degree of sea level rise (SLR) expected over a 50-year timeline (i.e. 1950s bathymetric layer to 2007 measured depths). If future SLR occurs at a similar rate, water clarity would have to improve, to allow the same light level to penetrate to the offshore edges of seagrass meadows growing in areas that would be expected to be deeper due to SLR. Bay-wide, chlorophyll-a levels would have to decrease by 13 to 35% by the year 2020 to allow seagrasses to maintain their present-day coverage in the future, deeper, waters of Sarasota Bay.

INTRODUCTION

In Southwest Florida, a substantial amount of research has focused on the relationships between population growth, changing land use patterns, pollutant loads, estuarine water quality, and seagrass health. In Tampa Bay, historical losses of seagrass coverage have been linked to both direct and indirect impacts (Lewis et al. 1985; Lewis 1989; Haddad 1989). In contrast, recent (1982-1996) increases in seagrass coverage in Tampa Bay have been linked to improved water quality (Johansson 1991; Johansson and Ries 1997; Lewis et al. 1998). Improvements in water quality have in turn been attributed mostly to improvements in the treatment and disposal of wastewater discharges by the cities of Tampa, St. Petersburg and Clearwater since the mid to late 1980s (Johansson and Greening 1999).

A similar situation exists in Sarasota Bay, where recent increases in seagrass coverage were related to reductions in anthropogenic nitrogen loads to the bay by the City of Sarasota and Manatee and Sarasota counties (Kurz et al. 1999).

Nitrogen loads into Tampa Bay and Sarasota Bay (generated primarily by domestic wastewater treatment facilities) were reduced by 57% and 46%, respectively, during the 1980s and 1990s (Tomasko et al. 2005). In response, both Tampa Bay and Sarasota Bay have lower phytoplankton concentrations, greater water clarity and more extensive seagrass coverage in recent years than in the early 1980s (Johansson and Greening 1999; Kurz et al. 1999; Tomasko et al. 2005). As there is no evidence of a concurrent and explanatory trend in rainfall during this same time period, it was concluded that variation in rainfall could not account for the observed increase in seagrass coverage in these two bays (Tomasko et al. 2005).

While chlorophyll-a can be a useful indicator of impaired conditions in an estuary, interpreting the causes and effects of varying chlorophyll-a concentrations is challenging. Chlorophyll-a is known to be highly variable and responsive to a variety of factors, including nutrient loads, rainfall, salinity, residence time, season, and even the time of day. In addition, factors other than chlorophyll, such as turbidity and color, may also affect water transparency. If those factors affect water clarity to such a degree that they may also affect phytoplankton growth rates, then issues other than nutrient availability alone may need to be accounted for to develop potential nutrient limits for water quality protection and/or restoration.

For Sarasota Bay, there appear to be two main courses of action that could be pursued, as relates to the issue of potentially biologically relevant values of chlorophyll-a, as summarized below.

Approach 1 - Acceptance of existing standards for chlorophyll-a−This approach would be to accept the state's determination that a chlorophyll-a concentration of 11 µg/liter, and the water clarity that accompanies such conditions, is an appropriate target. Or, that a 50% increase in levels of chlorophyll-a over "historical' levels is an indication of an impaired condition.

Approach 2 - Establish site-specific and resource-linked targets for chlorophyll-a−A second approach would be to develop site-specific and scientifically-defensible water clarity and chlorophyll-a goals that could be linked to a living resource. An approach such as this, using science-based management to protect the health of living resources, is similar to that currently being used by the Tampa Bay Estuary Program.

This paper summarizes efforts conducted in Sarasota Bay related to Approach 2, the development of site-specific and resource-linked targets for chlorophyll-a. Additionally, this paper addresses the issue of whether or not "offshore" or "deep" edges from seagrass mapping efforts are the same as a "deep" edge that would be found from site visits and careful depth measurements. Finally, the potentially important issue of sea level rise, as it relates to the development of resource-based water clarity and chlorophyll-a targets for Sarasota Bay is addressed.

MATERIALS AND METHODS

The techniques used in this paper involved the following steps: 1) identification of 'mapped' deep edges of seagrass meadows, 2) determining the 'surveyed" deep edges of seagrass meadows, 3) assessing the percent coverage of seagrasses at these two depths, 4) determining the depths at these two potentially different locations, 5) determining the light levels at these depths, and 6) determining the levels of chlorophyll-a that would allow for adequate light penetration to target depths, and 7) determining the implications of sea level rise on potential chlorophyll-a targets. For this effort, the original Sarasota Bay segmentation scheme used in the SBNEP's Framework for Action (SBNEP 1992) and the Comprehensive Conservation and Management Plan (SBNEP 1995) was utilized. Table 2 contains information on the relationship between portions of Sarasota Bay and the corresponding SBNEP bay segment numbers used in this paper.

Site Identification

Segments 7, 8, 10 and 11 (Sarasota Bay)−PBS&J, working in concert with Janicki Environmental, verified seagrass coverage and water depth information along the mapped deep-edge of the seagrass beds within Sarasota Bay. As such, Janicki Environmental created a "Universe" of sampling stations along the deep-edge of the consistently mapped seagrass beds performed by the Southwest Florida Water Management District (SWFWMD). Twenty randomly selected stations were provided by Janicki Environmental for each of the segments in Sarasota Bay (bay segments 7, 8, 10 and 11). PBS&J randomly selected eight sites for field verification of both seagrass coverage and water depth in each segment (Figure 1). The sites were visited on two calm weather days, February 20th and 21st, 2007. All sites underwent QA/QC review for site accuracy; four sites were removed from analysis based on spatial inconsistencies with map layers.

Segments 1, 3, 13, 14 and 16–PBS&J identified 7-9 mapped deep-edge sampling points based upon the 2006 SWFMWD seagrass coverage maps. Four sites were randomly selected to verify both seagrass coverage and water depth in bay segments 13, 14 and16 (Figure 1). These data were collected on May 4, 2007 during calm weather conditions. Five sites were randomly selected to verify both seagrass coverage and water depth in segments 1 and 3 (Figure 1). These data were collected on May 14, 2007 during calm weather conditions.

Seagrass Coverage–At all mapped deep-edge sites, a modified Braun-Blanquet method was used to determine seagrass coverage. Three assessments of seagrass coverage and species diversity were recorded at each site using a 1m^2 quadrat. Each site was evaluated to determine if it satisfied the Virnstein et al. 2000 method for classification of the deep-edge of a seagrass bed; <10% seagrass coverage (Virnstein et al. 2000). If it was determined that the site had >10% seagrass coverage, divers continued to swim offshore until they eventually located the deep-edge, as described above. The location and water depth of the surveyed deep-edge were recorded.

Depth Measurements–Water depth information was collected at a total of 89 sampling sites throughout all bay segments. A total of 39 depth measurements were recorded at the mapped deep-edge sites and an additional 50 measurements were made at the surveyed deep-edge sites. Eleven of the 50 surveyed deep-edge sites were consistent with the location of the mapped deep-edge. Three depths were recorded at each site using a customized staff gauge with a flat bottom disk designed to reduce errors associated with field measurements. All data were collected during calm bay conditions. The GPS coordinates were recorded for each station using a WAAS (Wide Area Augmentation System) enabled Garmin GPSmap 60CSx.

Depth data were corrected for tidal stage and referenced to Mean Low Water (MLW; Table 2). The 6-minute verified water level data from the St. Petersburg, FL (8726520) tide station was used to model the tide for the Cortez, Sarasota and Venice tide stations (http://tidesand currents.noaa.gov). The tidal prediction for each station was used to modify the St. Petersburg data. The St. Petersburg tide data were adjusted for high and low tide and a time shift was applied for each station.

NOAA NOS Bathymetry–Bathymetry data was collected in Sarasota Bay by the U.S. Coast and Geodetic Survey between 1953 and 1955 (Figure 2; www.estuarinebathymetry.noaa.gov). The majority of depth soundings were completed in 1954. Six surveys were completed collecting approximately 52,299 depth values using a portable depth recorder. The average distance between readings was 51 meters. The vertical datum was referenced to Mean Low Water. The horizontal datum was referenced to NAD83. This data set provided bathymetry information for all Segments listed in Table 2.

Depth Comparison–Using GIS, the closest NOS water depth point to each field site was selected. The 1950s NOS and 2007 surveyed water depths of all 89 sites were statistically compared using a non-parametric two-sample Mann-Whitney (Wilcoxon) test. Additionally, the water depth at the 2007 mapped deep-edge and the 2007 surveyed deep-edge sites were compared using the Mann-Whitney (Wilcoxon) test, and the distance between the mapped deep-edge and surveyed deep-edge was determined.

Chlorophyll-a and Light Attenuation Data–Chlorophyll-a and the light attenuation coefficient (K) data for each bay segment were downloaded from the Florida STORET database (www.dep.state.fl.us/water/storet). These data were available for all segments during the period of August 2001 to February 2006. Light attenuation coefficients (K/meter) in Segment 1 and 3 were calculated from the raw underwater incident light readings where z=0.5m (personal communication, Greg Blanchard). Additionally, only uncorrected chlorophyll-a values were available for Segment 1 and 3. Corrected chlorophyll-a was available for all other segments. Statistical significance was determined based on the r^2 value and sample size (Schefler 1979).

The average K was determined for each segment based on the monthly averages from August 2001 to February 2006. Percent sub-surface irradiance was calculated for each location at four corresponding depths (Mapped Deep-Edge (1950s), Mapped Deep-Edge (2007); Surveyed Deep-Edge (1950s) and Surveyed Deep-Edge (2007); Formula 1), and then the average percent sub-surface irradiance (or PAR -photosynthetically active radiation) for each segment at each depth was determined.

Formula 1: Percent PAR = +exp (-k*z)*100
Where k=average light attenuation coefficient and z =water depth in meters at MLW

Chlorophyll-a Target−The chlorophyll-a target for segments 7, 8, 10, 11, 13, 14 and 16 were determined based on the average percent sub-surface irradiance at the Surveyed Deep-Edge (2007). The light attenuation coefficient was calculated based on PAR and the water depth at which light readings were made (Formula 2).

Formula 2: K = -1/z * ln(Iz / Io)
Where Iz = PAR at depth z (in meters) and Io = directly sub-surface PAR

The current target chlorophyll-a concentration was calculated based upon the linear model relating K and chlorophyll-a for each segment (Table 3). A target chlorophyll-a concentration was also calculated based upon anticipated sea-level rise in Sarasota Bay for the years 2020 and 2050 (after Clark 1992). Using the measured rate of sea-level rise between 1954 and 2007, sea-level rise was simulated for the next 13 and 43 years. The modified water levels were used to recalculate the new (lower) K value required to keep the same percent PAR at a location with additional water depths (due to sea level rise) and the revised chlorophyll-a target was calculated based on the new estimate of K.

RESULTS

Water Depth Comparison−Tide-corrected water depth measurements from 89 sites were compared to the water depth estimates for these same locations using the bathymetry data compiled in 1953-1955 by the USGCS. The average (n = 89) water depth for sites visited in Sarasota Bay, based on the 1950s bathymetry data, was 1.09 m at MLW compared to an average depth of 1.26 m at MLW water depth for those same locations, calculated from directly measured water depths in 2007 (Figure 3). The change in water depth was 0.17 m which is equivalent to 3.2 cm/decade. There was a statistically significant difference between the 2007 water depths and the 1950s data (N=89; p=0.01). This difference in water depths is within the range of documented rates of sea level rise in St. Petersburg during 1950 to 1990 (2.4 cm/decade), and estimated rates of future sea level rise of approximately 5.2 cm/decade, as outlined in Clark (1992).

Additionally, the mapped deep-edge (2007) sites were compared to the surveyed deep-edge (2007) sites; the average water depth was 1.16 m at MLW and 1.34 m at MLW, respectively (Figure 4), but this difference was not statistically significant (N=39; p= 0.09). The average distance between the mapped deep-edge (2007) and the surveyed deep-edge (2007) was 42 m.

Chlorophyll-a Targets

Segments 1 and 3: A significant correlation between chlorophyll-a and light attenuation was not found for Segments 1 and 3, which correspond to Anna Maria Sound and Palma Sola Bay. In these parts of Sarasota Bay, prior studies concluded that non-chlorophyll suspended material (NCSM) was the dominant attenuator of light in the water column (Dixon and Kirkpatrick 1995). On average, NCSM explained 77% of the partitioned non-water attenuation for diffuse light in Segment 1 and 58% in Segment 3. Consequently, a chlorophyll-a target was not calculated for these segments.

Segment 7: Based on the data from August 2001 to February 2006, the average chlorophyll-a concentration and light attenuation coefficients were 5.4 µg/l and 0.6681 K/m, respectively (Table 4). The linear model (Table 3) was used to produce target chlorophyll-a values for Segment 7. Based on the Surveyed Deep-Edge water depth and percent PAR, a chlorophyll-a target of 5.3 µg/l was calculated to maintain deep edge light requirements. A potential IWR target was calculated based upon the chlorophyll-a value which exceeds the historic value by 50% for WBID 1968B (Sarasota Bay; 5.5µg/l). The 2020 and 2050 target chlorophyll-a concentrations were calculated based on anticipated sea-level rise. A projected water depth of 1.63m in 2020 and 1.76m in 2050 was used. To maintain current water clarity conditions with increased water depth, a target chlorophyll-a concentration of 4.6 µg/l is proposed for 2020. The proposed target chlorophyll-a concentration for 2050 is 2.6 µg/l.

Segment 8: Based on the data from August 2001 to February 2006, the average chlorophyll-a concentration and light attenuation coefficient were 5.4 µg/l and 0.6681 K/m, respectively (Table 4). The linear model (Table 3) was used to produce target chlorophyll-a values for Segment 8. Based on the Surveyed Deep-Edge water depth and K, a chlorophyll-a target of 5.0 µg/l was calculated to maintain current deep edge light requirements. A potential IWR target was calculated based upon the chlorophyll-a value which exceeds the historical value by 50% for WBID 1968B (Sarasota Bay; 5.5µg/l). The 2020 and 2050 target chlorophyll-a concentrations were calculated based on anticipated sea-level rise using projected water depths of 1.61m and 1.74m, respectively. To maintain current water clarity conditions with increased water depth, a chlorophyll-a concentration of 4.3 µg/l is proposed for 2020. The proposed target chlorophyll-a concentration for 2050 is 2.2 µg/l.

Segment 10: Based on the data from August 2001 to February 2006, the average chlorophyll-a concentration and light attenuation coefficient were 3.8 µg/l and 0.6372 K/m, respectively (Table 4). The linear model (Table 3) was used to produce target chlorophyll-a values for Segment 10. Based on the Surveyed Deep-Edge water depth and K, a chlorophyll-a target of 1.9 µg/l was calculated to maintain current deep edge light requirements. A potential IWR target was calculated based upon the chlorophyll-a value which exceeds the historic value by 50% for WBID 1968B (Sarasota Bay; 5.5µg/l). The 2020 and 2050 target chlorophyll-a concentrations were calculated based on anticipated sea-level rise, therefore, projected water depths of 1.85m and 1.98m were used, respectively. To maintain current water clarity conditions with increased water depth, a target chlorophyll-a concentration of 1.2 µg/l is proposed for 2020. Due to further increases in water depth, a target chlorophyll-a concentration could not be derived for Segment 10 for 2050 conditions.

Segment 11: Based on the data from August 2001 to February 2006, the average chlorophyll-a concentration and light attenuation coefficient were 4.5 µg/l and 0.6837 K/m, respectively (Table 4). The linear model (Table 3) was used to produce target chlorophyll-a values for Segment 11. Based on the Surveyed Deep-Edge water depth and K, a chlorophyll-a target of 6.4 µg/l was calculated to maintain current deep edge light requirements. A potential IWR target was calculated based upon the chlorophyll-a value which exceeds the historic value by 50% for WBID 1968B (Sarasota Bay; 5.5 µg/l). The 2020 and 2050 target chlorophyll-a concentrations were calculated based on anticipated sea-level rise using projected water depths of 1.19m and 1.32m, respectively. To maintain current water clarity conditions with increase water depth, a target chlorophyll-a concentration of 3.3 µg/l is proposed for 2020. The proposed chlorophyll-a concentration for 2050 is 1.7 µg/l.

