CONSTRUCTED WETLANDS FOR MINEWATER TREATMENT

R&D Technical Report P2-181/TR

I Wiseman

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This report contains the results of a study into the effectiveness of passive constructed wetlands as a means of abandoned coal mine drainage treatment. The information in this document is for use by EA staff, the Coal Authority and consultants involved in the design and operation of constructed wetland schemes. It will be of use for any individuals or organisations dealing with the impacts of abandoned mine discharges.

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CONTENTS Page

LIST OF FIGURES

Page

LIST OF FIGURES (CONTINUED) **Page**

LIST OF FIGURES (CONTINUED) Page

LIST OF FIGURES (CONTINUED) **Page**

LIST OF TABLES **Page**

GLOSSARY OF ABBREVIATIONS

ALD	Anoxic Limestone Drain
AMD	Acid Mine Drainage
ASPT	Average Score Per Taxon
BMWP	Biological Monitoring Working Party
BSR	Bacterial Sulphate Reduction
BTO	British Trust for Ornithology
CEH	Center for Ecology and Hydrology
EA	Environment Agency
EAW	Environment Agency Wales
EIFAC	European Inland Fisheries Advisory Council
EPT	Ephemeroptera, Plecoptera, Trichoptera macroinvertebrate orders
EQS	Environmental Quality Standard
GPS	Global Positioning System
HABSCORE	Habitat Scoring System
HQS	Habitat Quality Score
HRT	Hydraulic Retention Time
LIFE	European LIFE fund for the Environment
NFC	National Fisheries Classification Score
NLS	National Laboratory Service
NPTCBC	Neath Port Talbot County Borough Council
NRA	National Rivers Authority
OAT	Ochre Accretion Terrace
QUASAR	Quality Simulation along Rivers
RAPS	Reducing and Alkalinity Producing System
RML	Richards, Moorehead and Laing Ltd.
TAPIR	Trend analysis software
TWINSPAN	Two-way Indicator Species Analysis
UKAS	United Kingdom Accreditation Service
VIS	Visual Impact Assessment Scheme
WBS	Waterways Bird Survey
WDA	Welsh Development Agency
WGCC	West Glamorgan County Council
WRC	Water Resources Council

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EXECUTIVE SUMMARY

Background

The Pelenna wetlands were constructed between 1995 and 1999 with an aim of treating the most contaminated minewater discharges affecting the Pelenna valley. The design of the schemes aimed to lower the dissolved iron concentration in the river to below the Environmental Quality Standard (EQS) of 1 mg/l. Prior to the system being constructed information had been gathered on the water quality and ecological status of the catchment.

The schemes utilise passive constructed wetland systems. These treat iron by creating suitable conditions for oxididation and reduction processes to occur. The first phase of the Pelenna wetlands was one of the first passive treatment schemes to be constructed for minewater treatment in Europe. Subsequent phases were constructed utilizing a number of novel design features.

Objectives

The main objective of this R&D project was to assess the contaminant removal performance, sustainability and environmental benefits of the treatment systems. This would provide information on how well the variety of passive wetland treatment systems work. The likely lifespan of the systems and their controlling factors would be identified. Finally the improvements in water quality of the rivers and macroinvertebrate, fish and riverine bird populations would be quantified.

Results/Discussion

Dissolved iron was the major contaminant of concern in the Pelenna minewaters. The wetland treatment systems were removing between 82 to 96 % of the incoming iron loading. Once all the phases were constructed the water quality at all sample points in the receiving watercourses quickly improved with dissolved iron dropping below the EQS limit. Following on from this, improvements in the abundance of macroinvertebrates were seen below the minewaters in the spring and summer after minewater treatment. Diversification of the macroinvertebrate species followed another year or two later. The delay in diversification was due to the lack of downstream drift from upstream populations that were impoverished due to episodic background acidification events. The trout populations recovered the year after minewater treatment, as the attempted spawning was more successful than in previous years. Improvements in the numbers of riverine birds were seen following treatment. Increased numbers of dippers and evidence of successful breeding were especially good indicators of the improved water quality.

