The effects of sedimentation removal on the habitat quality of the upper lake and River Llan at Penllergare Valley Woods

A report for the Penllergare Trust by:

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Overview

In April 2012 the leases of Penllergare Valley Woods (Site of Importance for Nature Conservation, SINC), Swansea were assigned to the Penllergare Trust, effectively securing them for public benefit until 2116. Subsequently the Trust was awarded £2.4 m by the Heritage Lottery Fund through its Parks for People programme to support the first phase of an ambitious £2.9 m restoration scheme focussed on the upper end of the valley. A significant proportion of the restoration of the Valley has been focussed on the upper lake, which required de-silting to enhance the biodiversity value and water quality and restore the aesthetic properties to resonate that of its 19^{th} Century past.

As the de-siltation process was likely to cause disruption to the ecological processes within the lake, the Penllergare Trust employed the assistance of Swansea Universities WISE Network and ATM students to conduct a baseline ecological census of the Lake and adjoining River Llan. An ecological appraisal of the freshwater macroinvertebrates within two interconnected water bodies was conducted over three months during the summer of 2013. Multiple biotic (macroinvertebrates and vegetation) and abiotic (substrate, pH, conductivity, total dissolved solids, total dissolved oxygen) indices for monitoring water quality were investigated.

Prior to dredging both the upper lake and River Llan were of fair to good ecological condition according to the BMWP score (148 and 220 respectively). Furthermore, favourable abiotic conditions were recorded that could promote and sustain a diverse, functional ecosystem. Post-dredging no changes were observed in macroinvertebrate abundance or richness, or abiotic factors in the River Llan. However, significant reductions of pH, conductivity and total dissolved solids were observed in the lake, suggesting the disturbance resulted in the release of a number of organic and inorganic compounds. Annual monitoring of the macroinvertebrates and abiotic conditions are recommended, along with suggestions to potentially reduce the siltation within the lake and treatment of invasive species.

1.0 INTRODUCTION

Penllergare Valley Woods is a 100 hectare (ha) temperate, mixed broad-leaved woodland situated in northern Swansea (The Penllergare Trust, 2008) and a CADW Register of Landscapes, Parks and Gardens of Special Historic Interest in Wales, Reference PGW(Gm)54(SWA). Furthermore the woodland was designated a Site of Importance for Nature Conservation (SINC) in the 2006 Local Biodiversity Strategy and Action Plan (David Clements Ecology, 2011) for Swansea. In accordance, the Friends of Penllergare Valley was formed to begin raising funds to implement effective management strategies within the site and subsequently formed the Penllergare Trust to provide further protection, conservation, and restoration for the cultural landscape and biodiversity (The Penllergare Trust, 2009).

In 2008 The Penllergare Trust published a conservation management plan for the valley woods (The Penllergare Trust, 2008). Proposals were designed to assess socio-economic, environmental and historical problems and focus aims for environmental improvement for the benefit of the local community and historical and biodiversity preservation (The Penllergare Trust, 2008). These plans also encompassed restoring the habitat quality of the freshwater bodies within the woodland, which is a legal requirement of the European Water Framework Directive (WFD, 2000). The WFD is a definitive act of legislation enforced by the European Union to ensure that all surface waters in Europe attain good ecological status by 2015. The Directive's stated aim is to establish a framework for action in the field of water policy in order to protect inland surface waters, transitional waters and groundwaters, and to promote sustainable water use based on the long-term protection of available water resources. Within Wales, over 43 % of rivers are classified as being heavily modified, particularly in low lying, coastal sites, with rivers and standing water bodies now classified as Priority Habitats in the UK Post 2010 Biodiversity Framework (JNCC & Defra, 2012). Consequently, stakeholders, government agencies and trusts such as the Penllergare Trust are charged with improving, maintaining or restoring the habitat quality of freshwater systems.

Prior to the establishment of the Penllergare Trust, the ecological status of the upper lake had become significantly degraded through a combination of excess siltation, invasive species and pollution. A concrete silt trap had been constructed above the upper lake, but by 1916, there was indication of sedimentation within the lake. In 1936 a large proportion of the area had been described as marshland from considerable siltation (David Clements Ecology, 2011). By the 1990s the Gabion weir was installed to form the new northern edge of the lake. Further siltation has formed an island along the middle of the lake (David Clements Ecology, 2011). This further isolated the lake waters, forming still water ponds and marsh microhabitats along the western margin and diverting the deep, flowing river channel along the eastern margin (David Clements Ecology, 2011). In addition, pollution events regularly occurred from disturbance and sewage systems in the upper catchment. For example, between 2004 and 2008, four potentially polluting events occurred on the river from discharge of sewage upstream and siltation as a result of works upstream. Since then, other potential pollution events causing increased water turbidity have also been speculated.

Table 1.0 defines the potential proposals for management and restoration set forth in 2008 (The Penllergare Trust, 2008). These encompass strategies to reduce the sedimentation through mechanically dredging the excess silt presently deposited. Following permission and funding acquired by a National Lottery Grant the Trust initiated the first phase of the dredging process in late July 2012. Fresh water lake sediment removal is usually undertaken to deepen a lake and increase its volume to enhance fish production, to remove nutrient rich sediment, to remove toxic or hazardous material, or to reduce the abundance of rooted aquatic plants. The first three objectives are usually met through sediment removal (Petersen, 1982). Disadvantages of dredging include cost, temporary phosphorus release from sediment, increased phytoplankton productivity, noise, lake drawdown, temporary reduction in benthic fish food organisms, the potential for toxic material release to the overlying water and potential for environmental degradation at the dredged material disposal site. The technique is recommended for deepening and for long range reduction of phosphorus release from sediment (Petersen, 1982).

Dredging is likely to disrupt a number of natural processes within both the lake and River Llan, potentially having acute and chronic negative implications on the biotic and abiotic conditions (Murphy *et al*., 1999). Consequently the Trust collaborated with WISE Ecologists from Swansea University, along with Access to Masters (ATM) students, to establish a baseline survey of the macroinvertebrates present within the lake and River Llan. Freshwater organisms have evolved morphological and behavioural adaptations to the aquatic environmental conditions such as flow rates, substrate, water chemistry and species interactions within their habitat (Giller and Malmqvist, 1998; Robinson *et al.,* 2002), and are now widely regarded as one of the best biological indicators of pollution in streams and rivers

(Wright *et al.*, 2007). Macroinvertebrates play crucial roles within freshwater systems such as in nutrient cycling and are key components of these ecosystems (Chapman *et al*., 1996; Solimini *et al.*, 2006). They include a number of different trophic guilds and consumer levels with some predatory taxa, foraging on other individuals such as Odonata larvae and others providing a highly productive food source for freshwater fish (Chapman *et al*., 1996; Solimini *et al.*, 2006; Harrison *et al.*, 2007). Macroinvertebrates are useful in monitoring the health of freshwater systems as they are impacted by ecosystem processes at multiple levels (Solimini *et al.*, 2006), are often highly abundant, show limited mobility and dispersal (Chapman *et al.*, 1996; Schumaker, 2007), and their lifecycles are variable with generation times lasting between two weeks and two years (van de Meutter *et al*., 2007). The long life-cycles of macroinvertebrates also make them useful indicators of historic conditions and both combined and varying pressures (Solimini *et al*., 2006). Another advantage of using macroinvertebrates is the ease of sampling (Chapman *et al.*, 1996; Schumaker, 2007). Capture of macroinvertebrates is simpler than capture of other freshwater biota such as plankton (Reynoldson and Metcalfe-Smith, 1992). It is also a cost-effective method of assessing freshwater system health (Chapman *et al*., 1996; Czerniawska-Kusza, 2005).

Multiple macroinvertebrate taxa (including Plecoptera, Ephemeroptera, Trichoptera, Gammaridae, Asellidae and Chironomidae) are now extensively used as indicators of ecosystem health as their absence suggests that pollution levels have increased (UNESCO/WHO/UNEP, 1996; Briers and Briggs, 2003; Czerniawska-Kusza, 2005). Taxa within the orders Ephemeroptera, Plecoptera and Trichoptera (EPT taxa) are highly pollution sensitive requiring neutral pH, cold water and high dissolved oxygen conditions (Schumaker, 2007). Freshwater crayfish, dragonflies and damselflies are moderately tolerant to pollution (Schumaker, 2007). Midge larvae and aquatic worms are pollution tolerant as individuals can tolerate low oxygen, warm water and variable pH (Schumaker, 2007). As sensitivity varies between taxa, abundances of each particular group can inform scientists about the water quality levels without analysis of water chemistry. The Biological Monitoring Working Party (BMWP) scoring system is used across the UK to assess water quality within freshwater systems, using presence/absence data of macroinvertebrate families (Czerniawska-Kusza, 2005). Families are all given a score based upon their sensitivity to organic pollution (Czerniawska-Kusza, 2005). For example, species in the Odontoceridae, which are very intolerant to pollution, are given high BMWP scores whereas families tolerant to pollution such as Chirinomidae are given low scores (Cota *et al.*, 2002). Samples are taken from the freshwater body of interest and the sum of BMWP family scores for taxa collected is calculated to give a total BMWP score for the site (Hawkes, 1998; Martin, 2004). This scoring system was employed within the present study to provide the baseline assessment of the habitat quality before dredging, and then repeated post-dredging to determine if there were any changes in environmental conditions.