Segment 13: Based on the data from August 2001 to February 2006, the average chlorophyll-a concentration and light attenuation coefficient were 7.7 µg/l and 0.9993 K/m, respectively (Table 4). The linear model (Table 3) was used to produce target chlorophyll-a values for Segment 13. Based on the Surveyed Deep-Edge water depth and K, a chlorophyll-a target of 3.9 µg/l was calculated to maintain current deep edge light requirements. A TMDL target was developed based upon the chlorophyll-a value which exceeds the historic value by 50% for WBID 1968D (Roberts Bay; 7.2 µg/l). The 2020 and 2050 target chlorophyll-a concentrations were calculated based on anticipated sea-level rise and projected water depths of 1.26m and 1.39m, respectively. To maintain current water clarity conditions with increased water depth, a target chlorophyll-a concentration of 5.5 µg/l is proposed for 2020. The proposed chlorophyll-a concentration for 2050 is 3.1 µg/l.

Segment 14: Based on the data from August 2001 to February 2006, the average chlorophyll-a concentration and light attenuation coefficient were 7.7 µg/l and 0.9105 K/m, respectively (Table 4). The linear model (Table 3) was used to produce target chlorophyll-a values for Segment 14. Based on the Surveyed Deep-Edge water depth and K, a chlorophyll-a target of 7.7 µg/l was calculated to maintain current deep edge light requirements. A potential IWR target was calculated based upon the chlorophyll-a value which exceeds the historic value by 50% for WBID 1968E (Little Sarasota Bay; 8.6 µg/l). In comparison, the TMDL target derived by the Florida Department of Environmental Protection and adopted for the improvement of water quality in Segment 14 is 8.6 µg/l. The 2020 and 2050 target chlorophyll-a concentrations were calculated based on anticipated sea-level rise and projected water depths of 1.01m and 1.14m, respectively. To maintain current water clarity conditions with increased water depth, a target chlorophyll-a concentration of 6.6 µg/l is proposed for 2020. The proposed chlorophyll-a concentration for 2050 is 3.8 µg/l.

Segment 16: Based on the data from August 2001 to February 2006, the average chlorophyll-a concentration and light attenuation coefficient were 5.6 µg/l and 0.9105 K/m, respectively (Table 4). The linear model (Table 3) was used to produce target chlorophyll-a values for Segment 16. Based on the Surveyed Deep-Edge water depth and K, a chlorophyll-a target of 4.7 µg/l was calculated to maintain current deep edge light requirements. An IWR target was calculated based upon the chlorophyll-a value which exceeds the historic value by 50% for WBID 1968F (Blackburn Bay; 5.5µg/l). The 2020 and 2050 target chlorophyll-a concentrations were calculated based on anticipated sea-level rise and projected water depths of 0.94 m and 1.08 m, respectively. To maintain current water clarity conditions with increased water depth, a target chlorophyll-a concentration of 4.0 µg/l is proposed for 2020. The proposed chlorophyll-a concentration for 2050 is 2.2 µg/l.

DISCUSSION

The results of this project can be summarized as follows:

- For much of the bay, correlations between chlorophyll-a and water clarity allow for the development of potential water clarity goals or "targets"
- For much of the bay, chlorophyll-a targets could be developed to protect seagrass resources as a "bio-indicator" of system health

- However, chlorophyll-a and water clarity do not appear to be correlated with each other in either Palma Sola Bay or Anna Maria Sound
- Detailed surveying of the deep edge of seagrass meadows results in a multi-year annual average light requirement of approximately 31 to 46% of sub-surface irradiance (at MLW) for Sarasota Bay
- Surveyed seagrass meadow deep edges are typically 17 cm deeper than "deep edges" delimited by aerial photography and mapping techniques
- Surveyed seagrass meadow deep edges typically extend 30 to 40 meters farther offshore than edges delimited by mapping techniques
- Based on a comparison between predicted depths from a 1950s-era bathymetric map and current depth measurements, it appears that the waters of Sarasota Bay are approximately 17 cm deeper now (2007) than was the case in the 1950s
- The rate of sea level rise (SLR) that appears to have been documented in Sarasota Bay is in line with estimates of the degree of SLR expected over a 50-year timeline
- If future SLR occurs at a similar rate, water clarity would have to be improved to allow the same light level to penetrate to the offshore edges of future seagrass meadows at their current locations
- To maintain the same light penetration to future, deeper waters, chlorophyll-a levels would have to decrease by 51 to 56% by the year 2050, to allow seagrasses to maintain their present coverage (Table 5).

It is not surprising that surveyed offshore edges of seagrass meadows, delimited via snorkeling, extend farther offshore and into deep waters than offshore edges delimited via aerial photography and photointerpretation. It would be difficult for aerial imagery, regardless of the technique used, to be sufficient to pick up all the seagrass resources found in any particular area. However, the more relevant implication of this finding is that "edges" found by snorkeling would have deeper depths, and would have different light levels at those depths than locations depicted by a map as being a deep edge. For this reason, it is important that the type of edge (surveyed or mapped) is consistent with the literature, for developing minimum light requirement targets for seagrasses. For example, estimates of minimum light requirements for seagrasses determined in Tampa Bay (Dixon 1999), Sarasota Bay (Dixon and Kirkpatrick 1995), and Lemon Bay (Tomasko et al. 2001) range as high as 42% of immediately sub-surface irradiance. In all three of these locations, the deep edge used for estimating light requirements was defined by field surveying the site, and not by photointerpretation of aerial photography and mapping techniques. The choice of what target light level to use for maintaining seagrass coverage is an important one for setting water clarity targets, and thus for setting possible chlorophyll-a targets.

An additional consideration for TMDL development in Sarasota Bay is whether or not there is evidence of system-wide improvements or degradation in water quality, based on ambient water quality monitoring programs or other objective assessments. That is, is impairment status derived from the rules and regulations outlined in the Impaired Waters Rule (IWR) consistent with more locally-collected data on trends in appropriate bio-indicators of system health?

In the SBNEP's document "Framework for Action", Lowrey et al. (1993) concluded that of the 16 bay-wide segments (plus the nearshore waters of the Gulf of Mexico) in Sarasota Bay, roughly two-thirds of those segments had trends in chlorophyll-a. Of those segments with trends, 93% of them had trends of decreasing values of chlorophyll-a, while only one segment had a trend of increasing values. Likewise, trends for total nitrogen were found in 43% of SBNEP segments, with twice as many having decreasing trends as increasing trends. For total phosphorus, 48% of bay segments had significant trends, with only 10% of those segments showing signs of degradation over time (i.e., increasing values)

For water clarity (measured as Secchi disk depths) 29% of bay segments had trends over time - all of those trends were of improving water clarity (i.e., none showed evidence of degrading water clarity).

In a later assessment of status and trends in water quality, Dixon and Heyl (1999) found that more recent (1989 to 1998) trends of declining concentrations of total nitrogen and total phosphorus (9 of 12 bay segments) were not accompanied by concurrent decreases in phytoplankton abundance, as no bay segments showed trends of decreasing chlorophyll-a concentrations during that same time period. The basis for this potential disconnect between declining nutrient concentrations and further decreases in phytoplankton abundance (between 1989 and 1998) merits further investigation, especially as it relates to the finding (discussed below) of continued increases in seagrass coverage in Sarasota Bay (Kurz et al. 1999; Tomasko et al. 2005).

In the SBNEP's Framework for Action, the technical synthesis chapter (Tomasko et al. 1992) outlined the finding that seagrass biomass and productivity was a better indicator of segment-specific nitrogen loads than either chlorophyll-a or Total Kjeldahl Nitrogen (TKN). A key finding of Tomasko et al. (1996) was that the status and trends of seagrass coverage in Sarasota Bay would be a better indicator of system health than traditional water quality monitoring programs.

As seagrass coverage has been found to be increasing in Sarasota Bay in recent years (Kurz et al. 1999; Tomasko et al. 2005), these results would indicate that findings of impairment for nutrients in various Sarasota Bay segments (e.g., Palma Sola Bay, Roberts Bay, Blackburn Bay) might be more a function of a potentially locally inadequate listing criteria via the IWR than a true indicator of a stressed ecosystem.

However, results of the depth measurements, comparing 1950s bathymetry estimates vs. field survey estimates from 2007 suggest that sea level rise could be an important factor to consider, when developing water quality goals for Sarasota Bay. The average increase in water depth of 17 cm, comparing 1950s depths to 2007 estimates, calculates out to a sea level rise rate of approximately 3.2 cm/decade. This difference in water depths is within the range of rates of measured sea level rise in St. Petersburg during 1950 to 1990 (ca. 2.4 cm/decade), and estimated rates of future sea level rise (ca. 5.2 cm/decade) as outlined in Clark (1992). Should these estimates of future sea level rise prove accurate, chlorophyll-a concentrations in Sarasota Bay would have to be reduced by 51 to 56 percent, to allow for the present-day light levels to penetrate to the same, but deeper, offshore edges of future seagrass meadows in the bay.

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Otters in Scotland: How Vulnerable Are They to Disturbance?

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Otters in Scotland: How Vulnerable Are They to Disturbance?

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The otter population in Scotland is increasing and their distribution is expanding; the fourth national Scottish survey confirmed that the species is present on all of Scotland's major watercourses (Strachan 2007). Otters are also expanding into rest of the UK, with the last England national otter survey recording positive evidence throughout England (Crawford 2003). This expansion in numbers, following improvements in water quality and subsequent increases in fish stocks, along with reduction in persecution and an increase in their level of legal protection has led to the species venturing more often into urban areas. In 2002, the Water for Wildlife Project found otters in over 100 towns and cities throughout the UK, with 13 urban areas having resident breeding populations (Wildlife Trusts 2002).

Strachan's 2007 study found that otters have moved into urban areas, such as Edinburgh and Glasgow, and reported that the species seemed unaffected by human activity. However, our experience of working in remote areas suggests that it is not just individuals living in urban environments that are tolerant of such disturbance. Through working on a variety of development schemes across Scotland, where otter resting sites of varying degrees of importance have been recorded, the author and colleagues within Atkins have established an appreciation of the susceptibility of otters to disturbance. This article presents these case studies and discusses to what degree otters are actually affected by human disturbance.

Observations by Kruuk and colleagues (Chanin 2003) indicate that otters will rest under roads, in industrial buildings, close to quarries, and at other sites close to high levels of human activity. These observations clearly indicate that otters are very flexible in their use of resting places and breeding sites and do not necessarily move away to avoid 'disturbance' from noise or the proximity of human activity (Kruuk 2007). Durbin (1998) studied otters on the Rivers Dee and Don (North East Scotland) but no evidence was found to suggest that otters actively avoid areas of human activity, either when foraging or when resting. This has been supported by the survey findings of Atkins' ecologists which have included otter resting sites immediately adjacent to roads, next to railway lines and in the middle of golf courses. Otters have also been seen passing under a railway bridge in broad daylight, whilst construction work was occurring directly above. All of these observations support the theory that otters are able to tolerate and become habituated to a degree of disturbance and noise.

Legislation

A summary of the legislation protecting otters in Scotland is given in Table 1. The legislation does not contain a definition of the term 'disturbance'. However, some guidance is provided by the European Commission (EC) (2007) and in a joint guidance note produced by Natural England and the Countryside Council for Wales (2007) on the interpretation of disturbance in relation to legally protected species. The EC guidance says that disturbance need not directly affect the physical integrity of a species but can nevertheless have a direct negative effect that is detrimental for a protected species, *e.g.* by reducing survival chances, breeding success or reproductive ability. For example, this could apply to anything that deters an otter from using its holt or anything that prevents an otter moving along a watercourse. The guidance goes on to define 'disturbance', in terms of Article 12, as any disturbing activity that affects the survival chances, the breeding success or the reproductive ability of a protected species or leads to a reduction in the occupied area. On the other hand, sporadic disturbances without any likely negative impact on the species should not be considered as disturbance under Article 12.

The EC recommends a species-by-species approach to be taken as different species will react differently to potentially disturbing activities. Therefore, the decision as to whether a project will cause disturbance that would result in an offence is left to the specialist otter consultant and the statutory nature conservation agency.

The legislation protects structures and sites that are used for shelter, protection, resting or breeding; these are referred to hereafter as resting sites. In Scotland, Scottish Natural Heritage's stance usually dictates that any resting site within 30 m of development of any scale and duration requires a European Protected Species (EPS) licence for 'disturbance' (and 100-200 m in the case of breeding sites, depending on the local circumstances). These licensing zones were agreed following consultation with a number of UK otter experts. While they may appear somewhat

Table 1. Legislation protecting otters in Scotland

Legislation	Offences
Conservation (Natural Habitats, &c.) Regulations 1994 (as amended) Reg.39.	Deliberately or recklessly capture, kill, injure, harass or take an otter; Damage or destroy a breeding site or resting place of an otter (note this is a strict liability offence *i.e.* the act does not have to be deliberate or reckless) Deliberately or recklessly disturb an otter while it is occupying a structure or place it uses for shelter or protection Deliberately or recklessly disturb an otter while it is rearing or otherwise caring for its young Deliberately or recklessly obstruct access to a breeding site or resting place of an otter or to otherwise deny it the use of the breeding site or resting place Deliberately or recklessly disturb an otter in a manner that, or in circumstances which are likely to impair its ability to survive, breed or reproduce, or rear or otherwise care for its young; and Deliberately or recklessly disturb such an animal in a manner that is, or in circumstances which are, likely to significantly affect the local distribution or abundance of the species to which it belongs.

arbitrary, it is useful to have a standard basic distance defined within the guidance so that developers are clear about what is likely to be required. Having no such distance defined puts the onus heavily on the developer to demonstrate that works close to an otter shelter will not cause disturbance, which is unworkable in practice. However, in some cases, the effort and timescale involved in obtaining a licence seems disproportionate to the actual risk of 'disturbance' due to the scope or the timescale of the works, especially when having a precautionary method of working (PMW) in place could avoid any adverse effects. A PMW is a document that sets out how works will be undertaken where it is considered reasonably unlikely that the proposed works will result in an offence. It is a method statement that details reasonable precautionary measures to be adopted throughout the works to avoid negative effects on otters.

It is the status of the resting site which is crucial in the assessment of whether 'disturbance' will occur, together with a sound understanding of the nature of the proposed works. Standard survey guidance does not provide a methodology for assessing status; surveyors should use their knowledge of otter behaviour to assess the importance of a resting place and thus to determine the likely impact of any proposed development. A summary of guidance for assigning status of otter resting sites, produced internally within Atkins, is provided in Table 2. Once the status of a resting site has been determined, the scale and duration of the proposed works should be considered to determine potential impacts.

Table 3 presents information on sites in Scotland where EPS licences have been obtained to allow works to be undertaken within 30 m of a resting site. The table provides information on the type of resting site (holt, hover or couch) and its status (high, medium or low).