During early 2000 and 2002 the Whitworth A treatment system blocked and the minewater discharged effectively untreated into the Nant Gwenffrwd. On both these occasions the water quality exceeded the EQS for at least three months. The macroinvertebrate populations were significantly affected, dropping back to pre-treatment levels in some sample points. The trout populations were similarly reduced with the main effect being seen in the numbers of trout fry. The blockage of the system was partly due to the design of the inlet structure and pipeworks, and this was exacerbated by a lack of a regular maintenance regime. Frequent

clearing of the inlets and pipework as part of a maintenance programme could have avoided the damaging overflows.

Studies of the internal processes occurring within the systems have identified that oxidation of the iron is the dominant removal process in all systems. Many of the treatment system components were not working exactly as they had originally been designed. They were however still removing iron and working effectively when receiving contaminated minewater. No trends were identified in the removal performance characteristics of the treatment systems and this indicates that the current excellent removal rates should continue.

The likely lifespans of the systems were identified where possible. The scheme that was identified as being most likely to suffer from a reduction in contaminant removal performance is Whitworth A. It is suggested that the ochre accretion on the Reducing and Alkalinity Producing System (RAPS) cell will overwhelm the cell between 2010 and 2016.

Conclusions/Recommendations

The R&D project has demonstrated that passive constructed wetland treatment systems can be an effective and low cost means of minewater treatment for both net alkaline and net acidic discharges. When maintained properly the schemes can reduce the dissolved iron concentrations below the EQS limit. When this has been achieved substantial recovery of the aquatic ecology has been demonstrated. The first signs of recovery are seen the first year after minewater treatment with diversification occurring one to two years later. Improvements are achieved right through the food chain making this method of minewater treatment an effective way of restoring biodiversity. However if minewater is allowed to discharge untreated again then the rivers will quickly return to their pre-treatment conditions.

The main recommendation from the project is that the inclusion of a routine maintenance programme is an essential part of any future treatment schemes. Constructed wetlands are a low maintenance alternative for minewater treatment; they are not however maintenance free schemes.

KEY WORDS

Constructed wetlands, minewater, Pelenna, South Wales, treatment performance, environmental benefits, ecological recovery, sustainability, macroinvertebrate diversity.

1 INTRODUCTION

1.1 Objectives

There was one overall objective of the constructed wetlands for minewater treatment R&D project. This was to investigate the performance, internal processes, environmental benefits and sustainability of constructed wetlands. The investigations were in order to assess their suitability for wider adoption as a sustainable treatment method for drainage from abandoned coal mines.

This overall objective is expanded in the points below:

1.1.1 Performance assessment of the minewater treatment systems

The aim was to provide a quantified assessment of how well the individual wetlands were removing contaminants and the controls on removal performance.

1.1.2 Assessment of internal processes and the sustainability of the wetlands

An analysis of the individual contaminant removal and associated wetland processes has been undertaken. An assessment of the dominant processes in separate areas of the systems and the variety of factors that control these processes have been made. Once these processes, factors and controls have been identified then the sustainability and longevity of the wetlands can be predicted. Where possible factors to enhance the sustainability and longevity of the systems have been identified.

1.1.3 Investigations into the environmental benefits of treating the minewaters

An objective of the project was to assess the benefits, if any, the reduction in contaminant load may be having on the receiving watercourse. This was monitored as changes in river water quality in the catchment. Improvements in water quality can then effect the aquatic ecosystem. An assessment of the populations of invertebrates, fish and riverine birds has been made and compared to background data collected prior to the construction of the wetlands.

1.2 River Pelenna Minewater Project

1.2.1 Background to the problem

Coal mining has had a lasting impact on the landscape and development of South Wales. Large areas were utilised for coal extraction and its associated industries and infrastructure. Nowadays there are few active areas of coal mining left. The effect this development and subsequent decline in the industry has had on the people, landscape and environment is very marked.