In addition to sampling the freshwater invertebrates at Penllerger Valley Woods a number of abiotic factors (mean total dissolved solids, pH, temperature and conductivity) were also recorded to determine if they changed during the dredging process and assess if this would manifest as changes in biota. These abiotic factors give an indication of the water quality, which is also represented by the biota present due to their tolerance levels (UNESCO/WHO/UNEP, 1996).

It was predicted that dredging could potentially cause excess sedimentation and aggradation within the River Llan (Roni & Beechie, 2013) as well as release potential pollutants and nutrients in suspension that would have been locked within the sediment (Søndergaard *et al*., 2003; Zhong *et al*., 2008). This is likely to affect the macroinvertebrate communities present and be represented by a change in abundance, richness and/or water quality score (UNESCO/WHO/UNEP, 1996). The lake benthos at Penllergare Valley Woods prior to dredging was homogenous fine silt which does not support a high diversity of macroinvertebrates (Brooks *et al.*, 2005). There were however, a variety of microhabitats on the river bed providing a number of different ecological niches, allowing a diverse invertebrate community to develop (Edgar and Manning, 1997; Kayaba, 2004). Therefore, both during operation and post-dredging it was predicted that within the Lake itself changes would be observed in most abiotic factors due to the disruption and total loss of habitat, this would then ultimately be represented by changes in the biotic communities. In the following report the results and significance of the findings are presented and discussed, along with a number of future considerations and management recommendations to prevent further sedimentation.

Table 1.0 Management and conservation concerns at Penllergare Valley Woods and associated restoration proposals (Adapted from The Penllergare Trust, 2008)

Issue	Restoration Proposals
Lake siltation	Intrusive construction and dredging works at locations in the lake Identify potential silt deposition areas to confirm silt volume, whilst defining sensitive wildlife features that will need protecting, recreating or mitigation for damaging effects from dredging works Clear fell trees and coppice bank vegetation in addition to dredging and removing circa 18,000 m^3 of silt down to 4 m to restore the lower three-quarters of the upper lake up to the Gabion weir Reuse the weir for silt trap management
Pollution events	Take legal proceedings against Felindre up-site where clay pollution is thought to come down the catchment from Deal with an exposed sewage pipe near the waterfall
Vegetation control	Replant native deciduous woodland as part of the Better Woodland for Wales plan Manage invasive plants on river bank sides with 5 m of the river to improve light to water and views

2.0 MATERIALS AND METHODS

2.1 Study site

The study was conducted in June, July and August 2013 at Penllergare Valley Woods (OS coordinates; 51.6737, -3.9928, UK Gridref SS 623 991); a 100 ha temperate mixed broad-leaved woodland located within the River Llan Valley. The topography of the site encompassed Pennant sandstone bedrock overlain by clay and gravel (Norman, 2000; Parks and Gardens, 2007). The River Llan flowed southwards directly though the centre of the woodland from Melin-Llan to Cadle (Welsh Historic Monuments, 2000) and consisted of numerous tributaries feeding into the river, bringing water and occasional pollutants from other sites. The woods were dissected geologically by the River Llan, a second order river that formed part of the Loughor catchment. The river drained a mainly rural catchment as it flowed through northern Swansea, and laid on a large syncline in the Upper Carboniferous system in the Pennant sandstone series (Penllergare Trust, 2008; Parks and Gardens, 2013). Two lakes were artificially built in Penllergare Park in the 1840s; the upper lake and lower lake (Morris, 1999). The area of the upper lake was reduced during construction of the M4 Motorway and presently occupies approximately 9500 m² (Morris, 1999). The lower lake once covered 77000 m² and was controlled by a stone damn and sluice gate (Morris, 1999). The lake was drained in the 1990s forming rhos habitat, but was re-flooded in 2002 (David Clements Ecology, 2011). The upper lake flows into the River Llan, which continues until interrupted by the Lower Lake, and downstream to Cadle, eventually discharging into the Loughor estuary (Morris, 1999, Penllergare Trust, 2013a,b).

The woodlands consists of a matrix of different habitat types including plantation woodland, native woodland, cultivated gardens, lakes and wetlands as shown in Figure 2.1 (Nicholas Pearson Associates, 2008). The dominant habitat type was mature, broad-leaved woodland with large areas of native woodland and ornamental woodland around the upper lake. The dominant tree species consisted of Pedunculate Oak (*Quercus robur*), Ash (*Fraxinus excelsior*) and Sycamore (*Acer psuedoplatanus*) as well as Rhododendron, (*Rhododenderon ponticum*), Alder (*Alnus glutinosa*) and Bamboo (Bambuseae spp.) (Welsh Historic Monuments, 2000). A number of ancient woodland indicator species such as Opposite-leaved Golden Saxifrage (*Chrysosplenium oppositifolium*), Wood Sorrel (*Oxalis acetosella*) and Wood Anemone (*Anemone nemorosa*) were also present at the site (Rose, 1981).

Considerable quantities of the invasive species Rhododendron (*R. ponticum*) and Himalayan Balsam (*Impatiens glandulifera*) also dominated the site (Welsh Historic Monuments, 2000). Dense stands of Himalayan Balsam occur on the central sediment island and on the outer banks of the lake sampled.

Habitat specialists such as dippers (*Cinclus cinclus*) were present at the lake and river, as well as grey wagtails (*Motacilla cinerea*). Grey wagtails are listed on the UK Birds of Conservation Concern (BOCC) Amber List and dippers are on the Amber List for Wales (David Clements Ecology Ltd., 2011). Other bird fauna of the site include mallards (*Anas platyrhynchos*), song thrushes (*Turdus philomelos*), kingfishers (*Alcedo atthis*) and buzzards (*Buteo buteo*). Photographs taken by visitors and spraint collections at the site have provided evidence of an otter (*Lutra lutra*) utilising the lower lake (The Penllergare Trust, 2013b). During the study the upper lake was thought to support a number of fish species including European eel (*Anguilla anguilla*), lamprey (order Petromyzontiformes), Salmoniidae species and Cyprinid species that can tolerate low oxygen levels (Ostrand and Wilde, 2001; The Penllergare Trust, 2013b). The three British species of lamprey (river lamprey (*Lampetra fluviatilis*); brook lamprey (*Lampetra planeri*), and sea lamprey, (*Petromyzon marinus*) are important species for conservation having recently suffered significant losses (Maitland, 2003). The European eel is a particularly important species as it is listed as critically endangered on the IUCN Red List of Threatened Species following serious declines in recruitment in recent years (IUCN, 2013).