The nine project examples in Table 3 include works done in already highly disturbed locations (urban areas or sites subject to human disturbance, for example in close proximity to road or rail links) and works in remote locations (which are all relatively undisturbed by humans and are situated away from urban conurbations, roads and dwellings). All of the developments were relatively small scale

Table 2: Summary of guidance for assigning status of otter resting sites (Emma Roper, Atkins internal document)

Resting site status	Definition
Low	Feature with limited evidence of otter activity - low number of spraints, not all age classes present. Insufficient seclusion to be a breeding site or key resting site, unlikely to have links to the key otter requirements. Most likely to provide a temporary 'stop off' for otters when moving through their territory. Loss/disturbance of such a feature is unlikely to be significant in terms of the individual or population.
Moderate	Feature containing sprainting with a range of age classes, but not in significant quantities. Availability may be limited by season, tides or flow. Unlikely to be suitable as a breeding/natal site but will be a key resting site and may be linked to other important features within the territory. The impact arising from a loss or disturbance of such a feature will be determined by the availability of more suitable or well used sites within the otter's territory.
High	Feature has a high level of otter activity, including an abundance of sprainting of all age classes, large spraint mounds, well used grooming hollows, paths and slides. Affords a high degree of cover and is linked to key features such as fresh water and abundance of prey. May be suitable as a breeding area (spraints may be absent from natal holts). The site is usually available at all times of year and at high and low tide/ flow. The loss/ disturbance of such as feature will often be considered significant in terms of the individual or population.

but some of them had a long duration. Examples of works include pipeline installation, outfall installation, upgrades to treatment works and rail defence work. None of the schemes involved night working, nor did any of the works directly obstruct the entrance to the resting site or prevent otter passage. None of the schemes resulted in any long-term increases in the presence of humans or in increased levels of noise or other disturbance. Based on the evidence in Table 3, otters continue to use resting sites near development schemes during and immediately following the works period, with similar activity levels to those recorded prior to the works. Further details of three of the case studies are provided below.

Case Study 1: Glasgow Suburbs – Minor Works in an Urban Area

These works comprised an upgrade to an existing outfall to improve the water quality along a burn, which would ultimately enhance the burn for otters, through likely increases in fish. A low status hover, approximately 18 m from the works, provided limited shelter. Interpretation of the field evidence suggested that otters would not be disturbed by the works and that a legal offence under the Habitat Regulations was unlikely given the following factors:

- the hover only offered limited shelter and was not suitable for prolonged day time use;
- the burn formed a peripheral part of an otter's territory and was not regularly patrolled;
- all works would be conducted during daylight hours when otters would not be using the hover and commuting would not be disrupted;
- alternative shelters were present 140 m upstream and 150 m downstream;
- the site was located within a urban area, where otters would already be habituated to a degree of human disturbance; and
- the nature of the works did not require any vegetation removal immediately around the hover and the hover would not be directly affected as works were located 18 m from the structure.

However, following consultation with SNH, it was advised that an EPS licence would be required for the scheme as the hover was within 30 m of the works. The licence included conditions to prevent disturbance such as limiting works to daylight hours, supervision of works by an ecologist and setting up a 10 m exclusion zone around the resting site. Surveys were undertaken prior to, during and following the works to assess the use of the hover. These surveys located a fresh spraint during the construction period and following construction, indicating that otters had continued to use this reach of the burn. This suggests, in the absence of any legal definition of the term disturbance, that otters had not been 'disturbed' by the works or at the least, the disturbance caused by the works was minimal and not measureable.

Case Study 2: River Clyde - Major Excavations in an Urban Area

Even in urban locations, it is expected that otters would be disturbed by major works occurring very close to holts, but evidence has been recorded showing that otters continued to use holts close to major works. At a site along the River Clyde, adjacent to a railway station and heavily used by dog walkers, a holt was located close to proposed pipeline construction works. The year-long works involved major excavations and the use of large machinery within 5 m of the holt (Holt 1 in Photo 1). A second holt (Holt 2 in Photo 1) had been recorded 50 m from the nearest work, which was considered to provide an alternative resting site during the works. It was considered likely that the works would result in an offence under the Habitats Regulations because the noise and high level of human activity could result in the otter avoiding use of this moderate status holt for a year or longer, which was considered to constitute

Table 3. Sites in Scotland where EPS licences have been obtained to allow works to be undertaken within 30 m of a resting site

Location	Type and status of resting site	Distance from works to resting site	Duration of works	Activity levels		
				Pre construction	**During construction**	**Following construction**
Urban sites						
Glasgow suburb	Hover - Low	18 m	11 weeks	Spraints: 1 recent, 3 old	Spraints: 1 fresh and 4 old	Spraints: 1 recent (plus spraints recorded during previous surveys)
Loch Fyne (within 2 m of road)	Holt - High	15 m	1 week	Spraints: 2 fresh, 1 recent, 20 old	Check not undertaken due to short duration of works	Spraints: 4 fresh, 1 recent (plus spraints recorded during pre-construction survey)
Helensburgh	Holt - Moderate	5 m	1 year	Spraints: 1 fresh, 3 recent and 2 old	Visit 1: 2 recent Visit 2: 2 fresh, 2 recent, 4 old	Spraints: 1 fresh tar spot and 10 old
Remote sites						
Isle of Lewis	Holt -Moderate	15 m	2 weeks	Spraints: 10 fresh	Spraints: 1 very fresh	Spraints: 9 recent and 1 fresh
Scottish Boarders	Holt - High Hover - Moderate	10 m 20 m	1 week	Spraints: 9 old Very fresh spraint	Check not undertaken due to short duration of works	Spraints: 1 fresh Spraints: 4 old (as recorded during post construction) Abundant fresh and recent spraint was record along the river between the two resting sites
Inverclyde	Couch - High Couch - High	40 m	6 weeks	Spraints: 6 old Spraints: 4 fresh, 4 recent and 3 old	Spraints: 1 recent, 6 old As pre-construction with fresh feeding remains	None Spraints: 1 recent High levels of fresh and recent feeding remains recorded around water body
Kilmelford	Works adjacent to important commuting corridor. Licence for a holt within 15 m of works, but this naturally slumped prior to start of construction	15 m	37 weeks	Spraints: 1 old Large amount of sprainting round loch	Very fresh spraint	Spraints: 1 very fresh and 1 old
Howdon	low status, open air resting site and grooming hollow	30 m	8 weeks	No evidence – heavy rainfall had washed out much of the site	Not undertaken	Otters seen by site operatives, footprints were recorded
Stromeferry	Holt - low Couches - high	10 m	23 weeks	Not surveyed	Regular sighting of otter reported during construction period	Fresh and recent spraint recorded

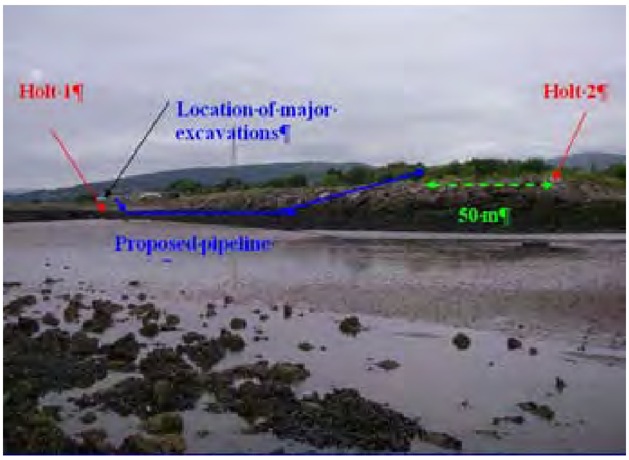

Photo 1. Case study 2 - works and holts along the River Clyde

disturbance. The EPS licence ensured that an exclusion zone was set up around Holt 1 and a tool box talk was provided to the site operatives to make them aware of the potential effects on otters.

Surveys undertaken during the construction period recorded fresh spraints on more than one occasion within Holt 1, along with a strong characteristic smell of otter. Despite having a similar holt available further away from the works, the otter chose to use its favoured resting site. Given the level of otter activity recorded it is considered that the otter was undisturbed by the works or at the least, the disturbance caused by the works were minimal and not measureable.

Case Study 3: Isle of Lewis – Minor Works in a Remote Area

It is not just in urban locations that otters have been found to be resilient to disturbance. For example, a site in a remote area of the Isle of Lewis, where a new section of waste water pipeline was required across a remote peninsula (approximately one hour's walk to the nearest road!). Here a vast amount of otter evidence was recorded, along with various holt structures across the peninsula. Given the remoteness of the area, it was considered that any otters present would be highly sensitive to disturbance, especially as a suspected natal holt was recorded within 200 m of the new pipeline route. The nearest structure to the works, a moderate status peat burrow holt was located within 15 m of the new pipeline. The assessment considered that an offence would occur as a result of the works, as the otter was likely to be disturbed by the presence of humans at this remote site and would avoid using the resting site. An EPS licence was granted to allow the pipeline installation to take place in proximity to this holt. Immediately prior to the works a large amount of fresh sprainting was recorded at the burrow (approximately 10 spraints). The works were supervised by an ecologist, with a tool box talk given to the contractors to make them aware of the site sensitivities and an exclusion zone was established around the resting site. A fresh spraint was recorded at the holt during the construction works.

Following the works, survey was undertaken; this found nine recent and fresh spraints at the entrance to the peat burrow demonstrating that otters were still highly active in the area despite the recent human disturbance. The evidence found during and following the construction works demonstrates that otters will continue to use favoured sites during and after works. This indicates that in this case at least, it was not adversely affected by short term human disturbance; in the absence of a legal definition, this suggests that otters were not 'disturbed' by the works or at the least, the disturbance was minimal.

Photo 2. Case study 3 - works on a remote peninsula of the Isle of Lewis

Summary

The case studies described above and in Table 3 indicate that otters often continue to use resting sites and watercourses throughout the duration of works, even works involving major excavations and over a long period of time. This indicates that, over the monitoring period, the local distribution of otters has not been significantly affected by the works. In terms of the other offences, the case studies show that, at worst, the effects of 'disturbance' were minimal and not significant and, at best, that otters were not 'disturbed'. These case studies can be set in the context of the EC (2007) guidance that sporadic disturbances without any likely negative impact on the species should not be considered as disturbance under Article 12. This leads on to the suggestion that if there are no likely negative impacts from specific works associated with development on the species, such works should not be considered as disturbance under the legislation. Longer-term monitoring studies are required to determine whether the works may have impaired the ability of an otter to successfully breed or have affected the abundance of the species.

It is clear that EPS licences are essential wherever an offence might occur, to ensure an appropriate degree of control over the construction process and protect the developer from potential prosecution. The licensing system and the agreed zones simplify the process of assessment and remove the need for lengthy debate about what does and does not constitute a disturbance offence. However, in some cases, the effort and timescale involved in obtaining a licence seems disproportionate to the actual risk of 'disturbance' due to the scope or the timescale of the works, especially when having a precautionary method of working in place could avoid any adverse effects. The case studies above suggest there may be some projects where a precautionary method of working rather than a licence is a more pragmatic approach to take. Such a decision should be taken following discussions between the Statutory Nature Conservation Organisation (SNCO) and a specialist otter ecologist, in the light of experiences such as those documented here. Longer-term monitoring would be useful, to check that populations are still present some years after construction. Collation of that information to get a country-wide picture, would allow such decisions to be made with a greater degree of confidence.

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Use of badger tunnels by mammals on Highways Agency schemes in England

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Use of badger tunnels by mammals on Highways Agency schemes in England

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SUMMARY

Monitoring of badger tunnels using clay mats on nine Highways Agency road schemes was undertaken to establish their effectiveness in terms of use by large mammals (primarily European badger *Meles meles*), as well as the efficacy of tunnel design advice provided by the agency. The results indicate that tunnels are an effective means of mitigating the effects of all types of new road schemes on badgers; 89% of the tunnels monitored were used. The results suggest that no one factor is of over-riding importance in tunnel design; however, design features that appeared to be associated with the use of tunnels were good vegetation cover, habitat connectivity, good drainage and a tunnel width of at least 600 mm.

BACKGROUND

In the UK, the Highways Agency routinely evaluates the efficacy of conservation initiatives to assess the extent to which objectives have been met. In 2010, the Post Opening Project Evaluation, identified badger tunnels (culverts installed under roads to allow safe passage) as an intervention that merited further investigation to establish effectiveness in terms of use by large mammals (primarily European badger *Meles meles*), as well as the efficacy of tunnel design advice provided within the Highway Agency's *Design Manual for Roads and Bridges* (DMRB; HA 2001).

The primary reason for incorporating tunnels beneath highways during construction is to reduce habitat fragmentation impacts on mammals, and to minimise the risk of road traffic accidents caused by animals attempting to cross a road. When installing badger tunnels, the Highway Agency's manual recommends a number of design features to guide mammals through culverts or overpasses to prevent them from directly crossing a road. Badger tunnels should be made using 600 mm diameter concrete pipes. Appropriate landscape planting should be carried out to soften the approach to the tunnel, while fencing should be installed to direct mammals to the tunnel entrance and prevent them from accessing the road. The location of badger crossings is crucial to success; it is preferable if a crossing can be located on, or as near as possible to, the site of an active badger path. The manual does not provide guidance on optimal length of a badger tunnel (DMRB; HA 2001). As badgers are one of the species most commonly killed on roads in the UK, this study focuses primarily upon the use of tunnels by badgers, although other mammal species are also considered.

The monitoring method used in this study follows that developed by Baker, Knowles & Latham (2007). This involved using clay mats to record the imprint of mammal tracks; a simple and low-cost technique (Figure 1). The present study had two aims, firstly to establish whether badgers use crossings, and secondly, to identify any specific problems or factors associated with tunnel design that reduce or increase the likelihood of use by mammals.

Figure 1. A clay mat installed at the entrance of a badger tunnel to monitor use by mammals.

ACTION

Study sites: Nine major road schemes (dual carriageway or motorway) throughout England were chosen for study. These roads incorporated 38 mammal tunnels installed between 2003 and 2007 (Table 1). The tunnels varied in design in terms of materials, width and length. The 38 tunnels included both concrete and corrugated iron tunnels and were an average length of 44 m (minimum 20 m, maximum 120 m). Tunnel diameters were 300 mm, 450 mm, 600 mm, 700 mm or 1,000 mm, with the modal diameter being 600 mm (the 'standard' width). (Note, diameters given in mm as per industry standard).

Monitoring: Monitoring was undertaken from 24 August to 26 October 2010. The autumn was chosen as a suitable time (as in the 2007 study) as it was considered that the clay mats would remain moist and soft enough to record mammal footprints over about a week's duration (interval between each visit). It is also the time when mammal activity tends to be high (post-breeding dispersal of young animals). At each study tunnel, a clay mat (45 x 45 cm x 0.5 cm thick) was placed just inside the tunnel entrance in late August. Tunnel design (diameter, construction material, visibility of light through the concrete pipe), the condition of the tunnel and associated fencing were recorded. The amount of vegetation cover around the tunnel entrance and habitat connectivity was assessed, describing how the tunnel entrance linked to adjacent habitat features such as hedges and highway (roadside) planting.

Any evidence of animal tracks was recorded and species identified (Bullion, Strachan & Troughton 2001). The clay mat was then thoroughly wetted and smoothed over, leaving a clean surface to record future tracks. In addition to clay mats, passive infra-red motion activated cameras were set up at two tunnel entrances (A5 Nescliffe Bypass and A590 High and Low Newton Bypass) for one week to further assess suitability of monitoring using clay mats and to highlight any unexpected limitations associated with this technique.

Table 1. Road scheme, tunnel numbers and known design issues potentially affecting use by badgers.