The Pelenna valley, situated near the towns of Neath and Port Talbot in South Wales, is a classic example of the legacy of coal mining. Most of the physical infrastructure of the industry has gone. The houses that were built for the miners are still there but the pit heads and coal yards have been demolished. A few spoil tips are still evident but most have now been reclaimed and revegetated.

In the area around Tonmawr, a village in the Pelenna valley, the earliest records of coal mining go back to a level that was opened in 1823 (Reynolds 1985), this was followed by the development of a number of pits over the next 100 years or so. By the early 1960s coal mining had ceased in the valley.

Following the closure of these mines the workings flooded and mine drainage discharged into the Gwenffrwd and Blaenpelenna, tributaries of the River Pelenna. The discharges stained the two tributaries and the River Pelenna orange. They caused elevated iron concentrations for approximately 7km, as far downstream as the confluence with the River Afan (Edwards *et al.* 1997). Assessments of the ecological status of the polluted rivers showed that juvenile trout (*Salmo trutta*) populations and macroinvertebrate assemblages were impoverished. This was found both upstream and downstream of the minewater discharges on the Gwenffrwd and Blaenpelenna, and to a lesser extent on the River Pelenna (Edwards, 1995). Macroinvertebrate assemblages in the headwaters of the catchment were typical of acidified streams, while communities downstream of the major minewater discharges were even more impoverished. Poor survival rates of trout eggs, alevins and parr were observed in the Blaenpelenna and Gwenffrwd, particularly downstream of the minewater discharges. The toxic effects of metal concentrations, acidity of the minewaters and episodic surface-water acidification, coupled with the smothering effects of ochre on the substrate, were thought to be responsible for the impoverishment of the aquatic fauna.

1.2.2 Origins of the treatment project

A project was set up to deal with the impacts these discharges were having on the local watercourses and the general environment. In 1992, The BOC Foundation for the Environment funded an investigation into the water quality of the catchment. This aimed to identify the levels of treatment necessary at each minewater discharge to achieve acceptable standards downstream. This work was managed by one of the predecessors of the Environment Agency (EA), the National Rivers Authority (NRA), which provided water quality data for the Institute of Hydrology to run its QUASAR (QUAlity Simulation Along Rivers) model. The study recommended that the iron concentrations in the Nant Gwenffrwd and Nant Blaenpelenna be reduced by 95% and 50% respectively. This should then achieve the European Inland Fisheries Advisory Council (EIFAC) standards and make the watercourse suitable for recolonisation by salmonid fish (Ishemo and Whitehead, 1992). Following on from this work, a contract was let by the NRA in spring 1993 to consultants Richards, Moorehead and Laing (RML) Ltd. They undertook a feasibility study of the most suitable and cost effective methods of achieving the required treatment levels. To meet the dissolved iron targets in the watercourses would involve treating the five major discharges in the valley. Based on the recommendations of RML (RML, 1993) a treatment project was proposed. The former West Glamorgan County Council (WGCC) and NRA undertook the River Pelenna Minewater Project. These organisations are now the Neath Port Talbot County Borough Council (NPTCBC), and the Environment Agency Wales (EAW). Funding was provided by the LIFE fund, a European financial instrument for the Environment, and the Welsh Development Agency's (WDA) Land Reclamation Programme. The chosen method of treatment was to utilise constructed wetlands as a means of passively treating the minewater discharges.

The scheme was constructed in three phases between 1995 and 1999. Phase I treats Whitworth No1 minewater, and had a number of demonstration and research features incorporated to aid in the design of the following two phases. These latter phases treat Garth

Tonmawr, Whitworth A, Whitworth B and the Gwenffrwd minewaters. The construction costs for each phase were £214,314, £254,882 and £360,198 respectively. With the entire project management and monitoring costs, the total cost of the scheme was £1.4 Million.