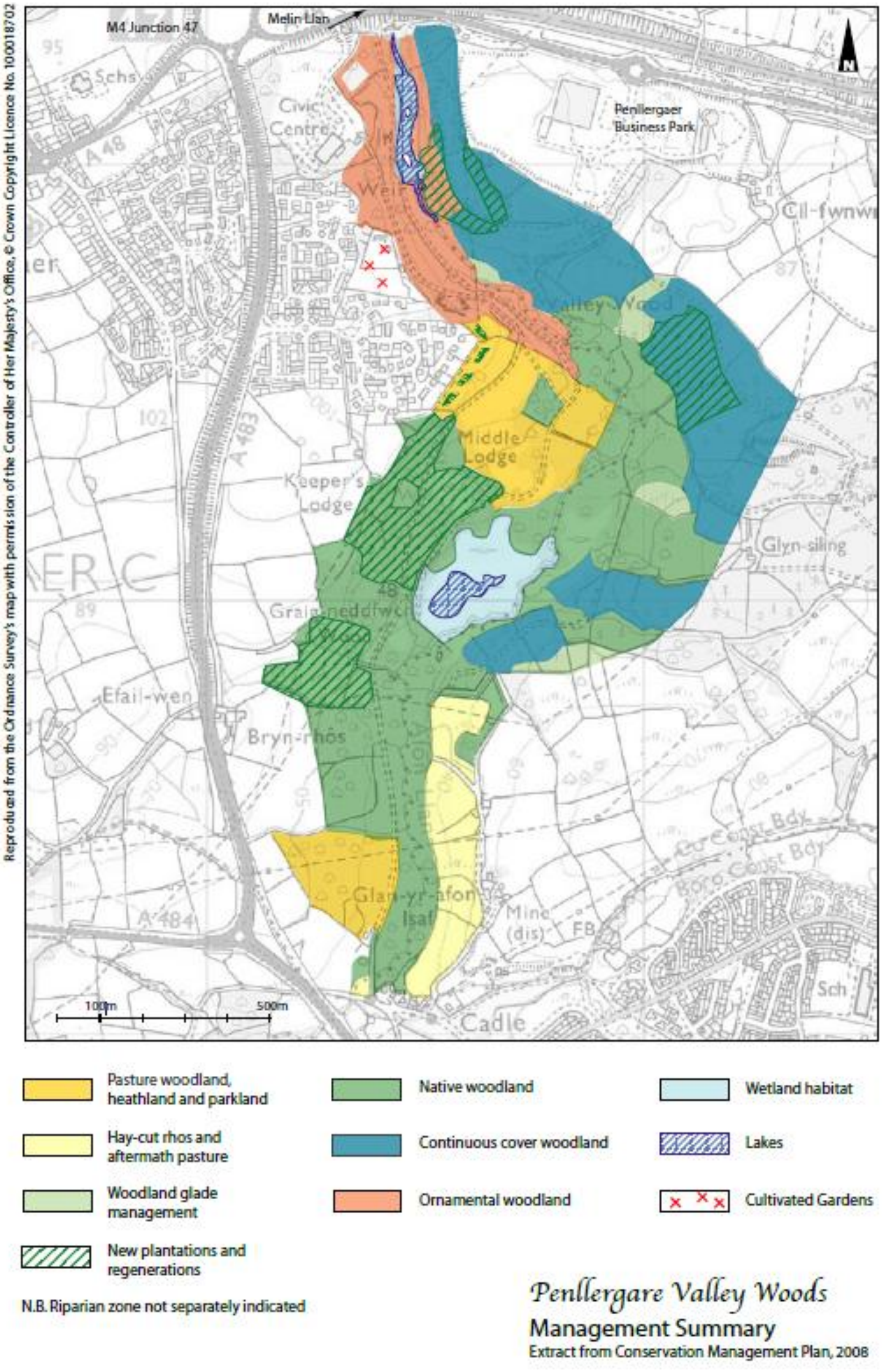

Figure 2.1 Phase 1 Habitat types at Penllergare Valley Woods site (from Nicholas Pearson Associates 2008).

2.2 Sampling protocol

2.2.1 Lake habitat description

The upper lake at Penllergare Valley Woods originally covered an area of approximately 13500 m² but had been reduced to 9500 m² at the time of study due to sedimentation and construction of the motorway (The Penllergare Trust, 2013a). It was approximately 300 m long and had an average width of approximately 35 m. During the time of sampling much of the lake was heavily silted and was undergoing succession into marshland (Murphy, 2006; The Penllergare Trust, 2013b). The lake was dissected by a central island of sediment, dominated by stands of Himalayan Balsam (*Impatiens glandulifera*). The east side of the lake consisted of a flowing river channel running directly from the River Llan inlet to the waterfall which adjoined the two study sites. The west side of the lake was more heavily subjected to siltation resulting in no visible standing water. Further south there were a number of small ponds which variable in size, depth and abundance of vegetation. At the south end of the lake, the water formed a heavily vegetated large pool (1.5 m depth). This area contained significant quantities of submerged vegetation including Horned Pondweed (*Zannichellia palustris*) and emergent vegetation including Branched Bur-reed (*Sparganium erectum*). The lake was dammed at the southern end where it led onto the waterfall (Morris, 1999). The depth of the upper lake was highly variable due to sedimentation but it ranged between 10 cm and 150 cm; the dredging work planned for September 2013 aimed to deepen the lake to 4 m.

2.2.2 River habitat description

For the purpose of the study the River Llan was surveyed from the southern point of the lake for 1 km. The lake cascaded over a waterfall approximately 3 - 4 m in width and 4.5 m high. The pool below the waterfall was roughly circular and was slightly dammed by large stone slabs (Welsh Historic Monuments, 2000). The riparian habitat consisted of occasional trees and scrubs (Welsh Historic Monuments, 2000) along with woodland plants, these were segregated by sections of open bank. Further south the east side of the valley changes to Larch (*Larix spp*.) plantation whereas the west bank gave way to sparsely growing Birch (*Betula spp.*) trees and Rhododendrons in the scrub layer (Welsh Historic Monuments, 2000). Beyond this point the banks give way to low-lying scrub with a bamboo grove covering both banks (Welsh Historic Monuments, 2000). The river consisted of a riffles/glide/pool matrix with a number of fish passes and weirs along its length (The Penllergare Trust, 2013b). The average width of the river was 5.4 m and the substrate predominantly consisted of pebbles but was also frequently interrupted by deep pools (90 cm) of silt and occasional boulders. The mean depth at the centre of the river was 25.8 cm. No macrophytes were recorded within the river.

2.2.3 Lake and river sampling sites

A total of 12 sites were sampled in the River Llan, one upstream of the upper lake and one 2 km downstream of the upper lake. The remaining 10 river stations were spaced equally over a 1 km stretch every 100 m downstream from the base of the waterfall. The spacing was kept equal to ensure that sampling was systematic and that there were no biases in terms of macroinvertebrate family abundances due to repeat samples of the same habitat. At each station kick samples (Sutherland, 2006) were taken for 3 minutes in the centre and edge of the river approximately 30 cm from the east bank (Figure 2.2.3.a). The microhabitat of each river sampling station was determined by assessment of the benthic features. Each river station was classified as a riffle, pool, glide or a combination of these. The Wentworth scale (Wentworth, 1922) was used to classify the sediment scale at each station.

Stations could not be spaced evenly in the lake due to limited accessibility and suitability for invertebrate sampling. Six stations were set up on the lake banks with an additional six stations on the island banks (Figure 2.2.3b). Regular sampling was employed to encompass as many of the habitat types and lake/river characteristics as possible. At each station a pelagic site and a benthic sample point were selected which were spaced 1 m apart for standardisation. By taking benthic samples 1 m from the pelagic sample, the impacts of the first collection on the second are minimised.

At each station a list of vegetation within 5 m of the sampling point was recorded in June 2013 (Rose, 1981). In June most species were in flower allowing for easier identification. The DAFOR scoring system was employed to allocate an abundance value for each species. Percentages were estimated for the amount of aquatic submerged vegetation, emergent vegetation, overhanging cover and the amount of bank covered by vegetation at each sample station.