Scheme name and location	Scheme type and length	Mammal tunnels	Known design issues
A590 High and Low Newton Bypass, Cumbria, northwest England	3.8 km 2-lane dual carriageway	4 badger tunnels; 1 badger/otter tunnel	1 tunnel 450 mm width i.e. less than standard 600 mm
A66 Temple Sowerby (northwest England)	5 km 2-lane dual carriageway	1 badger tunnel	Tunnel longer than average at 60 m
A1(M) Wetherby to Walshford, North Yorkshire, northeast England	5.3 km 3-lane motorway	1 badger tunnel	Tunnel longer than average at 60 m and with plank bridge crossing to access tunnel
A63 Selby Bypass, North Yorkshire, northeast England	10 km single carriageway	3 badger tunnels	1 tunnel 300 mm i.e. less than standard width
A5 Nesscliffe Bypass, West Midlands, central England	4.5 km 2-lane dual carriageway	4 badger tunnels	2 tunnels larger than standard width (i.e. 700 mm and 1,000 mm); 1 tunnel longer than average at 70 m
A6 Rothwell Bypass, East Midlands, central England	6 km single carriageway	13 badger tunnels	Some tunnels deep beneath carriageway, therefore possible restricted air flow; close to public footpaths
A428 Caxton to Hardwick, Cambridgeshire, eastern England	7.7. km 2-lane dual carriageway	4 badger tunnels	1 tunnel with poor drainage
A120 Stansted to Braintree, Essex, southeast England	14 km 2-lane dual carriageway	6 badger tunnels	3 tunnels longer than average at around 70 m
A34/M34 Chieveley Junction South, Berkshire, southern England	Junction	1 badger tunnel	Tunnel longer than average at 120 m

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CONSEQUENCES

Mammal use: Overall, 35 of the 38 tunnels (92%) were used by large mammals, with 89% used by badgers during the autumn 2010 monitoring period. Species recorded were badger, Eurasian otter *Lutra lutra*, red fox *Vulpes vulpes*, European hedgehog *Erinaceus europaeus*, brown rat *Rattus rattus*, domestic cat *Felis catus* and domestic dog *Canis lupus familiaris*. Use of the tunnels by badgers was greater than any other species. In terms of the regularity of use, 37% of the tunnels were used frequently by badgers (i.e. footprints recorded on 7 or 8 of the monitoring visits), 29% showed moderate levels of use (i.e. prints recorded on 4-6 monitoring visits) and 23% were used infrequently (i.e. prints recorded on only 1-3 monitoring visits). Figure 2 shows prints on one of the clay mats. These results indicate that the tunnels installed under both dual carriageways and motorways are being used on a regular basis.

Figure 2. Badger prints on a clay mat within a badger tunnel.

Tunnel design: The results emphasise the importance of some elements of tunnel design that may encourage use by badgers. The key features that appear to be associated with more frequent use were:

1) Good habitat connectivity with existing landscape features such as hedges and ditches. Figure 3 suggests that good and moderate habitat connectivity is more likely to result in a tunnel being used than those with poor connectivity to such features;

2) Good vegetation cover around the tunnel entrance. Figure 4 indicates more frequent use by badgers of tunnels with good cover;

3) Good drainage; tunnels with poor drainage were never or infrequently used;

4) A tunnel width of at least 600 mm. Tunnels wider than 600 mm were regularly used. The two tunnels of 300 mm and 450 mm were never used or infrequently used by badgers (it is acknowledged that the small sample size precludes a definitive conclusion).

It was found that use of the tunnels was not significantly influenced by tunnel construction material (concrete or corrugated steel), whether light was visible through the tunnel, or tunnel length. The lack of a relationship with tunnel length Figure 5 was surprising (there is anecdotal evidence that badgers tend not to use long tunnels, particularly if light is not visible through the tunnel).

Effectiveness of clay mats as a monitoring method: The clay mats were effective as a means of monitoring mammal tracks. The technique does however have limitations, which include drying out and cracking in hot weather, or water logging in wet conditions. Where water logging occurred there was evidence that badgers tried to avoid walking on the mats (partial prints on mat edges suggested that badgers had tried to walk around them). A few simple measures could be taken to reduce these limitations, such as the use of larger clay mats (thus animals cannot pass without treading on them, placing mats further in the tunnel entrance (out of direct sunlight or precipitation), and more regular monitoring and maintenance (e.g. every 3 to 5 days instead of every 7 days).

The use of motion-activated cameras at two sites did not pick-up any additional species to those identified by the clay mats.

Other observations: A number of other interesting observations were made. In some tunnels, prints were sometimes recorded in one direction only. This suggests that badgers use tunnels to access feeding grounds, subsidiary or outlier setts; consequently, they may not return the same night (or for several nights). Alternatively, badgers may be using other means of returning, such as other tunnels or bridges, or traversing directly over the road. At three locations (on the A5 and A6), badgers had pulled bedding into the tunnels. This suggests they use tunnels as resting sites as well as underpasses. In one case, the tunnel was blocked at one end by a large boulder making it impassable to badgers, but prints were recorded regularly at the open end of the tunnel and badgers appeared to be using the tunnel as a sett.

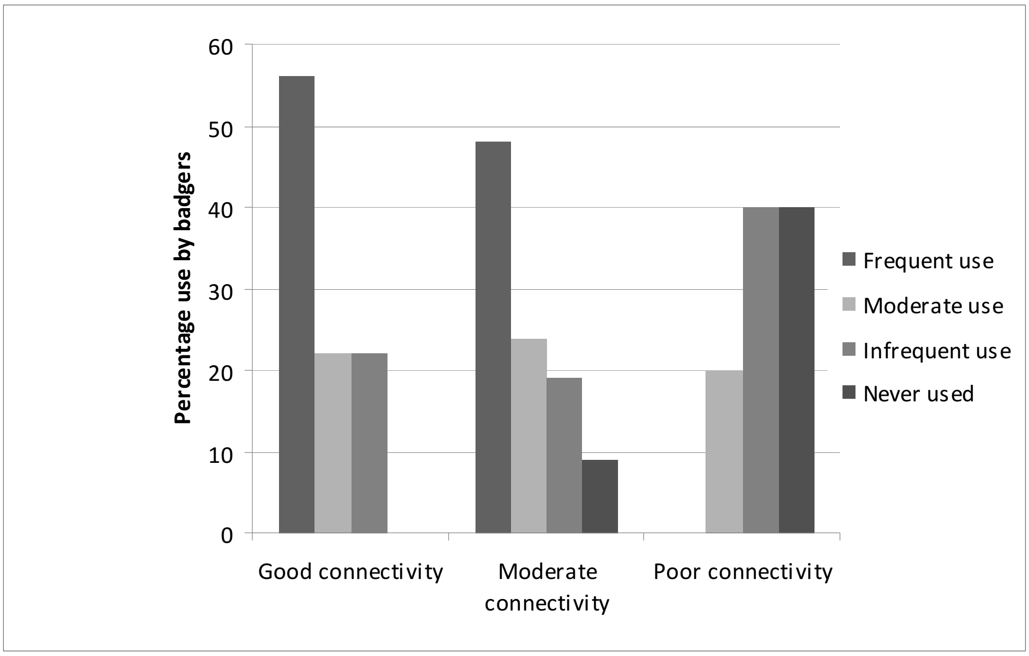

Figure 3. Tunnel use by badgers in relation to habitat connectivity.

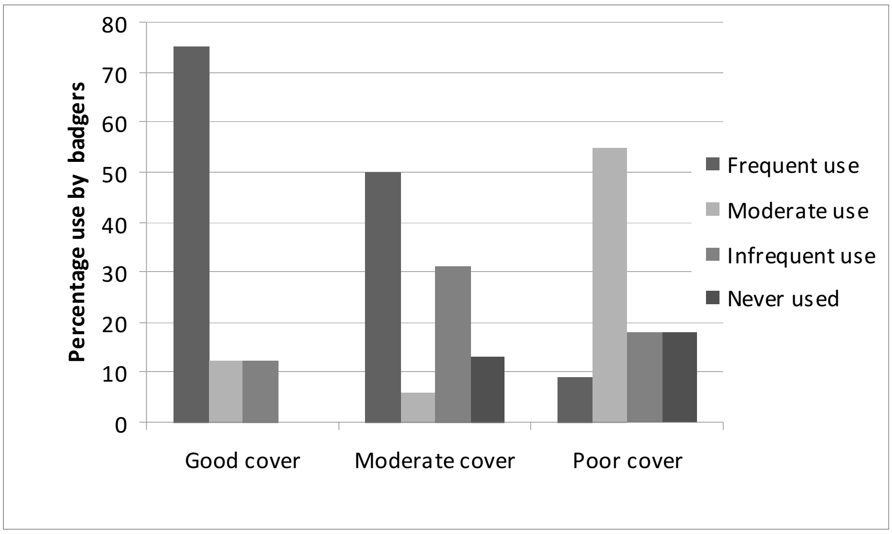

Figure 4. Tunnel use by badgers in relation to vegetation cover at tunnel entrance.

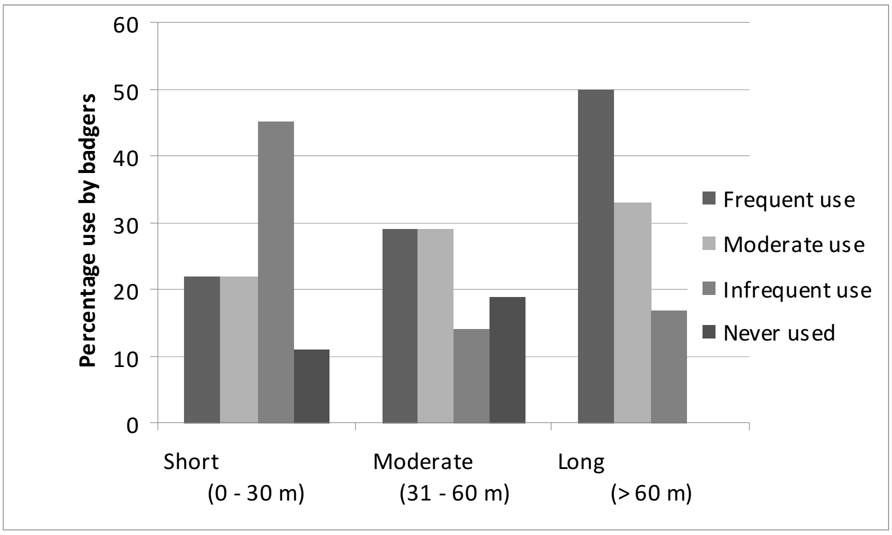

Figure 5. Tunnel use by badgers in relation to tunnel length.

Conclusions: The results indicate that badger tunnels installed under the study roads help mitigate the effects of habitat fragmentation resulting from new road developments. Tunnels provided safe passage under the roads for several large mammal species, particularly badgers; 89% of tunnels monitored were used by badgers and 92% were used by a wider range of large mammals. Clay mats were an efficient and low-cost means of monitoring mammal use of tunnels. It is acknowledged that this method has some limitations such as mats drying out in hot weather thus becoming too hard to record tracks but regular maintenance should avoid such problems, and they cannot be used to assess actual numbers of individual crossings through a tunnel over a given time period.

In terms of enhancing use by badgers, good tunnel design should incorporate adequate drainage and the tunnel width should be 600 mm (results suggest that a tunnel of smaller width is less likely to be used). Tunnels should ideally be located where existing habitat connectivity is good, and with vegetation providing some cover around tunnel entrances in order to increase their suitability for use by mammals as road crossing structures.

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Catherine Sellars

Habitat Suitability Index Scores as an Indicator of the Presence of Great Crested Newt

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Habitat Suitability Index Scores as an Indicator of the Presence of Great Crested Newts

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Introduction

There is abundant anecdotal evidence from ecologists suggesting that great crested newts (GCNs) are often found in water bodies that appear to be unsuitable or have low potential to support this species. The purpose of this study is to determine whether Habitat Suitability Index (HSI) scores are reliable in predicting the presence (and absence) of GCNs in water bodies and whether HSI scores are accurate enough to allow ecological consultants to use this information with confidence. This has been achieved through comparing HSI scores of water bodies with the results of presence/absence surveys, using data from Atkins surveys of 67 water bodies undertaken in 2009.

The GCN HSI was developed by Oldham *et al.* (2000) to evaluate habitat quality of a water body. It is a quantitative measure and is recommended by Natural England as a potentially useful tool in GCN surveys and mitigation (Natural England 2010).

HSI is a number from 0 to 1 based on an assessment of 10 habitat variables known to influence the presence of GCNs. For each water body this includes an assessment of 10 characteristics of the water body: geographical location, surface area, desiccation rate, water quality, amount of shade, number of waterfowl, presence of fish, density of water bodies in the area, quality of terrestrial habitat and the macrophyte (vegetation) cover.

An HSI score of 1 indicates optimal habitat (*i.e.* a high probability of GCNs being present in a water body). An HSI score of 0 indicates unsuitable habitat for GCNs. There are five HSI water body suitability categories: excellent (scores greater than 0.8), good (scores between 0.7 and 0.79), average (scores between 0.6 and 0.69), below average (scores between 0.5 and 0.59) and poor (scores below 0.5).

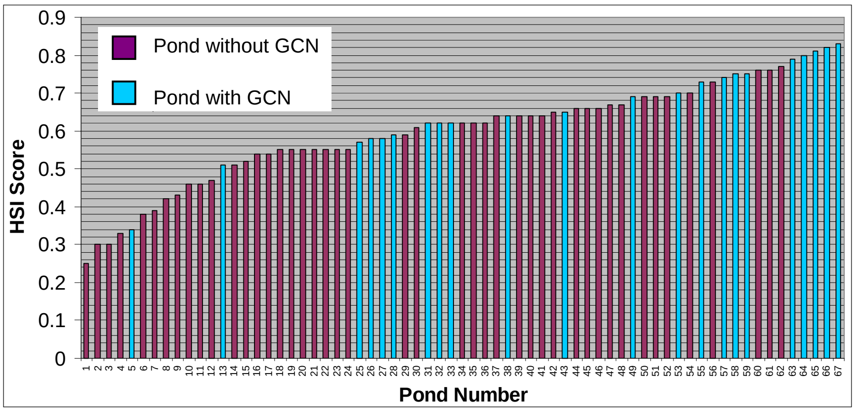

Figure 1. Comparison of HSI scores for ponds with and without great crested newts

Natural England's guidance (Natural England 2010) notes that the use of HSI scores is not a substitute for newt surveys. However, this guidance suggests that if a water body has an HSI less than 0.5 (a poor potential to support GCNs) that with due care, and in limited circumstances, the HSI might be used in the absence of newt presence/absence surveys to help conclude that an offence is highly unlikely and that therefore work could proceed in that area without a GCN licence. However, in these circumstances Natural England urge that reasonable precautions be taken to avoid affecting GCNs during works (Natural England 2009). A reasonable precaution may include the use of a Precautionary Method of Working (PMW). A PMW is a document which sets out how works will be completed where the presence of GCNs is possible but considered to be highly unlikely. It is a method statement which details the precautionary approach to be taken throughout the works to ensure that, should a GCN be found, it will not be harmed. Methods of working can include an ecologist supervising vegetation clearance and carrying out destructive hand searches. There are, however, risks to this approach; if GCNs are found subsequently, it may lead to delays to the works. It is therefore imperative that the basis on which any such judgements are made is robust.

Methodology

A total of 67 water bodies have been included in this study from around England. The water bodies, in the majority, were located in rural areas in the South East and the Midlands. These water bodies have all been subject to HSI assessment and GCN presence/absence surveys (which conformed to Natural England guidelines) by Atkins ecologists between March and June 2009.

Results

A total of 22 of the 67 water bodies were found to support GCNs.

The likelihood of water bodies supporting GCNs rises as the HSI score increases (Figure 1). This indicates that HSI is generally reliable for predicting the suitability of water bodies for GCNs.

Table 1 shows there is one water body in the 'poor' HSI category which supports GCNs (with an HSI score of 0.34). This pond has a small population of GCNs.

There is also a cluster of water bodies in the 'below average' and 'average' HSI categories that support GCNs. One of these water bodies (Pond 38 with an 'average' HSI of 0.64) has the largest GCN population of the 22 water bodies with newts present (with a maximum of 43 being counted on one night). This data suggests that the anecdotal evidence of GCNs occurring in lower quality habitats is correct. The average HSI score for water bodies supporting GCNs is 0.67 and the average HSI score for water bodies that have been found not to support GCNs is 0.57.