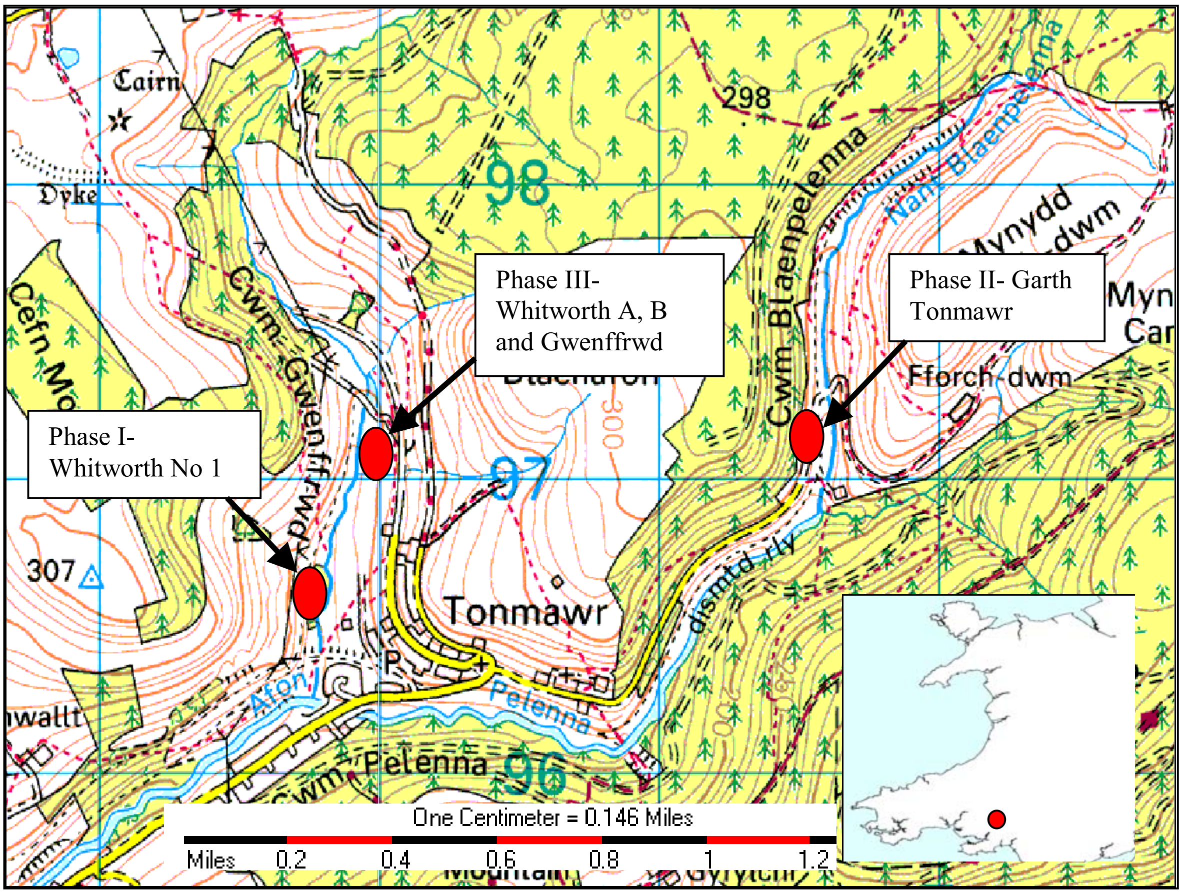


Figure 1.1 Location of the three phases of the Pelenna constructed wetlands.

1.2.3 Design of the treatment systems

The design of each system was based on the hydrochemistry of the minewaters. The iron loading of each discharge is of particular importance as a main control on the land area needed for treatment. The pH and alkalinity are also important in choosing the types of treatment elements needed. The details of these determinands for the five discharges are shown in Table 1.1. Data for Whitworth B and Gwenffrwd minewaters are presented for two dates, these are pre and post a natural underground diversion of these minewaters that is explained further later in this section.