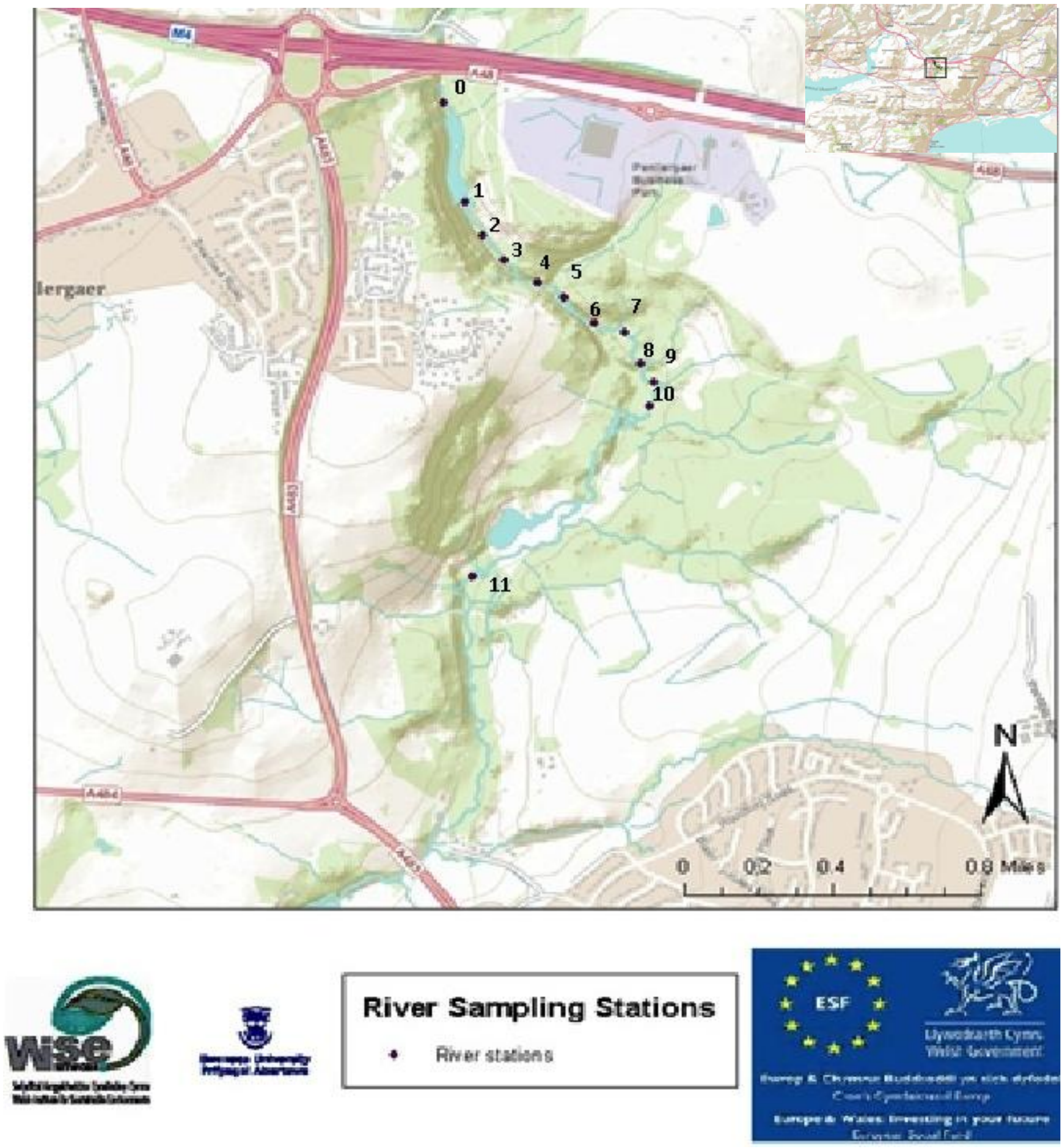

Figure 2.2.3a. Freshwater macroinvertebrate river sampling stations at Penllergare Valley Woods, 2013.

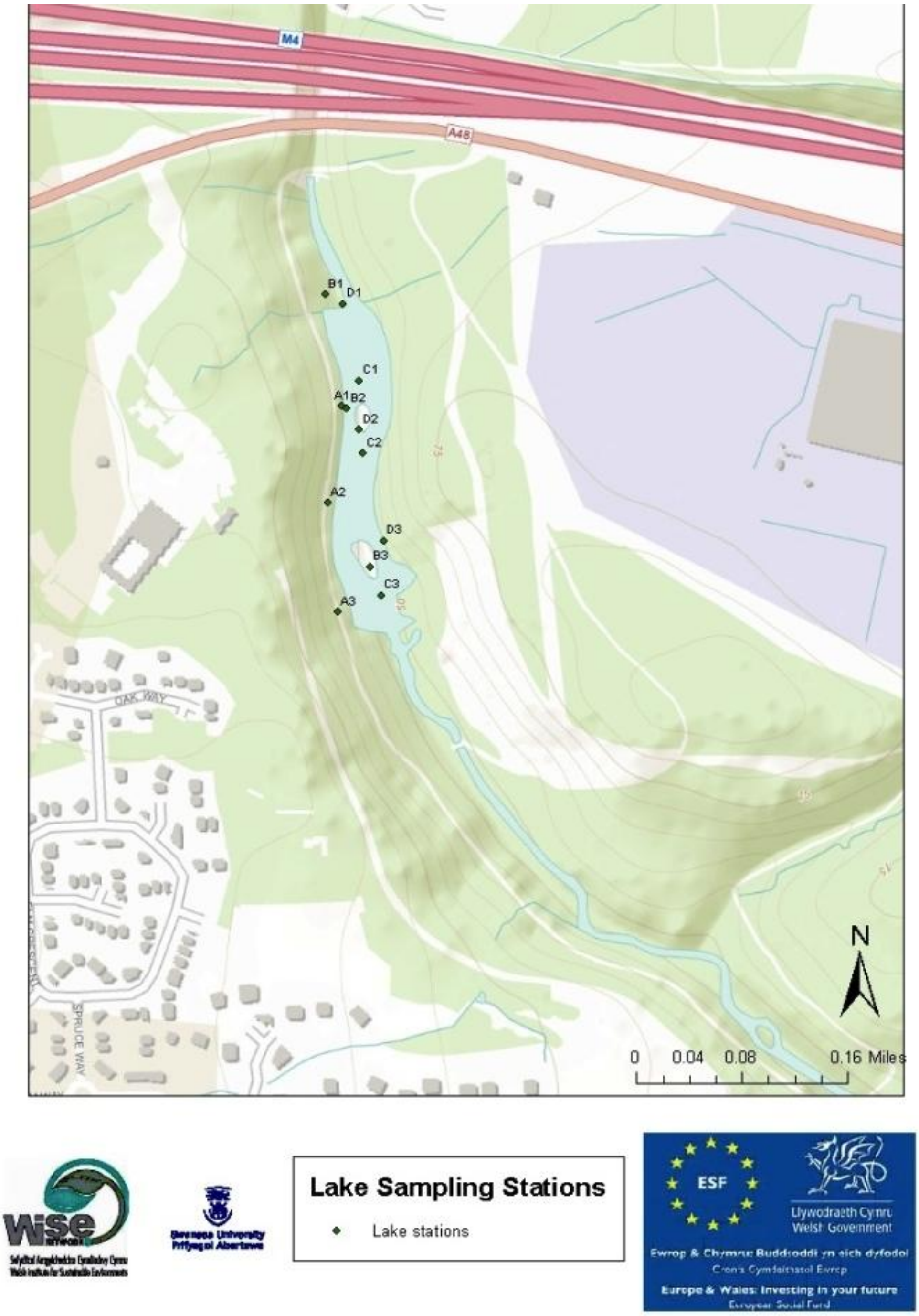

Figure 2.2.3b Map of lake sampling stations at Penllergare Valley Woods.

2.2.4 Freshwater macroinvertebrate sampling methods

Macroinvertebrate samples were collected from all 24 stations during June, July and August 2013. Weather conditions were recorded for each sampling day and no samples were collected in rain or poor weather conditions. This was because the activity of biota can be influenced by the weather with individuals seeking refuge during heavy rain conditions

(Student Watershed Research Project, 2013). Each sample site was visited in a random order to reduce the impacts of temporal variation (Gowing *et al.,* 2003). The time of sampling was recorded for each site as well as GPS coordinates (recorded using Garmin eTrex device). Prior to the collection of invertebrates temperature (°C), pH, total dissolved solids (TDS, ppm), conductivity (µS cm^{-1}) and dissolved oxygen (mg L^{-1}) were recorded using a Hanna HI98129 waterproof instrument and a Lutron Dissolved Oxygen Meter (PDO-519). For measurements of conductivity and total dissolved solids 60 ml pots were filled with water sourced from the lake or river and the HI98129 instrument was held in this water to obtain results. For pH, temperature and dissolved oxygen readings the HI98129 and PDO-519 were held directly in the water body. At each sampling point depth measurements were taken (cm). Conductivity is a measure of the water's ability to pass electron flow, these conductive ions come from dissolved salts and inorganic materials such as alkalis, chlorides, sulfides and carbonate compounds (also sodium, magnesium, potassium, calcium, and bromine). The more ions that are present the higher the conductivity. The concentration of these ions is heavily influenced by any pollution events where discharges enter the water body directly or reach the system through pollution of other streams or lakes within the catchment (Behar, 1997). Each freshwater system has a baseline value of conductivity that is determined by the ion content of underlying bedrock and soil (Behar, 1997). TDS combines the sum of all the ions within the system, and can include organic solutes (such as hydrocarbons and urea) in addition to the salt ions. Most freshwater species have specific tolerances within ranges of these ion contents and changes can cause shifts in biotic communities, limit biodiversity, exclude less-tolerant species and cause acute or chronic effects at specific life stages. High salt levels in water affect biota by dehydrating animals (Giller and Malmqvist, 1988). Excess of these ions is generally caused by eutrophication from agricultural fertilisers and run-off, sewage and/or pollution, whereas limitations generally occur during oligotrophic condition (Brönmark & Hansson, 2005). Different biota have different pH tolerances with some invertebrate taxa surviving at pH values as low as 3 and others surviving at higher pH values of 8.2 (Johnson *et al.,* 1999). Values of pH have been found to considerably impact macroinvertebrate community structure, particularly in small freshwater bodies (Nicolet *et al.,* 2004; Biggs *et al.,* 2005). Chalk, alkaline streams such as the River Llan exhibit particularly diverse and characteristic flora and fauna due to the alkalinity of the system (Lewis *et al.,* 2000).