Table 1. Percentage of water bodies with GCN population for each HSI category in Atkins 2009 dataset

HSI Category	No. of Water Bodies with GCNs	No. of Water Bodies Without GCNs	% of Water Bodies Supporting GCNs
Poor	1	11	8%
Below average	5	12	29%
Average	6	17	26%
Good	6	5	55%
Excellent	4	4	100%

Figure 2 compares the proportion of water bodies in each HSI category that support GCNs in the current Atkins dataset (based on 22 out of 67 water bodies) and the expected proportions from the Oldham *et al.* (2000) study (NARRS). This figure shows that the higher the HSI category, the more likely it is to support GCNs (*e.g.* with 100% of water bodies in the 'excellent' category in the Atkins study supporting this species) and conversely, the lower the HSI category for a water body the less likely it is to support GCNs (*e.g.* with only 8% of water bodies in the 'poor' category supporting this species).

Figure 2 shows that the survey results used in this study do not correlate precisely to those of the Oldham *et al.* study. This is particularly the case in the 'poor' and 'below average' categories where the results show a greater percentage of water bodies supporting GCN populations than the Oldham *et al.* study. The lack of a clear relationship in the 'below average' to 'good' categories could indicate either that the HSI does not take these lower quality habitats into account as much as it should or that recording the characteristics of water bodies in the mid-range of quality is more problematic. The assessment of certain pond characteristics using HSI relies on subjective assessments rather than objective measurements (*e.g.* dessication rate and number of water fowl). As such the accuracy of recording these relatively complex and numerous characteristics, particularly for water bodies in the mid-range of quality, may contribute to the HSI for these ponds being under-estimated.

Figure 2 also shows that the percentage of water bodies in the 'average' and 'good' HSI categories are less than the guidance suggests. However, the 'excellent' category is very similar (with all four water bodies in the Atkins sample supporting GCNs). In any case, water bodies with these HSI scores in these categories are likely to be subject to presence/absence surveys and as such present less of a risk to any proposed works.

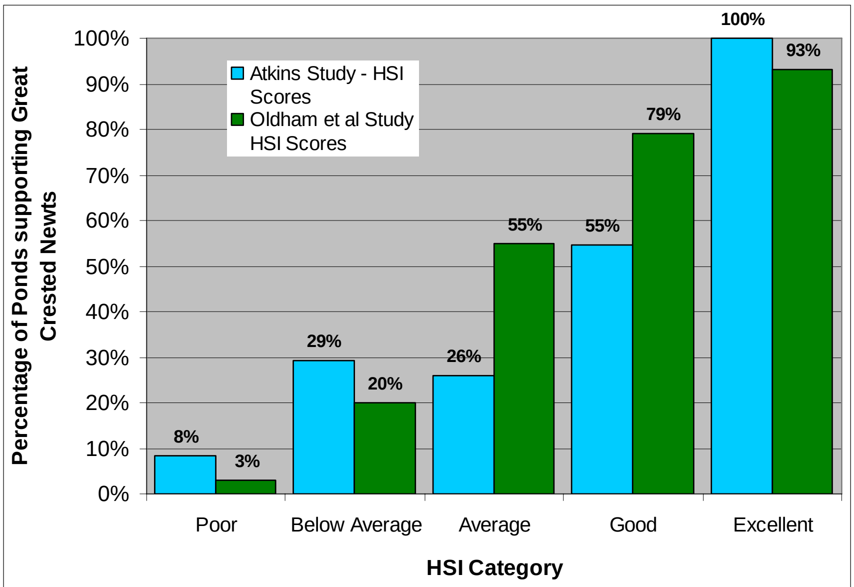

Figure 2. Comparison of Proportion of Ponds Supporting Great Crested Newts from Atkins Data and Existing Guidance

In general, the Atkins data suggests that it is unlikely that GCNs will be found in water bodies that have an HSI score of less than 0.5. It also suggests that GCNs are highly likely to be present in water bodies with scores above 0.8. However, for scores between these two values (*i.e.* in the 'below average' to 'good' categories) there is no clear relationship between HSI scores and the presence of GCNs, confirming the anecdotal evidence of the presence of GCNs in water bodies of apparently low quality. In the sample used for this assessment, GCNs were recorded in 26% to 55% of water bodies in these HSI categories.

Practical Applications

This study has found that HSI scores are relatively reliable in assessing the suitability of good quality water bodies to support GCNs. However, this study has also noted that there are exceptions to the reliability of the score, and that some water bodies do not appear to fit into the predicted pattern.

For ecologists this is particularly important when considering water bodies with 'poor' or 'below average' HSI scores. Ecologists frequently have to make decisions about whether to undertake presence/absence surveys or whether the works could be completed under a PMW. The HSI score plays a large part in this decision making process. If a water body has a 'poor' HSI score it is possible that ecologists will use this as justification that a PMW is a suitable approach (in accordance with Natural England 2010). If this approach is taken without consideration of other factors in relation to the proposed works (*e.g.* size and scale of the proposed works and the potential of the terrestrial habitats to be affected to support GCNs) this could result in significant programme delays and the associated additional unforeseen costs for the client if GCNs are subsequently found (*e.g.* carrying out presence/absence surveys and applying for a development licence from Natural England).

A water body with a 'poor' HSI score does not necessarily mean that GCNs will not be present. The results of this study suggest that the likelihood of finding GCNs in water bodies with low suitability, *i.e.* a 'poor' HSI score, may be greater than the guidance sets out (8% in this study as opposed to 3% in the Oldham *et al.* (2000) study), although it is acknowledged that this is based on one water body with GCNs, out of 12 in the 'poor' category in the Atkins study. This is something that ecologists should be aware of as even water bodies with very low scores have been found to support GCNs (see Photograph 1). If

Photograph 1. Water body with a 'poor' HSI score which supports GCNs

the potential impacts on the pond and associated terrestrial vegetation are high, it would present less of a risk to undertake surveys instead of relying on a PMW.

Ecologists need to consider carefully how to apply HSI scores, particularly relating to water bodies with a 'poor' or 'below average' score. For this study the average HSI score for water bodies supporting GCNs is 0.67 (in the 'average' HSI category) and the average HSI score for water bodies that have been found not to support GCNs is 0.57 (in the 'below average' HSI category). There is not a large difference between these scores and ecologists should use the HSI system with care, also taking into account the location, scale and type of the proposed works to be carried out in the vicinity of the water body. The use of the Natural England Rapid Risk Assessment (Natural England 2010) is a useful tool in this decision making process.

One way to minimise risks when determining whether presence/absence surveys are required or if a PMW is an appropriate approach is to relate each of the potential impacts of the proposed works to each of the offences relating to the protection of GCNs listed in Regulation 41 of the Conservation of Habitats and Species Regulations 2010. This will ensure that the effects of the proposed works have been fully assessed and that PMWs are written in such a way that the wording is strong enough and clear enough to ensure that all aspects of the proposed works and potential impacts on GCNs have been considered. Generally speaking this is the approach taken by Atkins when producing PMWs.

We are continuing to analyse HSI scores and the results of GCN presence/absence surveys (particularly for water bodies with lower HSI scores). The relationship of water bodies with low HSI scores (that are found to support GCNs) with water bodies up to 500 m away is also being investigated. This is to assess whether GCNs are only present in ponds with a low HSI when there are ponds with a high HSI score in the vicinity (suggesting that they help to support meta-populations of this species rather than support a population in their own right).

Acknowledgements

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Newts prove no bar to operations

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Newts prove no bar to operations

Co-operation between a site operator and ecology experts has ensured that mineral extraction can continue at a quarry in Cheshire alongside a growing population of great crested newts, *Luke Gorman* explains

With ecological legislation and planning conditions putting ever more emphasis on the preservation and enhancement of biodiversity, operators of mineral extraction sites increasingly have to consider the implications of their work processes and restoration plans on the environment.

This priority can be viewed as a constraint on site working. However, if it is managed appropriately, it need not be costly and does not have to delay or negatively affect the extraction process. This has been demonstrated at Crown Farm Quarry near Sandiway in Cheshire. Tarmac owns and operates the sand extraction quarry, which has planning permission to continue extraction until 2014. Atkins has acted as ecological consultants at Crown Farm Quarry since 2003.

A number of ponds, pools and ditches have been created at the quarry as a result of historic quarrying activites and an unexpected rise in the water table. The dynamic nature of mineral extraction sites often leads to the the creation of diverse habitats that provide excellent conditions for a range of species. Water bodies created by mineral extraction often provide an aquatic environment for amphibians and can provide habitat suitable for other species such as birds and invertebrates.

In 2003, great crested newts were discovered in some of the waterbodies in the operational part of the quarry following an environmental impact assessment to support planning permission to extend mineral extraction at the site. Great crested newts are mobile and will colonise new sites that provide suitable conditions. They are a European protected species and they and their habitats are subject to full legal protection.

No negative impact from ongoing works

Once the newts' presence on site was confirmed, it was necessary to apply for a development licence to Natural England to allow continuation of mineral extraction at the site. Such licences permit works to be undertaken in accordance with a method statement agreed by the ecological consultant, the developer and Natural England. These method statements detail methods of working that must be adhered to in order to protect individual newts and their habitats.

No mineral extraction or other quarrying activities were expected within areas providing the best-quality terrestrial habitat for great crested newts. Vehicles already used well-established hardstanding tracks which are inhospitable to them. In addition, since a great crested newt population had naturally established itself on an operational mineral quarry, we could conclude that the ongoing works were not having a negative impact on the population.

A licence was obtained on the basis that areas of exisiting good-quality terrestrial and aquatic habitat would be protected, hand searches for great crested newts would be carried out whenever work processes required disturbance of terrestrial habitats suitable for amphibians, new refuges would be created and one member of the Tarmac site staff would be trained to deal with any newts found on-site. This approach protects the most suitable habitats and enables the efficient relocation of any newts found within work areas to newly-created refuge areas.

Tarmac has put habitat improvement measures in place to ensure that the newt population is maintained. These include two terrestrial refuges near existing breeding ponds, using discarded rubble from the site. The refuges were built by site operatives supervised by an ecologist and took a couple of hours to complete, minimising the cost implications. These refuges provide safe additional terrestrial habitat that can be used for shelter and hibernation, while reducing the need for newts to disperse across the quarry and into operational zones.

Habitat development regularly monitored

A condition of the licence is that a survey to monitor the population of great crested newts is required every two years to ensure that the mitigation measures are working effectively and the population is not being negatively affected by the ongoing quarrying activities. The maximum length of the development licence at Crown Farm Quarry is limited to two years. The results of the surveys need to be submitted to Natural England in order to support applications for extensions to the licence.

Newt population size assessments have been undertaken at Crown Farm Quarry in 2003, 2005, 2007 and 2009. These assessments provide an insight into the changing dynamics of the population during this time, both in response to mitigation measures protecting them at site level and to the natural succession of vegetation in the water bodies. This has reduced the suitability of some of the water bodies for breeding newts

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Crown Farm Quarry: a number of ponds, pools and ditches have been created at the quarry

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Great crested newts: habitats subject to legal protection requiring regular monitoring

but improved the suitability of others.

The 2003 survey indicated that a small population of great crested newts was present at the site. Following the mitigation works, the 2005 survey showed that the population was being maintained and a further survey in 2007 indicated that a medium-sized population was present. The 2009 survey recorded a substantially larger population than any of the previous surveys, although it would still be classified as a medium-sized population by Natural England.

The population assessments at Crown Farm Quarry have shown that the great crested newt population has been maintained and may be expanding. Their expansion contributes towards both national and Cheshire Biodiversity Action Plan targets which aim to ensure that there are no losses to the post-1990 range of great crested newts, increase the number of suitable ponds, increase the number of ponds occupied and to achieve an increase in the newts' range.

In the winter of 2009 Tarmac restored a first water body at Crown Farm Quarry. The linear ditch provides important breeding habitat and an important link between great crested newt breeding ponds in the north and south of the site. Over the years, this ditch had become choked with bulrush and areas had dried out. The restoration of this ditch was undertaken during winter to reduce the chance of newts being present in the ditch, as they generally hibernate in terrestrial habitats.

A mini-digger was used to remove bulrushes and restore dry areas to open water. This operation created approximately 150m^2 of additional aquatic breeding habitat for the newts while also improving aquatic dispersal opportunities within the quarry. The costs amounted to fuel for the mini-digger and the time of one site operative to man the machine. Such relatively simple habitat management shows how meeting biodiversity targets and managing habitat for protected species need not be costly or time consuming. Improvement measures can be worked into the day-to-day running of the quarry.

Early consultation provides benefits

Tarmac aims to restore or improve at least one water body within Crown Farm Quarry each year in order to maintain the great crested newt population at the site and meet biodiversity targets. When newts are present on a site, planners and quarry operators may face difficulties at the planning stage of extraction operations. It needs to be demonstrated that the site can be worked efficiently without adverse effects on the newt population and enhancements promised at the application stage can be delivered.

Our experience at Crown Farm Quarry has confirmed that the early involvement of ecologists, a positive approach from management and innovative mitigation measures mean that extraction operations have been able to continue without delay. The aquatic and terrestrial habitats for the newts on site have been improved and allowed for the expansion of the great crested newt population with minimum expenditure, enabling Tarmac to contribute towards local and national biodiversity targets.■

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Mitigation: fencing ruled out

The great crested newt population at Crown Farm Quarry is small but widespread in part of the site that had already been worked out adjacent to the mineral processing area. Conventional approaches to mitigation usually result in long stretches of temporary or permanent newt exclusion fencing, which requires constant inspection and maintence to guarantee its integrity as a barrier to their movement.

It would have been extremely difficult to segregate working areas containing no newts from non-working areas with newt populations through exclusion fencing, for four main reasons. Firstly, for the fencing to be effective in protecting newts, its location would have significantly disrupted quarry operations. Secondly, it would have isolated the newt population within the site and reduced dispersal opportunities. This could have negatively affected the conservation status of the newt population at the site.

Thirdly, the distribution of newts across Crown Farm Quarry and the intricacies of the quarry workings, trackways, crossing channels and storage areas adjacent to breeding pools, fluctuating water levels and the extent of standing water would have raised serious issues in maintaining the integrity of the newt fences. Finally, the cost of extensive newt fencing would have been substantial.

An innovative approach to mitigation for newts needed to be devised which would resolve these constraints. Atkins devised a strategy that avoids the use of newt exclusion fencing yet still prevents incidential injury or fatality to newts and disturbance to their habitat while allowing them to disperse freely across the site and outside the site boundary.

P O'Donoghue, L Dolan, P D Dansie and I Sharkey

Can Non-Intrusive Geo-Physical Techniques Assist in Mapping Setts of the Eurasian Badger?

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Can Non-Intrusive Geo-Physical Techniques Assist in Mapping Setts of the Eurasian Badger?

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Badgers and Roads

Atkins (Ecology) in Ireland is providing ecological design advice to the main contractor on the N7 Nenagh to Castletown road scheme. This involves advising on ecological matters relating to road construction, including the potential direct and indirect impact of the proposed road corridor on Eurasian badgers *Meles meles*. The N7 road scheme runs for approximately 35 km through an area of predominantly lowland agricultural grassland in Counties Tipperary, Laois and Offaly, Ireland.

Road impacts on badgers, in particular through disturbance, habitat fragmentation and road mortality are well documented (Harris *et al.* 1994, Forman *et al.* 1995, Roger *et al.* 1997, Clarke *et al.* 1998). Therefore, as per standard best practice (NRA 2006, HA 2001, *etc.*) pre-construction badger surveys were undertaken along the length of the scheme in 2004 (Hyder McCarthy Consultants 2005) and again in 2006 (Flynn Furney Environmental Consultants 2006). A follow-up pre-construction validation survey was also undertaken in 2008 to update the findings of previous surveys (Flynn Furney Environmental Consultants 2008). These studies identified a number of badger setts located inside and just outside the lands made available (LMA) for proposed construction and this informed the design of standard badger mitigation measures including mammal underpasses and mammal resistant fencing and badger gates [as per the *Design Manual for Roads and Bridges* (DMRB) (HA 2001) and National Roads Authority (NRA) Guidelines (NRA 2006)].