Table 1.1 Iron loading, pH and alkalinity of the five Pelenna minewater discharges

	pH	Alkalinity (mg/l $CaCO_3$)	Flow (l/s)	Total Iron (mg/l)	Iron Loading (kg/day)
Whitworth A[1]	6.0	64.5	10.2	59.3	38.8
Whitworth B *pre diversion*[2]	6.1	33.3	1.0	4.5	0.5
Whitworth B *post diversion*[2]	5.5	19.3	18.0	7.3	3.3
Gwenffrwd *pre diversion*[3]	4.5	12.2	11.0	7.1	9.9
Gwenffrwd *post diversion*[3]	6.3	17.0	9.7	1.5	5.7
Garth Tonmawr[4]	5.6	36.1	22.1	29.2	52.9
Whitworth No 1[5]	6.3	72.5	5.9	23.3	9.3

Notes: Means of data from monthly spot sampling, the periods of sampling are as follows:
[1]Whitworth A- 04/93 to 19/03/02
[2]Whitworth B- (pre diversion) 04/93-10/98 (post diversion) 10/98-19/03/02
[3]Gwenffwrd- (pre diversion) 10/91-10/98 (post diversion) 10/98 to 19/03/02
[4]Garth Tonmawr- 10/91 to 19/03/02
[5]Whitworth No 1- 10/95 to 19/03/02

Phase I: Whitworth No 1

This system was constructed in 1995 to treat a discharge from the former East End Colliery. It comprises of four cells of equal area totalling 900 m^2 (Figure 1.2). These were constructed in pre-cast concrete with a geo-textile base liner. The cells operate in parallel by splitting the incoming water into four channels. A number of demonstration features were designed into the system in order to determine which treatment environments were most effective. This would then allow feedback to the design of Phases II and III. The system incorporates two substrate types (bark mulch and mushroom compost), two vegetation types (Cat's-tail-*Typha latifolia* and Soft rush-*Juncus effusus*) and two flow regime environments (surface-*aerobic*, subsurface-*anaerobic*). A photograph of the system is shown in Figure 1.3.