The sampling methodologies employed for the collection of macroinvertebrates were different for the lake and the river. At the river stations a kick sampling methodology was used (Sutherland, 2006). A D-shaped net with a mesh size of 2 mm was positioned within the water column, in contact with the river bed with the sampler in front facing the direction of water flow. The sediment was then disturbed by means of kicking up stones and silt for 3 minutes. All macroinvertebrates and sediment dispersed through the water then entered the net due to river flow.

In the lake the net was used to collect pelagic and benthic samples at each station. Pelagic samples were collected by moving the net in a figure-of-eight motion over a length of 1 m for 3 minutes. The speed and length of sweep were kept constant to ensure the reliability of results (Sutherland 2006) and efforts were made to avoid contact with the bottom of the lake. This was to minimise any collection of lake benthos in pelagic samples. Benthic samples were taken 1 m to the right of the pelagic sample (measured with 1 m ruler). The net was thrust into the mud ten standardised times with equal force to disperse the benthos within the water column. This sediment and dispersed macroinvertebrates were then collected with the net. Following collection in both systems all macroinvertebrates were placed into 60 ml pots of antifreeze with commercial antifreeze. Collection of macroinvertebrates from the trays was not timed as this would lead to bias and a skew towards those taxa more easily found or those that were most abundant. Any vertebrates collected, such as 3-spined sticklebacks (*Gasterosteus aculeatus*), vegetation, detritus or gastropods were released back at the point at which they were collected. All samples were identified at Swansea University. Individuals ≥ 0.3 mm were identified to family level where possible, although freshwater shrimp and water slaters were identified to species level. A number of taxonomic keys were used to aid this identification (Croft, 1986; Gledhill *et al.*, 1993; Wallace, 2006; Pryce *et al.*, 2007; Sutton, 2008; Cham, 2011). Certain Dipteran larvae and Coleopteran larvae however, could only be identified to order level.

Individuals were examined using a Kyowa SDZ-PL microscope at x 20 and x 180 magnification. Photographs were taken to provide a visual record of all families observed and for use as a reference when identifying other samples. Family names and abundances were recorded and specimens were returned to pots of antifreeze for storage and future species-level identification.

2.3 Statistical analysis

The mean abiotic factors within the Lake and River were compared using an independent samples *t*-test whereby pH, temperature, conductivity, and TDS were the dependent variables and habitat (i.e. river and lake) were the independent variables. Chi squared (χ^2) tests for association were employed to determine if the total number of families, total abundance and BMWP score differed between the river and the lake. In addition, the effects of dredging over time were analysed on abiotic factors utilising a One-way Analysis of Variance (ANOVA), with survey number as the independent variable and pH, temperature, conductivity and TDS as the dependent variables. As family number and invertebrate abundance were classed as frequency data non-parametric Kruskal-Wallis tests were employed to determine the effect of dredging on the biota, here survey number was the independent variable and number of families and abundance were the dependent variables. BMWP score was totalled utilising the sum of values allocated to each Family in accordance with (Hawkes, 1998, available at: https://www.fba.org.uk/sites/default/files/BMWPLIFEtaxa_Modified.pdf).

It must be stressed however that significant flaws reside when comparing the lake and river as these are different habitats and were subject to different sampling techniques. The statistics comparing each habitat show here are therefore purely for visual purposes within the present report.

3.0 Results

3.1 Summary of the Lake and River abitioc factors

Table 3.1 shows the mean abiotic conditions for both the lake and the river over the sampling period. Significant differences (highlighted in bold) were observed between the lake and River Llan in mean pH, dissolved oxygen, conductivity and total dissolved solids (TDS), however, temperature was similar between the two habitats (Table 3.1) over the study period. The lake had a higher concentration of TDS than the river, but higher concentrations of dissolved oxygen, greater conductivity and higher pH levels were recorded in the river (Table 3.1, Figure 3.1a - d).

Table 3.1. Mean abiotic conditions for the River Llan and Upper Lake in Penllergare Woods over the three month sampling period in 2013 (± SD).

Habitat	pH	Temperature (°C)	Dissolved O_2 (mg/L)	Conductivity (µS)	TDS (ppm)
River	7.63 (0.11)	15.74 (1.11)	9.64 (4.62)	251.83 (43.54)	125.11 (23.01)
Lake	7.23 (0.36)	16.01 (2.55)	7.23 (4.00)	184.22 (57.00)	382.24 (128.6)
T	-5.039	0.637	-3.75	-5.867	11.2
df	40.7	46.8	24.3	64.3	36.3
Sig	**0.001**	0.528	**0.001**	**0.001**	**0.001**

3.2 Lake and River biotic factors

A total of 42 freshwater macro-invertebrate families were identified over the three month surveying period in Penllergare Valley Woods freshwater bodies, of these 29 were recorded in the lake and 36 were recorded in the river ($\chi^2_1 = 0.3$, $P = 0.534$, Table 3.2, Figure 2.1f). The total abundance of these families was not distributed evenly for the river ($\chi^2_{35} = 58575$, $P < 0.001$) or the lake ($\chi^2_{28} = 17402$, $P < 0.001$) with the majority of organisms recorded within the Leuctridae (Rolled winged stonefly), Chironomidae (non-biting midges) and Ephemerellidae (mayfly) for the river and the Chironomidae, Corixidae (Lesser waterboatmen) and Asellidae (Water hog louse) for the lake. The least recorded organisms were in the Aphelocheiridae (Water bug), Erpobdellidae (Leech) and Ptychopteridae (Phantom caddisfly). A total of 7050 individuals were sampled and the total BMWP score for the site was 240.3. A significantly different number of individuals were sampled between the upper lake (N = 5242) and the River Llan (N = 1808, $\chi^2_1 = 1672$, $P < 0.001$, figure 2.1e), as would be expected within these different habitats. The BMWP sore also differed significantly with a score of 148.5 recorded for the lake and 220.4 in the River Llan ($\chi^2_1 = 14.04$, $P < 0.001$).

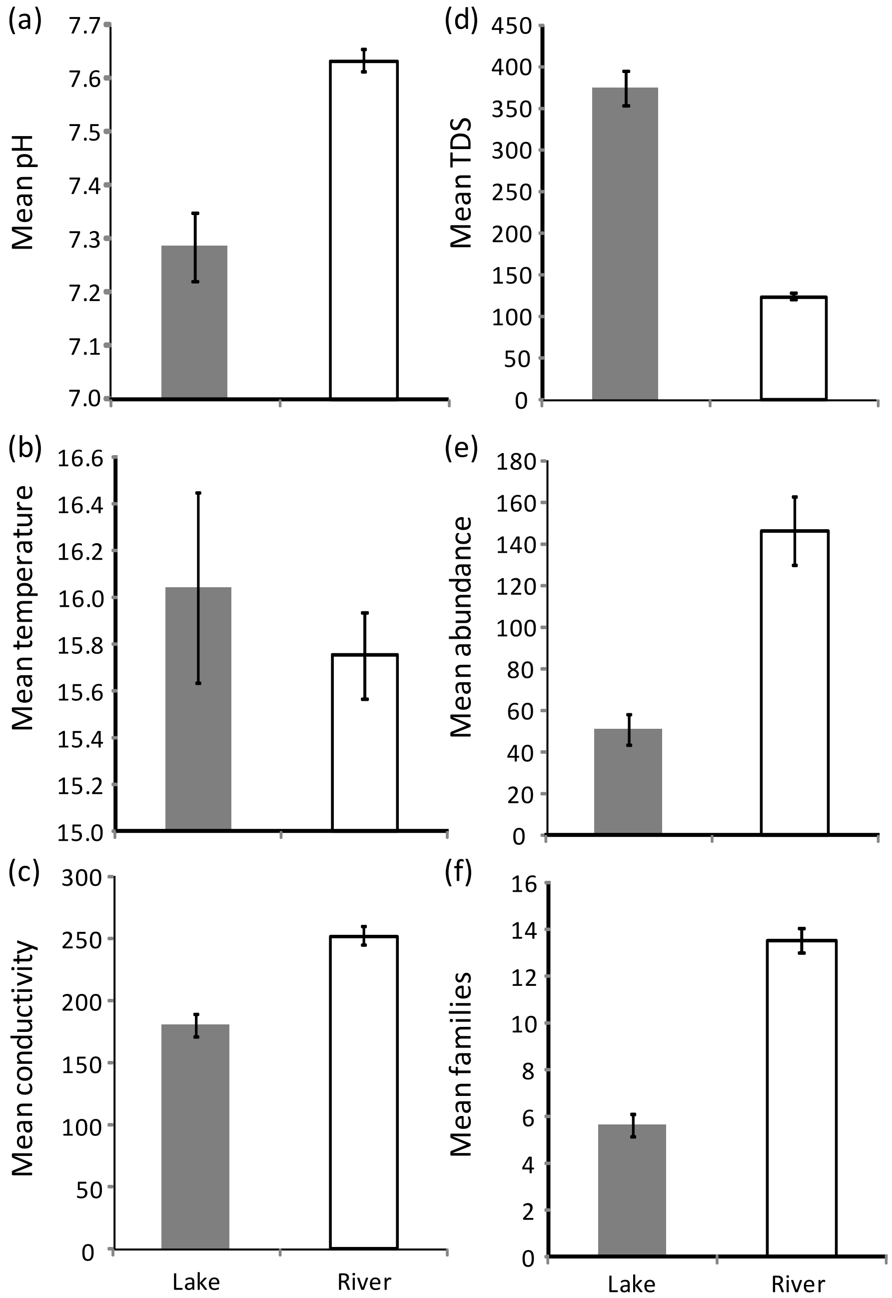

Figure 3.1. Comparison of the mean abiotic and biotic factors (± SE) recorded between July and September 2013 at Penllergare Valley Woods, 2013

Table 3.2. Summary of the families, total abundance and BMWP score of macro-invertebrates captured over the three month surveying period at Penllergare Valley Woods.