Badgers are protected under Irish law by the Irish Wildlife Act 1976 (as amended in 2000[1]). It is an offence under the Wildlife Act to intentionally kill or injure protected species or to wilfully interfere with or destroy the resting or breeding place of a protected animal. In assessing the risk associated with construction, the National Roads Authority has prepared guidelines (NRA 2006) for site works in the vicinity of badger setts. This guidance states that badger sett tunnels may extend up to c. 20 m from sett entrances and recommends that no heavy machinery be used within 30 m of a badger sett unless carried out under licence[2]. Furthermore it recommends that lighter machinery should not be used within 20 m of a sett entrance and that only light work, such as hand digging or scrub removal, should be undertaken within 10 m of sett entrances (NRA 2006). The DMRB considers c. 50 m (HA 2001) from a sett to be safe for machinery.

Those sett complexes located entirely within the LMA were excluded under licence [from National Parks and Wildlife Service; Department of Environment (NPWS), Heritage and Local Government], while for those outside the LMA it was unclear whether tunnels or chambers might extend underneath the LMA. It was therefore necessary to determine the extent of sett complexes at a number of locations[3] in order to determine i) whether proposed construction works might negatively impact upon them, or ii) whether the final road layout for construction would be impacted.

Sett Complexes

Badgers are common and widespread throughout Ireland (Smal 1995, O'Corry-Crowe *et al.* 1993, Sleeman and Mulcahy 2005, Delahay *et al.* 2008). They live in social groups mostly consisting of between two and six adults and their young (NRA 2006[4]). Territory size is on average 80 hectares (range of 25 to 200 ha). Recorded densities in East Offaly close to our study area are 0.7 groups/km^2 (O'Corry-Crowe *et al.* 1993).

Within each badger social group's territory there may be several setts of varying status and usage (HA 2001). Setts vary in size from simple single entrance setts to sett complexes with up to 40 or more entrances spread over 100 metres or more (NRA 2007). Generally, setts are categorised as main, annex, subsidiary or outlier setts, depending on factors such as number of sett entrances, patterns of occupation and connection to a main sett by well-worn pathways (Neal and Cheeseman 1996); an alternative strategy is to recognize only main or outlier setts. If not disturbed, setts can be used by successive generations over a considerable span of time. Furthermore, the influence of landscape, soil and bedrock type *etc.* on the size and design of any given badger sett, is such that the extent of a sett can be difficult to define accurately by examination of surface features alone. This is particularly so in Ireland where many sett entrances are in hedges, with tunnels radiating out under adjoining improved grassland where there are no visible sett structures [*e.g.* O'Corry-Crowe *et al.* (1993) found that most setts (55%) in their East Offaly study area were in hedges, which occupied only 3% of available habitat and that their location was little affected by soil type].

Non-Invasive Geophysical Techniques

As noted, the objective of this study was to determine whether any badger tunnels or chambers extend under lands made available for construction. In accordance with requirements arising from consultation with National Parks and Wildlife Service non-invasive geophysical survey techniques were used. While the possibility of using such techniques is highlighted on advertising material from a range of commercial geophysics companies, a review of the literature highlighted only a single published study; on St. Asaph Bypass, North Wales in 2003 (Nichol *et al.* 2003). This paper therefore outlines our experience of applying these techniques to mitigating impacts on badgers on the N7 road scheme in Ireland.

A combined approach of using Ground Penetrating Radar (GPR) and Electromagnetic (EM) techniques was employed to carry out this survey, together with a detailed assessment of surface features. With ground penetrating radar high frequency pulses of radio energy are transmitted into the ground. The transmitted pulses are reflected from material boundaries, building up a continuous cross section of the subsurface. Different frequencies are adopted in different situations, with high frequencies giving good spatial resolution of features and lower frequencies providing greater penetration to depth. Electromagnetic (EM38) Conductivity Mapping operates on the principle of inducing currents in conductive substrata and measuring the resultant secondary electro-magnetic field. The strength of this secondary EM field is calibrated to give apparent ground conductivity in milliSiemens/metre (mS/m).

Methods

Ground Penetrating Radar

In this study, Apex Geoservices Ltd were retained to undertake on site investigative works. The GPR survey was carried out using a MALA system, with a 500 MHz cart-mounted antenna, with a built-in odometer wheel. The data were recorded on the hard disk in the operating console and later transferred to a computer for processing and analysis. Notes were taken concerning the position of visible site details. GPR profiles were recorded across survey areas, with a nominal line spacing of 1 m. A time recording window of 56 ns was used giving a corresponding maximum usable depth of penetration of 2.8 m. Some areas were not surveyed due to obstacles such as trees or overgrown areas. All GPR profiles were surveyed using an RTK GPS system to 20 mm accuracy, in Irish National Grid co-ordinates.

In order to calibrate the system on site the location of known sett entrances and associated tunnels were surveyed. As each entrance had an associated tunnel radiating out from it, this provided an opportunity to survey for voids/tunnels in an area where badger tunnels were known to occur. A nearby culvert provided a further opportunity to calibrate the system on site.

The processing of GPR data was carried out using proprietary processing software (ReflexWin v4.5). The following processing was applied to the data:

- spatial relocation (data merge with RTK GPS data);
- temporal relocation (depth correction);
- amplitude recovery gain (time dependant);
- frequency bandpass filtering; and
- background noise removal.

Each GPR trace was analysed and the accurate location of the hyperbolic features were exported to the AutoCad plan drawing of the site with depth below ground level.

Electromagnetic Techniques

The equipment used was a Geonics EM38 Conductivity meter equipped with data logger. The instrument has an optimum depth of investigation of 1 to 1.5 m below ground. Conductivity values were recorded along all of the GPR profiles. Local conditions and variations were noted. The data were downloaded and contoured using proprietary software (Surfer v8.0). Variations in conductivity values were analysed in conjunction with GPR anomalies to identify subsurface voiding.

Result of the Badger Surveys

As noted, a series of pre-construction badger surveys were undertaken along the length of the scheme. Following finalization of the horizontal alignment of the scheme a total of six locations were selected for non-invasive surveys. For the purposes of this report we have presented the results of work from a single sett complex located along the southern boundary of the route. The sett entrance was located on the edge of an overgrown hedgerow, with one sett entrance clearly identifiable in the dense ground cover. Further field signs included a disused sett entrance c. 100 m to the north and evidence of prints and well used tracks.

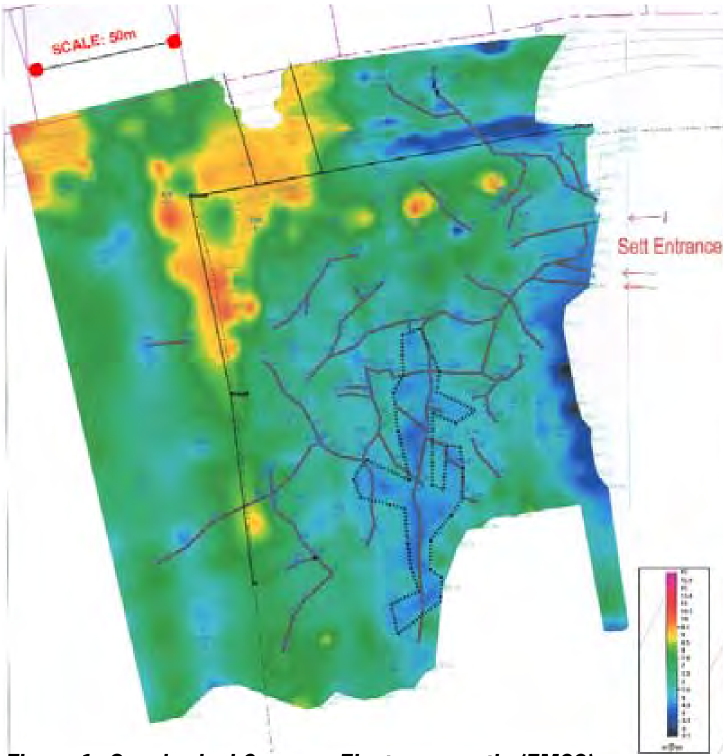

Figure 1: Geophysical Survey – Electromagnetic (EM38) survey results. Black lines indicate inferred tunnels. A central area is highlighted which reflects a core area of signal – potentially a concentration of tunnels and chambers.

As can be seen in Figure 1, three sett entrances were located along the eastern boundary of the sett complex; these were located at 13 m, 16 m and 17 m, respectively from the edge of site works. Each entrance allowed access to a single interconnected complex of tunnels. All structures identified in Figure 1 are between 0 m and c. 2 m below ground, within the known range for badger sett structures (Neal and Cheeseman 1996). However, a significant number were closer to 1 m in depth. The predominantly shallow nature of tunnels may however be explained by ground investigation results, which encountered groundwater intrusion at between 1.3 m and 3 m below surface level[5] in the environs of the sett complex. Using GPR depth readings a conservative core area of c. 22-24 m from the sett entrances can be mapped at depths of 0.5 m to 1.5 m. While the maximum axes lengths for the sett complex are 36 m (on the north-south axis) and c. 33 m (on the northeast-southwest axis), the shallow nature of some of these outlying tunnels likely indicates that these may be artefacts or perhaps old, collapsed tunnels. In a number of instances floating tunnels were also identified. As it is common for setts to be constructed at different levels, these may indicate sections of tunnel, which drop down below the maximum survey depth achieved in this study; though again a number of shallow floating voids are likely to be artefacts.

Discussion

Sett Architecture

The geophysical survey methods discussed above provide useful non-invasive tools for the investigation of the extent of badger setts in situations where direct investigative methods are not possible or are best avoided. In this study it emerged that the bulk of the sett complex was at a greater distance from the lands made available for construction than were the identified sett entrances. Thus unnecessary disturbance through digging and exclusion of badgers from the sett complex was avoided.

The study also highlighted the need to apply the NRA (2006) guidance, which states that badger sett tunnels may extend up to c. 20 m from sett entrances cautiously. As can be seen from Figure 1 local topography can influence the architecture of a sett; in this case very much biased in extent to the southwest of the sett entrances. Also in this case the outer tunnels do extend more than 20 m from the sett entrance. Thus when considering what type of construction works or site investigative works can be undertaken close to a sett the NRA categories should be used as guidance, but not as a definitive cut-off. Where doubt exists the advice of a qualified ecologist, who can review the context of the sett relative to ground conditions, local topography *etc.*, should be sought. It is in these cases that consideration could be given to the use of non-invasive geophysical survey techniques.

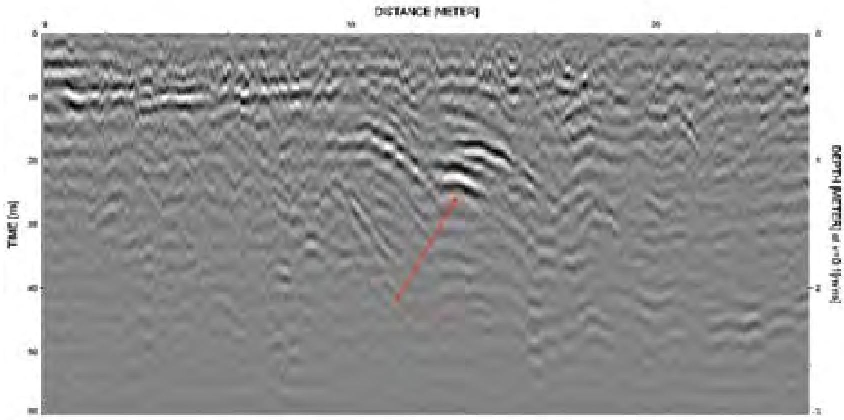

Figure 2: Example of Ground-penetrating radar (GPR) results – the distortion indicated by the red arrow is interpreted as a possible sett structure (tunnel).

Constraints to Consider

Clearly questions remain as to the widespread application of non-invasive geophysical survey techniques and when used, which method should be favoured. The first question is that of data validation. Due to the commercial constraints imposed on this work, it was not possible to validate the findings by undertaking a non-invasive survey of a sett proposed for excavation; those setts within the LMA already being excluded under licence. Such a study would allow for voids identified as putative tunnels to be validated as such. However, in this instance, a sett complex with known entrances from which tunnels were known to radiate, was used to ground truth work on site, as was a nearby culvert. It is proposed to undertake further such surveys in future projects where feasible.

In this study we utilised both Ground penetrating radar (GPR) and Electromagnetic Conductivity. GPR is influenced by both soil type and water content in the soil. The higher the clay content or the water content of soils within the survey zone then the shallower the depth to which the GPR can investigate. Made-ground, metallic objects or utilities can also distort results. Generally speaking bedrock depth is too great to influence results, though in areas of shallow bedrock, such as karst limestone this may be a consideration. Generally speaking the ground to be surveyed has to be reasonably flat, with ideally no hedge or tree roots that could be misinterpreted as voids, though can normally be discriminated (see for example the noise along the eastern side of Figure 1 – hedgerow). Ground conditions on many sites may therefore preclude its use.

In the current study, GPR was found to slightly out perform Electomagnetic (EM) techniques. However, in the current study EM38, which surveys to a depth of 1.5 m, was used (GPR surveyed to 2.8 m). An alternative strategy would, however, be to use EM31, which in horizontal dipole mode has a penetration depth of c. 3 m. It would be our intention to test EM31 at the next opportunity we have to investigate a sett.

While GPR can see the voids, EM gives you a measure of the conductivity. Generally a void gives a low conductivity reading; however, this could potentially be complicated if badgers are urinating within the sett complex; the salty nature of urine can give a high conductivity rating thereby cancelling out the anomaly. This effect may be more pronounced during the winter months when badgers are not coming to the surface to urinate and defecate.

EM signals are disrupted by the presence of metallic features with the resulting noise dominating the results. EM techniques should not be used in the immediate vicinity of fences and power lines where possible (an exclusion zone of 3 m for EM 38 and 10 m for EM31 is generally acceptable. GPR, however, is not affected by the presence of fences and/or power lines.

In the current study, the results of the GPR survey gave a good representation of the location of the tunnels and when viewed in combination with the EM data gave a clear interpretation, highlighting the value of an integrated approach to such studies.

While cost is a further consideration, in this study where we had a number of sites to visit the technique proved cost effective.

As noted, a large proportion of setts in Ireland are in hedges adjoining open agricultural lands; this methodology may therefore be more widely applicable in Ireland than perhaps in the UK.

Conclusions

Clearly, in certain cases this technique could be a valuable addition to the survey methods and mitigation options available to badger surveyors, as in this study where it assisted in defining a safe work area. Whilst the technique is now much easier to use in the field and is becoming more cost effective its precision must be further tested. In order for it to be more widely adopted a series of validated surveys would need to be undertaken under differing ground conditions. This would offer the opportunity to prepare best practice guidance for both ecologist and geophysical surveyors considering undertaking such work.

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Notes

[1] Wildlife Act 1976 (No. 39 of 1976) and Wildlife (Amendment) Act 2000 (No. 38 of 2000) - www.irishstatutebook.ie/

[2] Licences in Ireland are granted by National Parks and Wildlife Service, Department of Environment, Heritage and Local Government.

[3] In order to avoid disturbance of setts we have not included figures showing sett locations.

[4] Smal (1995) recorded an average group size of 5.9 adults per group in Ireland.

[5] Topsoils and subsoils in the environs of the sett complex comprised a mix of soft brown slighty sandy, slightly gravely clays. Gravels were fine to coarse and of various lithologies. In places the topsoil comprised black peaty clay with rootlets.

Paul O'Donoghue, Ross Macklin and Paul Dansie

White-Clawed Crayfish: Use of Drainage Ditches

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White-Clawed Crayfish: Use of Drainage Ditches

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White-clawed crayfish *Austropotamobius pallipes*, Ireland's only native crayfish, is the largest mobile invertebrate in aquatic systems in Ireland. It is viewed as a keystone species which is under threat across its European range from habitat loss, pollution, competition from introduced non-native species (such as the North American Signal Crayfish *Pacifastacus leniusculus*) and in particular from crayfish plague or *aphanomycosis* (Matthews and Reynolds 1995). As such it is listed for protection on Annex II and V of the Habitats Directive (92/43/EEC) and Appendix II of the Bern Convention and nationally under the Wildlife Act 1976 and the Wildlife (Amendment) 2000.