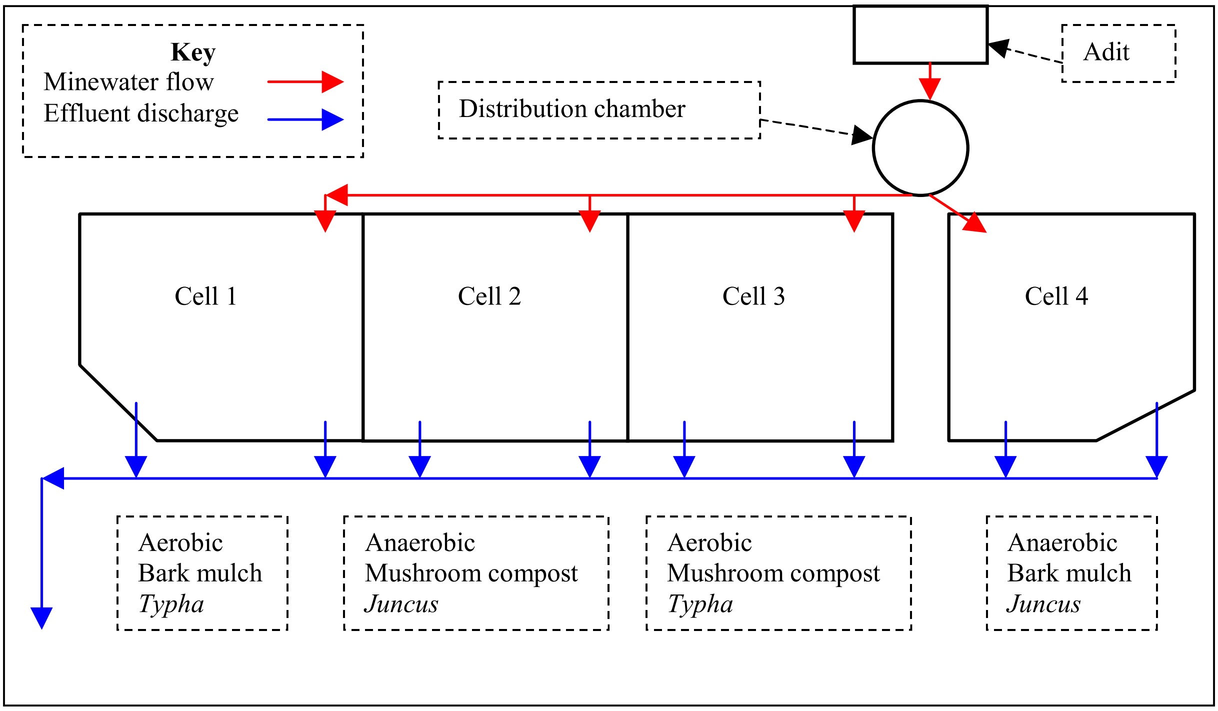


Figure 1.2 Layout of Phase I-Whitworth No 1 wetland treatment system.

Figure 1.3 View of cells 1 to 3 of the Whitworth No 1 wetland treatment system

Phase II: Garth Tonmawr

This system was constructed from 1998 to 1999 and treats the discharge from the former Garth Tonmawr colliery. This was the last colliery to close in the early 1960s (Paul Edwards, personal communication). The scheme consists of an existing natural wetland from where the water is piped under gravity across the Nant Blaenpelenna to the constructed wetlands. These are constructed with honeycomb mat stabilised local colliery waste for the bunds and red brick walls between the cells. The system incorporates five cells in series with a total treatment area of 6370m^2. These cells are shown in Figure 1.4 and in the photograph in Figure 1.5 and are arranged in the following order:

- Cell 1 Aerobic settlement lagoon (2480m^2).
- Cell 2 Reducing and Alkalinity Producing System (RAPS). This is an anaerobic downward flow wetland, formed from organic compost overlying a non-dolomitic limestone base (970m^2).
- Cell 3 Aerobic wetland (980m^2).
- Cell 4 Reducing and Alkalinity Producing System (RAPS). This is an anaerobic downward flow wetland, formed from organic compost overlying a non-dolomitic limestone base (980m^2).
- Cell 5 Aerobic wetland (960m^2).