Family	Common name	Lake	River	Total abundance	BMWP sore
Aeshnidae	Southern Hawker	10	0	10	6.1
Aphelocheiridae	Water bug	0	1	1	8.9
Asellidae	Water hog louse	152	51	203	2.1
Baetidae	Mayfly	33	275	308	5.3
Caenidae	Square-gilled mayfly	0	21	21	7.1
Capniidae	Winter stonefly	0	168	168	10
Chaoboridae	Phantom midges	8	2	10	0
Chironomidae	Non-biting midges	1049	293	1342	3.7
Chloroperlidae	Green stonefly	0	8	8	12.4
Coenagrionidae	Large red damselfly	8	0	8	3.5
Corixidae	Lesser waterboatman	232	0	232	3.7
Crangonyctidae	Shrimp	8	161	169	0
Dixidae	Meniscus midge	129	12	141	0
Dytiscidae	Diving beetle	10	70	80	4.8
Ecdyonuridae	Mayfly	0	260	260	9.8
Elmidae	Riffle beetle	9	493	502	6.4
Ephemerellidae	Mayfly	16	1247	1263	7.7
Ephemeridae	Mayfly	3	29	32	9.3
Erpobdellidae	Leech	1	0	1	2.8
Gammaridae	Shrimp	8	52	60	4.5
Glossiphonidae	Leech	2	6	8	3.1
Haliplidae	Water beetle	27	1	28	4
Hydrophilidae	Water scavenger beetle	0	2	2	5.1
Hydropsychidae	Caddisfly	0	110	110	6.6
Hygrobiidae	Screech beetle	12	6	18	2.6
Leptophlebiidae	Prong-gilled mayfly	8	37	45	8.9
Leuctridae	Rolled winged stonefly	22	1675	1697	9.9
Nemouridae	Spring stonefly	0	2	2	9.1
Notonectidae	Backswimmer	18	0	18	3.8
Odontoceridae	Caddisfly	19	74	93	10.9
Perlodidae	Perlodid stonefly	1	68	69	10.7
Philopotamidae	Silken tube-spinner caddisfly	7	7	14	10.6
Piscicolidae	Leech	0	5	5	5
Polycentropodidae	Caddisfly	0	5	5	8.6
Ptychopteridae	Phantom caddisfly	0	1	1	0
Rhyacophilidae	Caddisfly	1	52	53	8.3
Sericostomatidae	Bush-tailed caddisfly	0	2	2	9.2
Sialidae	Alderfly	5	1	6	4.5
Simuliidae	Black fly	1	39	40	5.8
Thaumaleidae	Solitary midges	7	5	12	0
Tipulidae	Cranefly	2	1	3	5.5
Total Family		29	36		
Total abundance		1808	5424	7050	
Total BMWP		148.5	220.4		240.3

3.3 Effects of dredging on the abiotic and biotic conditions within the upper lake and River Llan

Over the three month study period there was no significant change in any of the abiotic factors recorded within the River Llan, except temperature, suggesting the dredging did not impact the lower river system (Table 3.3.). Furthermore no significant change was observed in either the number of families or abundance of individuals captured (Table 3.3) post-dredging. Within the lake a number of changes occurred as pH (Figure 3.3a), conductivity (Figure 3.3b) and TDS (Figure 3.3c) decreased following dredging (Table 3.3, Figure 3.3). Other factors such as temperature and family number and abundance did not change however (Table 3.3).

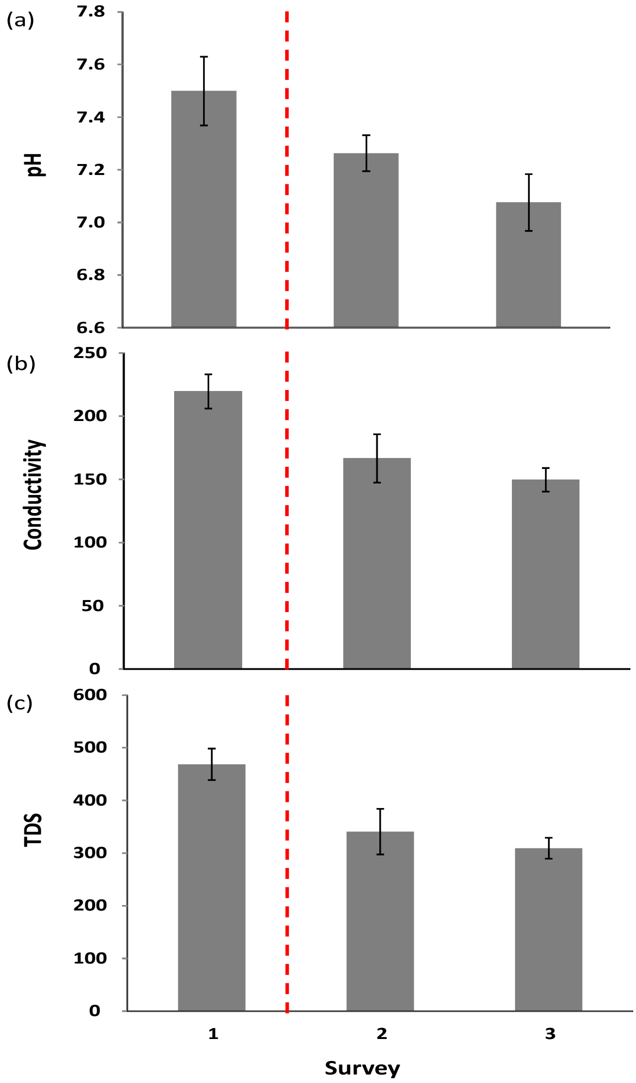

Figure 3.3 Significant changes in abiotic conditions following dredging (red line) Upper Lake within Penllergare Woods in 2013

Table 3.3 Statistical analysis of the changes in abiotic and biotic factors in the River Llan and Upper Lake in Penllergare Valley Woods in 2013 (figures in bold represents a significant difference of 95 % confidence).

	pH	Temperature	Conductivity	TDS	Abundance	Families
River	$F_{2,35} = 0.39$ $P = 0.681$	$F_{2,35} = 20.70$ $\boldsymbol{P < 0.001}$	$F_{2,35} = 0.32$ $P = 0.731$	$F_{2,35} = 0.099$ $P = 0.906$	K-W = 1.585 $P = 0.395$	K-W = 2.285 $P = 0.319$
Lake	$F_{2,34} = 3.99$ $\boldsymbol{P = 0.028}$	$F_{2,34} = 1.92$ $P = 0.163$	$F_{2,34} = 6.13$ $\boldsymbol{P = 0.006}$	$F_{2,34} = 6.55$ $\boldsymbol{P = 0.004}$	K-W = 2.699 $P = 0.259$	K-W = 0.287 $P = 0.866$

4.0 Discussion

The findings of the present study suggest the River Llan and associated upper lake were of excellent ecological quality (value > 100 are classified as excellent, Hawes, 1998) when sampling commenced. The number and diversity of macroinvertebrate families exceeded 30, with over seven thousand individuals sampled. These findings compare well to other freshwater habitats within the region of South Wales (Hirst *et al*., 2003) suggesting that a number of freshwater systems are now within range of the specified targets set within the Water Framework Directive (WFD, 2000). The productive invertebrate population can be directly related to the abiotic conditions recorded (pH, TDS, conductivity) which were all found within the optimal range for most species. As would be expected, the river supported a greater diversity than the lake and differences were observed in key abiotic factors (Giller & Malmqvist 1999; Brönmark & Hansson, 2005).