On a broad scale its distribution is largely dictated by a combination of local geology and water quality factors. It favours areas with relatively hard, mineral rich waters on calcareous rocks (Holdich 2003) and as such is widely distributed throughout the limestone rich Irish midlands and west. In Ireland, crayfish have been recorded from a wide variety of habitats, including canals, mill races, streams, rivers, lakes, reservoirs and water-filled quarries. However, little mention is generally made of the use of drainage ditches that are hydrologically connected to such habitats despite these being ubiquitous in the Irish landscape. Demers *et al.* (2005) found that crayfish were most commonly encountered in unpolluted waters, but that they were also found in slightly polluted and moderately polluted water, so the potential would seem to exist for movement of crayfish into such habitats, as long as other factors are suitable.

In a recent study of 27 watercourses conducted by Atkins in east Co. Galway (hydrometric areas 26 and 29), which included rivers, streams and drainage ditches on a total of seven different sub-catchments, crayfish were recorded at six sites (22% of sites) (under NPWS licence no. C69/2007).

Of the above 27 sites, five drainage ditches were surveyed; three additional watercourses which were degraded small stream habitats, in many ways characteristic of drains, were also surveyed. Adult crayfish (*i.e.* two to three year plus individuals) were recorded in two ditches and in two of the three degraded ditch-like small streams; in all cases these were either trapped or caught in a sweep net. One site was within 30 m of a river where large numbers of young crayfish were also captured. In a separate study in Co. Tipperary, an adult crayfish was also encountered in a drainage ditch which connected a wetland pond to the River Multeen Special Area of Conservation.

While we do not as yet have adequate data to look at habitat factors positively associated with distribution, the presence of in stream macrophyte cover, especially *Apium nodiflorum*, was notable, as was the presence of cobble and gravel substrate in the base of the ditches and positive for the presence of crayfish.


White-clawed crayfish Photo: Gordon Howe (Environment Agency)

The above results indicate that drainage ditches cannot be discounted as a potential habitat for white-clawed crayfish and highlights the need for further work in this area. Consideration should be given to the potential role of ditches as refugia (*e.g.* for repopulating watercourses in the event of pollution events; or as safe refugia for young in high flow conditions or in cases of high population density), availability of foraging habitat, and the possibility that they may play a role in terms of crayfish movement and habitat connectivity.

In particular, the study highlights the need for the adoption of an agreed methodology for the evaluation of ecological importance and assessment of development related impacts on ditches in Ireland. To this end we understand that Ms Jane Kavanagh is reading for a PhD under Dr Simon Harrison in University College Cork on the *Freshwater Ecology of Drainage Ditches* from which it is hoped to develop an appropriate survey and evaluation methodology.

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Veronica Lawrie

Developing a Grasp of European Eel Conservation

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Developing a Grasp of European Eel Conservation

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'Every November, when the Moon is at its darkest, there's a stirring. On riverbeds, lake bottoms and marshlands around Europe writhing masses of snake-like fish respond to an ancient urge and turn towards faster-moving water. This is the time when adult eels begin a 4,500-mile journey down deep ocean trenches and across undersea mountain ranges to end their lives spawning in the windless waters of the Sargasso Sea.'[1]

This is the received wisdom on eel migration, but in fact much of the ecology of the European eel *Anguilla anguilla* remains a mystery to biologists. In an attempt to draw together what is currently known about these intriguing and increasingly endangered creatures a review of relevant current literature is drawn together in the following paragraphs. In order to protect and enhance this species in the UK we first need to understand their ecology.

The journey that adult eels make to the Sargasso Sea to end their lives after spawning has been inferred from catching increasingly smaller juvenile eels (Leptocephali – see life cycle section below) in the approach to the sea. No spawning has been observed directly in the Sargasso Sea and no adult eels have been found there. To date, the only adult eel that has been found in proximity to this sea was recovered from the stomach of a sperm whale off the Azores. Further mystery enshrouds the Sargasso Sea itself. It is the only sea without shores, an area of ocean distinct from the surrounding Atlantic, bounded by the currents of the Gulf Stream, a windless expanse of floating seaweed reputed to trap ships; indeed it is located within the Bermuda Triangle.

Biologists still have much to learn about the European eel. One thing we do know is their numbers have dramatically declined throughout Europe in the last 25 years and, as a result, in 2007 the species was added to the International Union for Conservation of Nature Red List of critically endangered species[2]. 'Critically endangered' is the highest risk category for wild species, and means that the species numbers have decreased or will decrease by 80% within three generations. The causes of this decline are not fully understood but are likely to be manifold and include nematode infection, obstacles to migration as a result of development and overfishing.

The species is further included in the UK list of priority species as well as the Natural Environment and Rural Communities (NERC) Act 2006 Section 41 list (England) and Section 42 list (Wales) of species of principal importance for the conservation of biodiversity. The NERC Act places a legal duty under Section 40 of the Act, such that - 'Every public authority must, in exercising its functions, have regard, so far as is consistent with the proper exercise of those functions, to the purpose of conserving biodiversity'. The Eels (England and Wales) Regulations 2009[3] came in to force on 15 January 2010 to implement the short- and long-term measures set out in the Eel Management Plans[4], which are intended to ensure at least 40% of adult eels return to the sea to spawn.

Taxonomy

The European eel is one of many species of eel that exist. True eels (Anguilliformes) are an order of fish, which consists of four suborders, 19 families, 110 genera and approximately 600 species. The term 'eel' is also used for some other similarly shaped fish, such as electric eels and spiny eels, but these are not members of the Anguilliformes order.

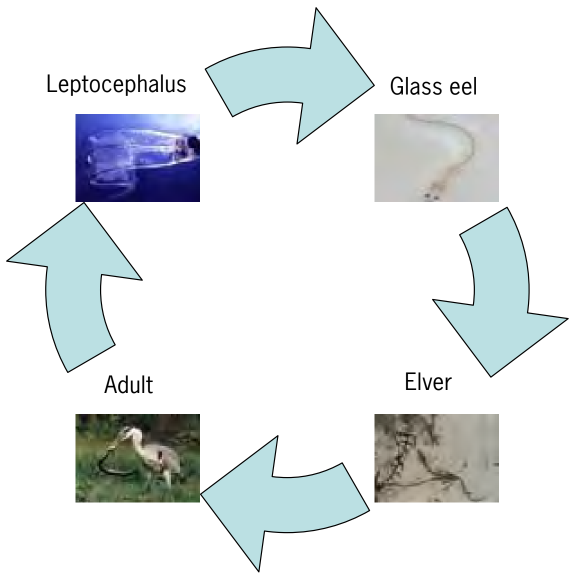
There are two eel species found in UK waters; the conger eel *Conger conger* which only lives in marine habitats and the European eel *Anguilla anguilla* which lives in freshwater habitats, both still and flowing, as well as inshore coastal waters.

Life Cycle

Thought to start life in the Sargasso Sea, the European eel larvae are flat, leaf-like creatures (known as a leptocephalus) that are carried on oceanic currents towards Europe. As they reach the coasts of Europe and enter estuaries they have transformed into small, transparent glass eels[5]. The glass eels metamorphose into pigmented elvers as they enter the UK estuaries with the spring tides in April and May, migrating upstream into freshwater where they stay and mature for up to 20 years, attaining a size of 60-80 cm. Adult eels spend most of their lives in freshwater, but they are capable of surviving for short periods of time out of water and can cross land and damp meadows in their search for water systems.

Eels are an important source of food for fish, birds and mammals, including other protected species such as bittern and otter. They are also an important commercial fish with long-standing freshwater fisheries across the UK.

The life cycle is illustrated below (photos courtesy of the Zoological Society of London, the Friends of Troopers Hill[6] and Wikipedia).

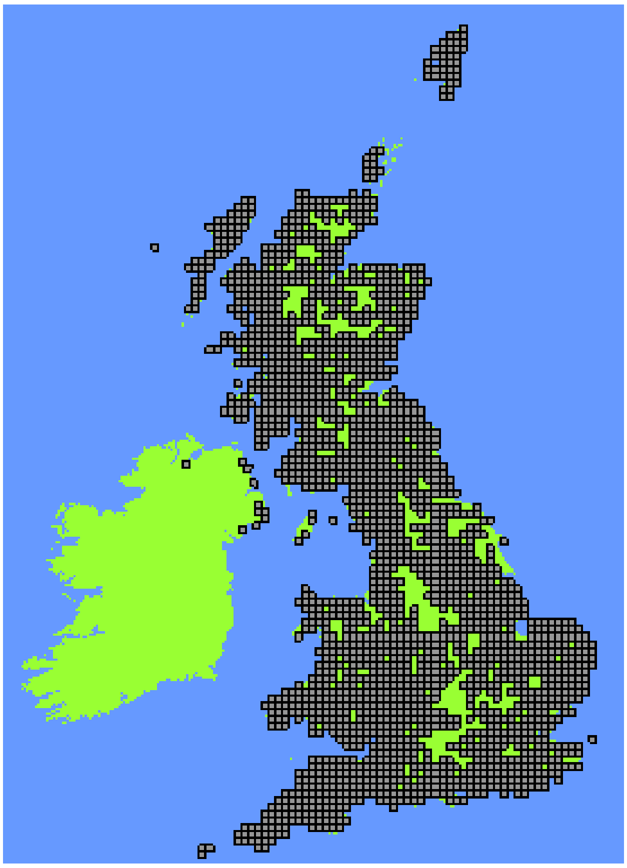


Habitat Requirements and Distribution

Eels can be found in wetland habitats including rivers, streams, waterfalls, bogs, marshes, swamps, fens, peatlands, lakes, canals and ponds as well as more saline habitats of river estuaries and the UK coastline. Since eels are capable of surviving for periods out of water, and will cross damp meadows to reach new habitat areas for maturation or migration, their ability to colonise a range of freshwater habitats has resulted in eels becoming one of the most widely distributed British freshwater fish.

The presence of eels within a site may be an indicator of a healthy and diverse ecosystem, or at least a eutrophic one, as they require abundant invertebrate prey. As they grow, freshwater eels diversify their prey range to include invertebrates of all sizes, newts, fish spawn, fish and even young waterfowl (ducklings)[7]. They require adequate physical cover to protect them from excessive predation, such as overhanging trees and dense marginal vegetation.

Eels are found throughout the UK as shown on a recent distribution map below[8]; gaps on the map may indicate gaps in recording effort rather than an absence of eels.



Research Projects

There are several research initiatives that are underway to improve our understanding of eel ecology and the factors leading to their decline. The pan-European research project known as the Eeliad[9] is a cooperative project with institutes and researchers from across Europe working together to better understand and safeguard this critically endangered species.

The project aims to:

- identify spawning areas and marine migration routes;
- identify biological and ecological characteristics of eels that contribute to migration success and reproduction;
- develop understanding of the recruitment processes of eels from hatching to their entry to river catchments;
- refine our understanding of the stock structure of European eels; and
- make recommendations to national and international organisations for regional conservation measures to maximise recovery of the European eel stocks.

INDICANG[10] is another research project that is linked in to the Eeliad with work being undertaken in the UK. Furthermore, the Zoological Society of London has been monitoring eel migrations in the River Thames and its tributaries since spring 2005 as part of the Tidal Thames Conservation Project[11]. These research projects are gradually improving our knowledge of eel ecology.

Conservation and Enhancement Recommendations

During a walk-over survey it would be hard for an ecologist to predict the distribution and abundance of eels on the basis of the habitats within a site alone. Such detailed survey information may not be necessary as all wetland habitat has the potential to support European eel and their presence can be assumed. Indeed, surveys conducted by the Environment Agency show eels to be present in nearly all river systems in England and Wales, although there are some areas where they are scarce or absent, particularly the upper reaches of rivers.

Mitigation for eels is necessary if a development results in impacts to wetland habitats. Even if the wetland habitats within a site are not affected there are opportunities to enhance a site for eels.

Approaches to mitigation and enhancement will vary between sites, but may include: (i) improvements to water quality - ongoing programmes aimed at achieving better river quality and good ecological status under the Water Framework Directive will contribute to increasing eel populations; and (ii) provision of abundant marginal vegetation and over-hanging trees - which will both increase the abundance of invertebrates and small fish for eels to feed on and offer refuge sites for eels from predators.

Another key mitigation strategy for conserving eels is to ensure there are no obstacles to migration for elvers moving upstream and eels as they travel downstream on their return to the Sargasso Sea. Elver and eel passes[12] can be installed over obstructions, and standard fish passes can be adapted for eels. Such passes have been used at numerous locations by the Environment Agency (EA) who will regulate and advise on any projects where eel mitigation is undertaken. Elver passes can be a low cost option (from £100) and often involve a narrow strip of bristles that water gently trickles down (see photograph of the construction of such a pass). Elvers will wriggle up between the bristles, which are spaced at varying intervals for elvers of different sizes to grip on. Monitoring the success of the passes is important and can be done by netting elvers as they swim over the top or by capturing the image on a night camera.

An EA consent is required for capturing eels and elvers, for scientific purposes, and the eel fishing byelaws, which are to be updated in 2011, must be adhered to. For example, in different areas of the country particular capture techniques are banned[13]. The byelaws aim to protect eels at particularly sensitive times

Eel pass construction
Photo: Roger Genge, Environment Agency

in their life cycle such as the spring migrations up streams and the autumn migration to the Sargasso Sea. Works affecting wetlands at these times of year are best avoided, especially at night time when eels are on the move.

In the absence of a comprehensive understanding of eel ecology we rely on the Eel Management Plans, that are enforced by The Eel (England and Wales) Regulations 2009, to conserve and enhance the species.

The views expressed in this article are the author's personal views. The comments of her colleagues at Atkins are gratefully acknowledged.

Notes

[1] Adapted from *The Times* website 2009: http://www.timesonline.co.uk/tol/news/environment/article6163961.ece. Accessed on 14 July 2010.

[2] IUCN website 2010: http://www.iucnredlist.org/apps/redlist/details/60344/0/full. Accessed on 14 July 2010.

[3] Eel (England and Wales) Regulations 2009: http://www.opsi.gov.uk/si/si2009/uksi_20093344_en_3#pt4-l1g14 eels. Accessed on 14 July 2010.

[4] Eel Management Plans overview: http://www.defra.gov.uk/foodfarm/fisheries/documents/fisheries/emp/overview.pdf. Accessed on 14 July 2010.

[5] ZSL website 2010: http://www.zsl.org/conservation/regions/uk-europe/european-eels,1035,AR.html. Accessed on 14 July 2010.

[6] Friends of Troopers Hill website: www.troopers-hill.org.uk. Accessed on 14 July 2010.

[7] Yorkshire Wildlife Trust website 2010: http://www.ywt.org.uk/_filestore/File/Wildlife%20Factsheets/Eel%20Fact%20Sheet.pdf. Accessed on 14 July 2010.

[8] The various Data Providers, Original Recorders and the NBN Trust bear no responsibility for any further analysis or interpretation of this data. National Biodiversity Network website 2010: http://data.nbn.org.uk/interactive/map.jsp?srchSp=NBNSYS0000188599. Accessed on 14 July 2010.

[9] The Eeliad project website: http://www.eeliad.com/. Accessed on 14 July 2010.

[10] INDICANG website: http://tamarconsulting.org/wrt/projects/indicang.htm. Accessed on 14 July 2010.

[11] Thames Tidal Project webpage: http://www.zsl.org/conservation/regions/uk-europe/european-eels,1035,AR.html. Accessed on 14 July 2010.