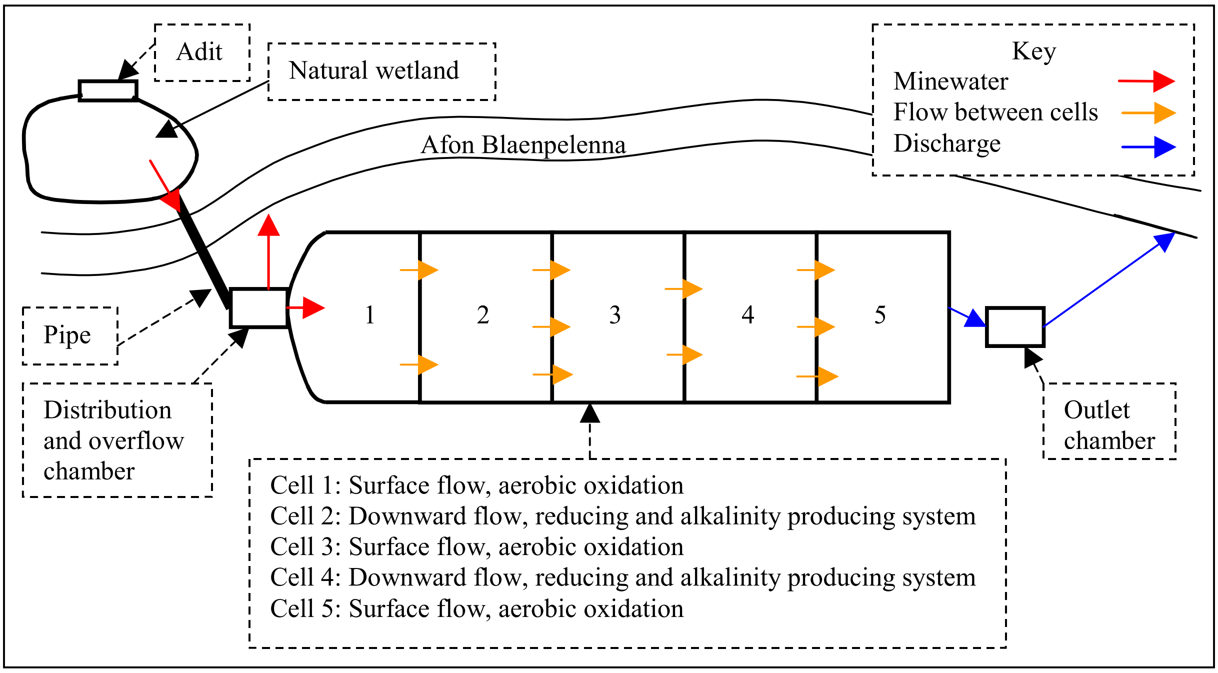


Figure 1.4 Layout of Phase II- Garth Tonmawr wetland treatment system.

Figure 1.5 Photograph looking down on the Garth Tonmawr system with the minewater entering the system on the right.

Phase III: Whitworth A, Whitworth B and Gwenffrwd minewaters

This system treats three separate minewaters, Whitworth A, Whitworth B and Gwenffrwd. These drain the former Llantwit Merthyr, Welsh Main and Wenallt collieries respectively. The initial construction of the wetlands was as described below.

Whitworth A minewater flows into a Reducing and Alkalinity Producing System (RAPS) with an area of 1825m^2. This consists of a 50-cm deep anaerobic organic substrate overlying a 50-cm limestone bed with a downward flow of water through the system. The discharge from the RAPS then enters an aerobic wetland of 4500m^2. This was created by building a bund around an existing area of natural wetland on the floodplain.

The Gwenffrwd minewater is split into two in a distribution chamber. Up to 225 l/min of the incoming flow passes through a RAPS cell of 2425m^2, while the remainder of the flow passes over an Ochre Accretion Terrace (OAT), this operates to aerate the minewater. The two flows are combined again as they enter a settlement pond of 850m^2. Finally the water enters an aerobic wetland of 2000m^2 before discharging to the Nant Gwenffrwd.

The Whitworth B minewater was left to enter the existing Whitworth Lagoon, a small aerobic wetland with an area of 1725m^2. This area should have been large enough to treat the loading from this discharge.

Around October 1998 the Gwenffrwd minewater discharge moved due to an underground diversion following heavy rain. It reappeared at the site of the Whitworth B outlet combined with this minewater. This left half of the phase III system being unutilised for treating contaminated minewater. The extra loading in the Whitworth B discharge was too great for the existing Whitworth lagoon and therefore significantly increased the contaminant load entering the Nant Gwenffrwd.

In response to this diversion modifications to the system were made from February to March 2001. Both the remaining Gwenffrwd flow and the combined Whitworth B flow were routed through the Gwenffrwd treatment system. The layout of the wetlands is shown in Figure 1.6. A photograph of the systems is included in Figure 1.7.