4.1 Effects of dredging on the biotic and abiotic conditions

The rate of sediment transport relative to the rate of sediment supply to a reach (termed the relative transport rate) determines whether any reach system is accumulating or exporting sediment, or is relatively stable. If sediment supply exceeds transportation, it results in aggradation (increase in land elevation due to sedimentation), whereas sediment supply lower than transport capacity results in the degradation of the river bed (Roni & Beechie, 2013). Changes in activities within the river system can result in increasing sediment movement and alters rivers to the oversupplied or aggregated state, however, these changes are generally short term unless the activity is prolonged. As the ratio of sediment to transport capacity increases, fine sediment can be accommodated through fining of the bed surface, pool filling, channel aggradation and channel widening. During the dredging process the River Llan appeared unaffected by the disturbance of the lake. No changes were observed in abiotic conditions (TDS, pH, conductivity) except temperature (however, this is likely to be a factor of seasonality as opposed to disturbance). Importantly, the biota did not change in either richness or abundance. Flow rates impact the habitat within a river (Bunn and Arthington, 2002) causing the build-up of soft silt sediment at the river edge compared to the cobble/boulder substrate in the middle. Firm substrates are preferentially colonised by macroinvertebrate taxa therefore this difference in substrate results in particularly high numbers of macroinvertebrates in areas with cobble/boulder substrate (Brooks *et al.*, 2005).

The instability of sandy sediments has been hypothesised as an important cause of reduced macroinvertebrate abundance and diversity in other studies (Giller and Malmqvist, 1998). Fine sediment also increases macroinvertebrate drift by filling the interstitial spaces between larger substrate elements preventing individuals from clinging on and avoiding being flushed down the river (Wood and Armitage, 1997). As there was no change in the biota post-dredging this suggests that the relative transport rate of the river was relatively stable and little degradation or aggradation occurred, therefore silty deposits did not build up to a level whereby it affected the macroinvertebrate abundance. It must be noted, however, that this study only continued for two months post-dredging therefore long term effects or changes to the system cannot be ruled out and future, long-term monitoring is recommended, particularly if dredging is to be repeated.

Conversely, within the upper lake sedimentation was likely to be occurring due to the low relative transport rate to sedimentation, as the net load of sediment exceeded the net loss (Roni & Beechie, 2013). There are a number of mechanisms that cause excess sedimentation within lotic habitats (Janse, 1997; Brönmark & Hansson, 2005). Sewage or deposits of fertiliser from agricultural run-off can lead to eutrophication whereby the increased loading of phosphorus enables the significant growth of primary producers (Søndergaard *et al*., 2003; Zhong *et al*., 2008). These communities rapidly become dominated by cyanobacteria communities which produce algal blooms that reduce the water transparency (Gulati & van Donk, 2002; Zhong *et al*., 2008). The bacteria that break down the organic matter require oxygen to do so, therefore the water can often become oxygen depleted and fish kills become common, furthermore grazing pressure is low due to the inedibility of cyanobacteria (Janse, 1997) creating trophic shifts. Sedimentation then increases due to the accumulation of dead microorganisms, plus the associated sedimentation. The second means of increased sedimentation occur either due to large landslides or the continued erosion of agricultural land adjacent to the freshwater habitat (Roni & Beechie, 2013). Within the upper lake the latter is likely to be the source of the siltation of the lake as no algal blooms have been reported and the lake is heavily dominated by macrophytes (Barko & Smart, 1986; Roni & Beechie, 2013). In order to prevent further sedimentation a number of management strategies can be implemented and are discussed in section 4.2.

No changes were observed in macroinvertebrate family richness or abundance proceeding dredging in the lake or river, yet changes in pH, TDS and conductivity were recorded for the lake, with abiotic factors reducing with each consecutive sampling session. The ranges of pH, conductivity and TDS did not exceed those that can become toxic to most freshwater organisms (pH: 6.5 – 8.2, conductivity: 200 – 1000 µS cm^{-1}, TDS: 0 – 1,340 mg L^{-1}) which explains the lack of change in macroinvertebrates observed.

4.2 Management recommendations

The frequency of dredging is highly dependent on the desired outcome (CIRIA, 1997). In lakes it is generally advised that dredging be kept to a minimum to permit colonisation of a variety of different species (CIRIA, 1997). Frequent dredging is advised should rare species be present to allow optimum conditions for their survival (CIRIA, 1997). Results from this study suggest that dredging should be repeated until the desired depth is achieved. However, unless the source of the sedimentation is suppressed it is likely that repeated dredging may need to be more frequent. This may impose considerable economical and environmental constraints, therefore prior to investing more resources into sediment removal a cost benefit matrix should be considered, whereby clear aims and objectives for why both time and money should be investigated in the dredging process. This should encompass the ecological, cultural and aesthetic values and also consider if such treatment would enhance environmental quality, connectivity, resilience and ecosystem services.

Pollutants from a number of sources (industrial, agricultural, mining, municipal sewage) are likely to have entered the catchment system within Penllergare Valley woods (Mulligan *et al*., 2001). The sediment within the lake can act as a sink for toxins (Kim *et al*., 2003). Dredging has the potential to re-suspend contaminants that were previously locked within the sediment into the freshwater system (Murphy *et al*., 1999; Kim *et al*., 2003; Hickey, 2009), releasing toxins such as sulphides and heavy metals (such as copper, lead, zinc and metalloid arsenic). These can become damaging to small organisms both directly (through ingestion, absorption via dermal contact or bioaccumulation from prey, Mulligan *et al*., 2001) and indirectly (reduced, stunted or retarded growth of primary producers, Barko & Smart 1986). Monitoring the heavy metal concentrations within the water column of both the lake and the river would therefore be recommended to identify possible disruption and transportation to the fluvial system post-dredging (Zong *et al*., 2008; Peng *et al*., 2009).

In order to prevent further accumulation of sediment, the source should be identified and measures undertaken to prevent or reduce additional deposits (Murphy *et al*., 1999; Roni & Beechie, 2013). Erosion of fine sediment on cultivated land through wind or precipitation is often the source of excess sedimentation within rivers and lakes (Ministry of the Environment, 2001). The increase in sedimentation and runoff from agricultural land can be damaging to both aquatic systems and degrade soil quality and ultimately productivity. Therefore the benefits to preventing excess runoff are twofold and taking precautions and enforcing effective management strategies can contribute not only direct benefits to habitat quality, but also enhance ecosystem services and contribute towards ecosystem resilience. Commonly fostered methods utilised to reduce water and wind related erosion encompass crop rotation and selection, contour planting, conservation tillage, leaving crop residue, or the addition of organic matter (Toy *et al*., 2002). In general these are combined with supportive practices such as contouring or orientating ridges or crop rows perpendicular to the runoff and wind direction to subdue erosion (Toy *et al*., 2002). Silt fences, vegetative strips or riparian buffers and straw bales also attempt to minimise or filter sediment (Sather & May, 2008). Buffers with mixed native vegetation have been shown to be the most effective at reducing erosion and filtering nutrients and fine sediment (Sather & May, 2008). In order to determine the likely sources, the partial sediment budget supply can be employed (Roni & Beechie, 2013). This aims to quantify changes in sediment supply by either quantifying erosion rates from aerial images or extrapolating empirical data or modelling (Roni & Beechie, 2013).

In lakes, ponds and rivers, the invasion of exotic species has resulted in the loss of biological diversity, changes in community structure and ecosystem functioning (Gurevitch & Padilla, 2004; Brönmark & Hansson, 2005). The sensitive nature of aquatic habitats to invasions by exotic species is exacerbated by human activity and can also be rapid because dispersal relies on passive transport (Molnar *et al*., 2008; Zanden & Olden, 2008). Invasive species can impact on an individual level by alterations in behaviour through competition and predation (Redriguez, 2006), or affect genetics and natural selection by altering gene flow through hybridisation (Facon *et al*., 2005). At the population level, invasive species can alter size and age structure, population growth, density, distribution, and can drive populations to extinction, thus altering community structures and causing trophic cascades. Nutrient exchange, resource acquisition, disturbance regimes and changes in physical structure can occur through invasive species on the ecosystem level (Gordon, 1998). These alterations to natural systems not only cause substantial impacts on biota, but also cause significant economic losses, encompassing losses to goods and services and as well as incurring costs for control (Pimental *et al*., 2005). Aquatic weeds that become invasive typically have rapid vegetative growth and high dispersal rates and can cause significant changes to ecosystem functioning (Gordon, 1998). The submerged plant Canadian Waterweed (*Elodea canadensis)* can cause significant eutrophication and lead to domination by Common Reeds (*Phragmites australis)* (Brönmark & Hansson, 2005). This species was found during the present study and removal should be considered as part of the rehabilitation management plan. Furthermore, a significant quantity of the non-native, invasive species Himalayan Balsam (*Impatiens glandulifera*) was recorded in dense stands across the riparian habitat on both the lake and the river. Himalayan Balsam is becoming of increasing concern in UK freshwater habitats (Oreska & Aldridge, 2011) due to its highly invasive traits as it is highly fecund and has explosive seed dispersal and rapid growth, allowing it to out-compete native vegetation (Sheppard *et al*., 2006; Booy *et al*., 2013). Additionally, it can affect ecosystem functioning and services due to its shallow buttress roots, which increase sedimentation and runoff and degrade the riparian corridor (Bartomeus *et al*., 2010; Booy *et al*., 2013). Potentially dredging the silt within the River Llan could enhance further colonisation of Himalayan Balsam by mechanical, human driven dispersal of the seed bank to the disposal site and downstream (Murphy *et al*., 1999). Therefore an assessment of the seed bank within the substrate to fully predict colonisation on the banks (Olano *et al.*, 2005) should be undertaken. The abundance of Himalayan Balsam seeds will indicate the level of treatment required to prevent a complete dominance and further colonisation of this invasive species. This will include regular hand pulling and treatment of germinating seedlings with a weed wipe.