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Little Egret Expansion in Ireland: Cork – A Case Study

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Little Egret Expansion in Ireland: Cork – A Case Study

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The expansion of the little egret *Egretta garzetta* into Britain and Ireland as a breeding bird has been well publicised in both the scientific literature and the general press. What is probably less well understood, however, is the speed and scale of the subsequent expansion of breeding numbers and locations.

In Ireland, breeding was first proven in 1997 (Smiddy and Duffy 1997). This followed a pattern of increased levels of vagrancy from roughly 1989 onward with birds largely resident in Ireland from 1990 onward. Most birds were recorded from coastal areas in the south and southeast. A large influx of birds occurred in the autumns of 1995 and 1996; this led to a wintering population of about 60 birds (Smiddy 2002) and may have provided the final impetus for breeding on the south coast. As noted, breeding was first proven in 1997, when 12 pairs were recorded at a site on the River Blackwater on the border between counties Cork and Waterford (on the south coast). By 2001 egrets had been recorded breeding at four separate sites in counties Cork and Waterford (Smiddy 2002), and the number of nesting pairs increased from 12 in 1997; 22 in 1998; 32 in 1999; 45 in 2000 to 55 in 2001 (Smiddy 2002).

By 2000 egrets had established a breeding colony in Cork Harbour (Ballyannan Wood; northeastern harbour) confirming a suspected westward expansion. When a further colony was discovered in the harbour in 2004 (Little Island; mid-harbour), the authors were prompted to undertake a systematic survey for egret breeding sites. Cork Harbour is a large complex system of basins, channels, estuarine areas and river channels, which offers a multitude of sites for foraging and breeding egrets. Further breeding sites were proven in 2005 (Fota Island; mid-harbour and Rostellan; eastern harbour) and 2007 (Atlantic Pond; public park within city bounds), bringing the known number of sites to five and the number of breeding pairs in Cork Harbour alone to over 70. Furthermore, two young birds were observed with two adults in late summer 2005 near Carrigaline on the upper Owenaboy Estuary (western harbour), indicating the possible occurrence of a sixth site. The number of egrets in this area of the harbour is also suggestive of another breeding site, which could support in the region of 5-10 nests (location unknown). Survey work in 2007 has also highlighted a possible seventh site within the eastern harbour, while there is anecdotal evidence of a pair from an eighth site (Minane Bridge). To date all sites also support breeding grey herons *Ardea cinerea*. Close human activity has not been a deterrent in site selection.

In 2007 egrets were breeding at up to four sites in West Cork, with Rosscarbery the furthest west known to the authors (*c.* seven nests in 2006). The expansion in range and breeding numbers/sites in Cork is mirrored elsewhere along the south, southeast and east coasts. Coincident with this increase in numbers has been an increased incidence of field feeding. Little egrets are also now being recorded from small streams and large inland rivers such as the River Bandon upstream of Innishannon and the River Blackwater upstream of Fermoy. The observed use of new habitats and feeding strategies raises the possibility of breeding away from traditional coastal sites at large river and wetland sites, and in fact the first inland colony was recorded near Fermoy in 2007 (on the Blackwater). The Bird Atlas 2007-2011 provides a perfect opportunity to examine this trend further and we would encourage surveyors to keep a lookout for breeding little egrets.

This winter has also seen the influx of large numbers of cattle egrets *Bubulcus ibis* to both Britain and Ireland prior to Christmas. Up to 20 birds have been recorded in West Cork with up to 10 roosting at a known grey heron/little egret breeding site; up to three birds were recorded in the same area last year. It will be interesting to see in coming years whether cattle egret may be the next addition to our breeding avifauna.

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Little egret
Photo: Paul O'Donoghue

Heather Mansfield

Roman Snail: An Introduction to its Ecology and Legal Protection

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Roman Snail: An Introduction to its Ecology and Legal Protection

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In 2008, the Roman snail *Helix pomatia* was added to Schedule 5 of the Wildlife and Countryside Act 1981 (as amended), and it became an offence to intentionally kill, injure or take individuals of this species (as did possession and sale). Also known as the 'edible snail', the primary reason for its legal protection in England and Wales (and elsewhere in Europe) was an increasing trend in collection of large numbers by amateur cooks and for commercial use in restaurants. However, the legal protection this species is now afforded has implications for development projects. Distributed throughout South East England (but especially the North Downs) and through the Chilterns and Cotswolds, and occupying a broad range of habitats (where suitable soils are present), this species could occur on a wide variety of sites. This article provides an introduction to Roman snail ecology and licensing requirements, and illustrates these using a case study in Surrey – the M25 Controlled Motorways scheme.

Atkins ecologists first came across Roman snails in early 2009, when working on behalf of the Highways Agency, undertaking an Environmental Assessment as part of proposals for the installation of new gantries along a stretch of the M25 motorway in Surrey (the M25 Controlled Motorways scheme). An empty Roman snail shell was found during an extended Phase 1 habitat survey, at the base of a steep chalk section of the motorway verge between junctions 7 and 8 of the M25. On a subsequent nocturnal survey, a live individual was found in an area of long, semi-improved grassland with dense patches of bramble, close to junction 8. Atkins ecologists have also found Roman snails on another section of the M25 motorway

Photograph 1. Roman snail habitat on M25 verge
Photo: Atkins Ltd

Photograph 2. Roman snail habitat on M25 verge
Photo: Atkins Ltd

(close to junction 6), when working on a separate project for the Highways Agency. Shells were found within plantation woodland on the verge and live individuals have been spotted numerous times in the tussocky grassland situated directly behind the woodland.

As a result of these findings, and a need to resolve the issue of the presence of this legally protected species within proposed construction areas for the above scheme, further surveys have been carried out and appropriate licences sought.

Habitat Requirements and Distribution

The Roman snail is known to inhabit open woodland, rough and tussocky grassland, hedge banks, chalk quarries and areas of scattered scrub. Photographs 1 and 2 show the areas of the M25 motorway verge where Roman snails have been found.

This species requires loose, friable soil for burying into for hibernation and also for depositing eggs. Lime-rich, free draining soil is a habitat requirement in the UK and studies have found a preference for south-facing slopes (Pollard 1975). Roman snails will not occur in sandy soil. They will also avoid grazed grassland and very open, exposed habitats.

Figure 1 shows a United Kingdom distribution map for Roman snail (Kerney 1999). The species is not native to the UK and is thought to have been introduced by the Romans. Much of its distribution in the UK is considered likely to be due to local introductions by humans. There are documented introductions elsewhere in England and also in Scotland and Ireland, and these are still shown on some distribution maps, but these introduced animals rarely survived for very long (Kerney 1999). This was presumably because soil and/or weather conditions

Figure 1. Distribution map for the Roman snail, from Kerney (1999)

were not suitable. The main hotspots for populations of Roman snails in England are along the North Downs (from Surrey to Kent), the Chilterns (especially in Hertfordshire) and throughout the Cotswolds and Mendip Hills fringes. There are also documented populations in Cambridgeshire.

Life History

Many aspects of the Roman snail's life history and behaviour contribute to its vulnerability to over-exploitation. In particular, their tendency to aggregate in high numbers and disperse only short distances leaves them vulnerable to collection. Individual snails may spend their entire lives within an area of approximately 30 m in diameter and take two to five years to reach maturity and reproductive success may be low, with many British populations found to have a low proportion of young snails (Alexander 1994).

In England, Roman snails are typically active from May to August. The earliest and latest dates for activity in an area of the Cotswolds were 30 April and 1 September (Alexander 1994), with peaks in activity most likely in May and June (Dr Martin Willing, Conchological Society, pers. comm.).

Roman snails hibernate in the ground by digging down into loose soils, pulling vegetation and soil over the top to close the top of the entrance to their chamber. They remain in hibernation until spring.

Identification

Adult Roman snail shells are typically larger than those of other snail species in England, measuring up to 5 cm across and displaying a pattern of brown bands (see Photograph 3). Crucially, the bands on their shell lack the zig-zag pattern found on the garden snail *Cornu aspersum* (= *Helix aspersa* - see Photograph 4). The body of the Roman snail is pale grey and measures up to 10 cm long on adults.

Empty Roman snail shells often appear very pale, and lack the brown colouration shown in Photograph 3, as do juvenile Roman snail shells (shown on the right in Photograph 5). Empty shells become 'bleached' and in this state are usually more than one year old (Dr Martin Willing, pers. comm.).

Surveying for Roman snails

Whilst no standard published survey technique for Roman snails currently exists, it is considered that the combination of careful hand searches and one or two nocturnal torch surveys in suitable weather conditions, as described below, will allow an assessment of presence or absence of Roman snail at a site.

Daytime Hand Searches

Two survey techniques were used by Atkins for the M25 Controlled Motorways scheme, once the presence of the species had been confirmed, following the identification of an old shell during the initial extended Phase 1 habitat surveys in 2009. Hand searches of areas of habitat to be affected were carried out. This involved searching through areas of long grass and scrub by hand, looking for Roman snails and old shells. Particular attention was paid to searching underneath logs, brash and artificial refuges present on the verge of the motorway. Some gantry locations were ruled as not suitable for the species, due to the presence of sandy soils. This hand searching technique was effective because each of the footprints for gantry construction were relatively small; the working area for each gantry footing (*i.e.* total vegetation clearance) was a maximum of 10 m x 15 m (150 m^2).

In larger areas of habitat, attention would best be focused on log piles and areas that could provide refuge (see Photograph 6). This is best carried out during the snail's active period (May to August), after recent rainfall, especially in warm, humid conditions. Individuals will bury into the topsoil during prolonged hot/dry spells. At sites with well-established colonies, evidence of Roman snail presence can be found at any time of the year, in the form of empty shells.

The tendency for Roman snails to aggregate in high numbers and the longevity of their shells means that hand searching over relatively small areas is an effective way to search for evidence of this species.

Torch Surveys

In areas deemed potentially suitable for Roman snails, a nocturnal survey was also carried out, in June, in order to look for active Roman snails. Ideal timing for torch surveys is late April to early June. This involved searching areas with a powerful torch at least one hour after sunset. This survey technique relies on appropriate weather conditions; it must be raining, have rained in the last 24 hours or be humid and it should also be warm.

A juvenile Roman snail was found during the torch survey for the M25 Controlled Motorways project.

Photograph 3. Adult Roman snail
Photo: Martin Willing

Photograph 4. Roman snail shell (left), garden snail shell (right)
Photo: Atkins Ltd

Legislation and Licensing

The Roman snail was added to Schedule 5 of the Wildlife and Countryside Act in April 2008. It is not a European Protected Species, although it does receive legal protection in other European countries. In the UK, it is protected in relation to Section 9(1), (2) and (5) of the Wildlife and Countryside Act only. This means that it is an offence to intentionally kill, injure or take this species. It is also an offence to possess a live or dead Roman snail (possession is only an offence if it has been illegally taken from the wild) and it is also protected against sale. It is not an offence to disturb Roman snail or to damage or destroy breeding places or resting places of this species. However, although disturbance is not an offence, a licence is needed to handle Roman snails, however briefly, because it is protected against 'taking'. This has implications for consultants carrying out surveys for this species. It is necessary to obtain a licence from Natural England for the purposes of science and education to allow to you to pick up and examine Roman snails.

Furthermore, where Roman snails occur within areas that are to be affected by development proposals, such that there is a need to move them to avoid killing or injuring of individuals, any intentional movement of Roman snails must be licensed or should be covered by a relevant defence in the legislation, because moving Roman snails, even short distances, constitutes 'taking'.

Licences can only be issued for specific purposes under the Wildlife and Countryside Act. There is no licensing purpose for development works. However, Natural England will consider issuing a licence for conservation purposes in certain circumstances. Any conservation licence application for Roman snails will need to demonstrate that the work proposed is essential and the impacts to the species cannot be avoided in any way. It would also need to demonstrate that the work will have some conservation benefit for the species. There is no standard methodology currently available for dealing with Roman snails and each licence application will be considered by Natural England on a case-by-case basis. The licence application for the M25 Controlled Motorways scheme is presented below as a case study example, to highlight the main issues for consideration.

Case Study: Licence Application for the M25 Controlled Motorways Scheme

A licence application for this scheme was made to Natural England in August 2010 and included information on four key areas, summarised below.

1. **Background to the project and details of why the work needed to go ahead**
   This included details about the scheme and how it would deliver safety improvements to the relevant section of the M25 motorway. Background to the Roman snail surveys and the habitats to be affected were provided. Across the 18 new gantry locations, vegetation clearance equalled 0.27 ha with a permanent habitat loss of 0.11 ha.
2. **Details of the population *i.e.* locations and numbers involved, and context in the wider area**
   The locations for each of the new gantries were provided, along with a brief description of habitat within each area. The results of the Roman snail surveys were set out. The location of this scheme, close to the North Downs (a hotspot for Roman snail in England), and within an area of well-connected habitat (the motorway verge) meant that populations were likely to be more robust than smaller populations elsewhere.
3. **Setting out the conservation aims and how these will be achieved**
   The conservation aim of the proposal in the licence application was to ensure the future longevity of the population of Roman snails in the area and help to maintain the conservation status of this species in the local area.
   Five new log piles would be created in areas outside of the gantry locations, in areas of habitat suitable for Roman snails to provide an enhancement to these species. Locations would be targeted at areas where woody cover is sparse. Log piles would be made from trees cut down as part of the gantry clearance and would be created under supervision by the ecologist.
   Areas of vegetation clearance would be hand searched for Roman snails and any individuals found would be moved to the surrounding suitable habitat (not more than 20-30 m from where they were found). This would take place outside of the hibernation period.
   Fencing would be erected around each of the works areas at each new gantry location. This fencing would be designed to deter Roman snails from re-entering areas prior to works commencing. Fencing would be 13 mm diameter chicken wire netting with metal stakes used at the corners for support. This sized mesh is small enough to prevent Roman snails getting through, due to the size of their shells, whilst containing holes large enough to discourage movement of snails up the fence. The fence would be buried in the ground to a depth of approximately

Photograph 5. Adult Roman snail shell (left), juvenile Roman snail (right)
Photo: Atkins Ltd

30 cm to prevent snails burrowing beneath. The top of the wire netting would be folded outward to create a 'lip' on the outside to further deter snails from entering. The fences would be 1 m high.

4. **Monitoring**
   One monitoring survey for Roman snails would take place in the year following completion of the works. This would take place within habitats around all of the new gantries and also immediately adjacent to the new gantries. The results of the survey would be assessed to ensure that the existing distribution of Roman snail within the local area has been maintained and would be used to inform further mitigation, if appropriate.
   Results of the monitoring survey would be passed to the Conchological Society national non-marine recording scheme and the local biodiversity records centre.

Delivering habitat enhancements for Roman snails will depend on the conditions at the site, but as well as creating log piles, could also be achieved by creating or introducing a base-rich, friable topsoil. In more open areas, creating more cover, through planting of scattered scrub, or relaxation of management regimes could deliver enhancements. Woodland edge could be improved through the creation of ecotone habitat, where this does not already exist.

The above application was granted by Natural England. However, subsequently, a decision was taken by the Highways Agency not to build new gantries in this part of the M25 Controlled Motorways scheme and therefore, this licence will not now be implemented.

Summary

The Roman snail is a relatively easy species to identify, once familiar with its characteristics. Identifying the potential presence of the species can be achieved through understanding of its habitat requirements and will be aided by the fact that, broadly, its distribution is quite well understood and likely to be relatively unchanging in England, due to its inability to colonise new areas quickly. However, increased surveying and reporting for the species, now it is legally protected, could lead to amendments to the distribution map, and it would no doubt be beneficial to send records to local biological record centres and to the Conchological Society of Great Britain and Ireland.

Dealing with Roman snails on development sites is relatively new and mitigation and habitat enhancement measures are currently largely untested. Collation of information from future projects will enable ecologists and stakeholders to refine techniques and test new approaches. As with habitat enhancements for other species, measures to improve habitats for Roman snails are likely to lead to benefits for other species in the local area.

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Photograph 6. Hand searching for Roman snails ***Photo: Atkins Ltd***

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