While no short term changes were detected throughout the dredging process in the present study, chronic long-term changes cannot be ruled out. Therefore it is recommended that repeated annual sampling occur for the initial few years, particularly if dredging is to be repeated (Murphy *et al*., 1999). Additionally, due to time restraints, the macroinvertebrates sampled were only identified to family level. While this is sufficient for determining habitat quality (UNESCO/WHO/UNEP, 1996), it can greatly underestimate the significance of the rarity, community structure and the ecological functional roles of the animals present. Correctly identifying these can enhance management practices and also ensure the correct legal designation is allocated to the habitat. Therefore future sampling should consider identifying individuals to species level where possible.

4.2.1 Condition indicators

The condition indicators listed in Tables 4.2.1a and 4.2.1b provide specific guidelines for management of the site. They detail targets for maintenance of good quality habitat in both systems (Hurford *et al.,* 2009). Definitions are also provided for what constitutes good quality. These tables were constructed using data obtained during this study. The undertaking of an ecological baseline study prior to dredging is recommended as part of the "best practice" protocol and ensure that the impacts and recovery following the work can be accurately assessed (Inland Waterways Advisory Council, 2008). Regular and consistent monitoring is required to assess the impacts of this management on macroinvertebrate community assemblages, water quality and ecosystem diversity over the short, medium and long-term. This study provides a baseline assessment and a framework to monitor any changes. If monitoring indicates species diversity has reduced and key species are absent inoculation by transferring species from the lower lake may be beneficial. This could be done by use of deployment of colonisers and then transplanting them to similar habitat locations in the impacted areas.

Table 4.2.1a Condition indicator table for the management of the river at Penllergare Valley Woods.

<table>
<tr><th>Condition indicator table</th><th colspan="3">The river at Penllergare Valley Woods will be in optimum condition when</th></tr>
<tr><td>Habitat extent</td><td colspan="2">Lower limit</td><td>Extent of river identified on remote sensing image</td></tr>
<tr><td>Habitat quality</td><td colspan="2">Lower limit</td><td>BMWP score >50
pH range between 6.5 mg/L and 10 mg/L
Conductivity between 150 and 500 µS
Total Dissolved Solids between 50 and 250 mg/L
Total dissolved oxygen levels between 7 and 11 mg/L for fish and between 4 and 7 mg/L for most macroinvertebrates
Number of macroinvertebrate families >40
Individuals of the families Ephemeroptera, Plecoptera and Trichoptera (EPT) present
Presence of microhabitats of conservation value-debris, fallen trees, dams, riffles</td></tr>
<tr><th colspan="4">Site specific habitat definitions</th></tr>
<tr><td colspan="2">Broad habitat name : River</td><td colspan="2">Site specific definition of broad habitat–River with pool, riffle and glide microhabitats and boulder/cobble substrate.</td></tr>
<tr><td colspan="2">Habitat class name : Boulder/cobble dominated river</td><td colspan="2">Site specific definition of habitat class-at least 41 families present in 10 orders including EPT taxa. Frequent families include Stonefly larvae (Plecoptera); Leuctridae, Mayfly larvae (Ephemeroptera); Ephemerellidae, Baetidae, Ecdyonuridae, Caddisfly larvae (Trichoptera); Odontoceridae, Hydropsychidae, Beetles (Coleoptera); Elmidae, True fly larve (Diptera); Chironomidae, Malocrustacan Crustaceans (Amphipoda); Gammaridae and Peracarid Crustaceans (Isopoda); Asellidae.</td></tr>
</table>

Table 4.2.1b Condition indicator table for the management of the lake at Penllergare Valley Woods.

Condition indicator table	The lake at Penllergare Valley Woods will be in optimum condition when	
Habitat extent	Lower limit	Extent of lake identified on remote sensing image
Habitat quality	Lower limit	BMWP score > 40 pH range between 6.5 mg/L and 10 mg L^{-1} Conductivity between 150 and 500 µS Total Dissolved Solids between 50 and 250 mg L^{-1} Total dissolved oxygen levels between 7 and 11 mg/L for fish and between 4 and 7 mg L^{-1} for most macroinvertebrates Number of macroinvertebrate families >34 Presence of microhabitats of conservation value-debris, fallen trees, variation in substrate composition, shelter for macroinvertebrates from predators i.e. fish
Site specific habitat definitions		
Broad habitat name : Lake	Site specific definition of broad habitat–Large body of water heavily silted on west side with numerous heavily vegetated small ponds and a flowing river channel on the east side.	
Habitat class name : Large lake with central sediment island and waterfall at southern end	Site specific definition of habitat class–At least 34 families present in 12 orders including EPT taxa. Frequent families include True Bugs (Hemiptera); Corixidae, True Flies (Diptera); Chironomidae, Peracarid Crustaceans (Isopoda); Asellidae and Beetle (Coleoptera) larvae.	

4.2.2 Mitigation to enhance the lake for macroinvertebrates

It is difficult to manage macroinvertebrate communities for conservation directly, therefore it is considered simpler and more effective to manage their habitat (Maitland and Morgan, 1997). Following the dredging process the lake was transformed into a different habitat due to the change in depth, size, bank structure and topography of the benthos. The probability of high rates of recolonisation and the restoration of high biodiversity are increased with adequate mitigation measures (Merz and Ochikubo Chan, 2005). It is important that efforts are made to prevent the loss of unique and varied lake microhabitats as these increase habitat heterogeneity, promoting macroinvertebrate diversity and abundance. Macroinvertebrate abundance and diversity was also higher at shallower depths, however, the dredging of the lake will result in a much deeper lake. It is therefore important to construct shallower areas to allow colonisation by a variety of macroinvertebrate taxa (Maitland and Morgan, 1997). The continued presence of sections of the island would also have positive impacts on macroinvertebrate diversity (Maitland and Morgan, 1997).

The de-siltation of the lake will leave a homogenous, fine sediment substrate so, to increase the diversity of macroinvertebrates in the lake; various types of substrate should be added to the lake (Maitland and Morgan, 1997; CIRIA, 1997). Habitat diversity is directly correlated with macroinvertebrate diversity, therefore it should be maintained. Habitats of conservation value such as fallen trees, areas with substantial debris and areas with abundant emergent vegetation should also be maintained (Borchardt, 1993; Maitland and Morgan, 1997). The addition of woody debris will decrease the severity of macroinvertebrate drift, thus limiting the number of organisms being rapidly washed out of the system (Borchardt, 1993). Large stones and/or concrete slabs can also provide hard substrates and therefore increase the habitat structural complexity of the new lake creating a greater number of niche opportunities (Mormul *et al.*, 2011).

5.0 Conclusion

Within the present study, The Penllergare Trust has made considerable efforts to improve and enhance the habitat quality, biodiversity and conservation of the freshwater bodies under their management. The present collaboration has produced a baseline assessment of the habitat conditions prior to dredging, as well as monitoring abiotic and biotic factors immediately post disturbance. This methodology has provided a foundation to compare future changes in ecological conditions and represents evidence of best practice which should be employed in future Priority habitat restoration projects. With the continuing wide-scale losses of woodland and freshwater habitats and their catchments across Wales, greater horizon planning is required (RSPB, 2013) and The United Nations Department of Economic and Social Affairs have launched a number of water conservation-related projects advocating cooperation and increased sampling of water quality to help protect and maintain current water resources (United Nations, 2013). This investigation has demonstrated that pre- and post scientific monitoring of habitats prior to anthropogenic disturbance can allow for detailed, accurate and sufficient mitigation to be implemented, while also contributing to the catalogue of species knowledge. Such methodologies and collaborations between conservation managers and researchers should be considered if future environmental development plans to ensure the national and international biodiversity targets are met.

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