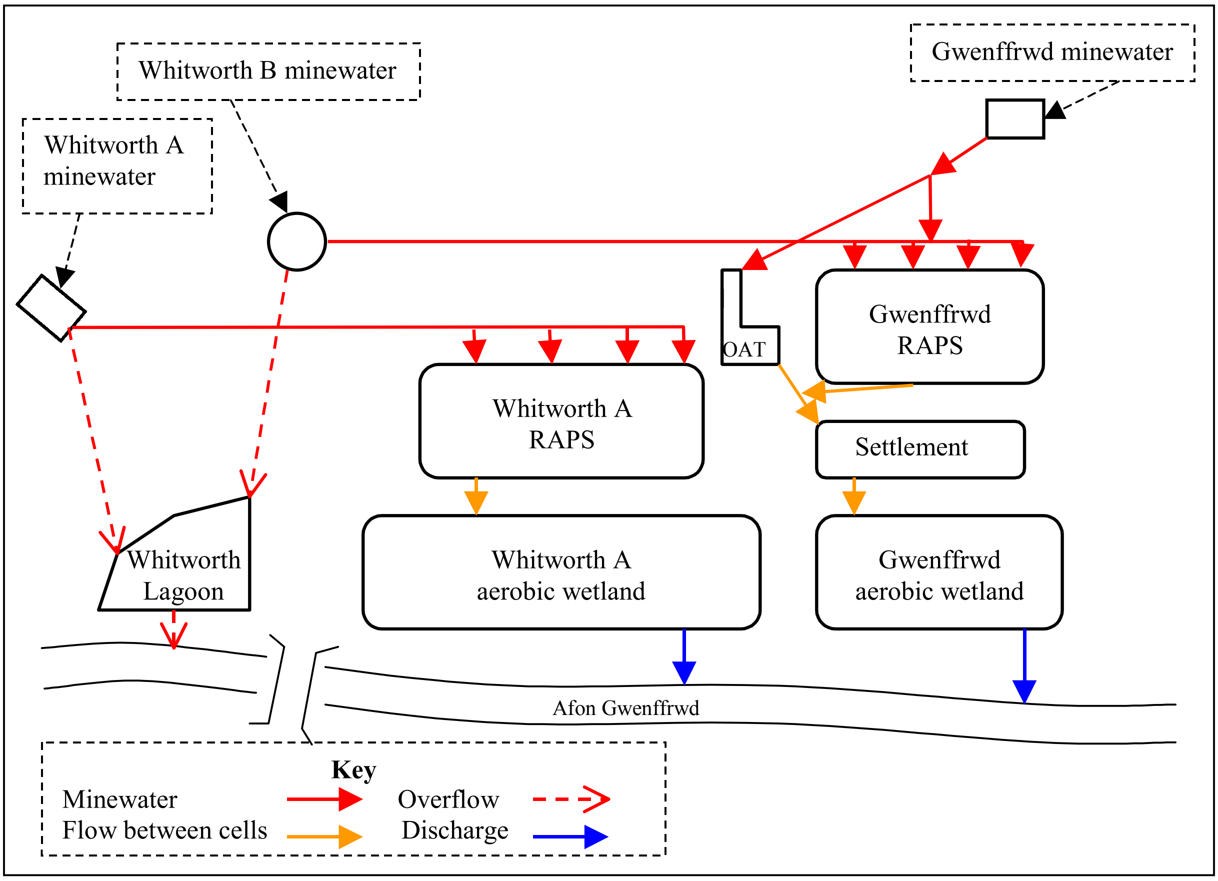


Figure 1.6 Layout of Phase III-Whitworth A, Whitworth B and Gwenffrwd treatment systems.

Notes: OAT = Ochre Accretion Terrace, RAPS = Reducing and Alkalinity Producing System

Figure 1.7 View of the Whitworth A leg of the Phase III wetlands.

1.3 Minewater Chemistry

1.3.1 Pyrite chemistry and mine drainage

The principal contaminants in coal mine discharges are generally iron and acidity. These are caused by the oxidation of pyrite within the mine. Pyrite (FeS_2) forms in marine sediments at the same time as the coal measures are laid down. It is frequently associated with the coal seams and surrounding strata of marine mudstones that accumulate in reducing environments.

Following removal of the coal the pyrite is exposed to atmospheric oxygen, water and bacterial catalysts which leads to the following oxidation reaction (Banks *et al.* 1997):

$2FeS_2 + 2H_2O + 7O_2 = 2Fe^{2+} + 4SO_4^{2-} + 4H^+_{(aq)}$
Pyrite + water + oxygen = ferrous iron + sulphate + protons

The ferrous iron, sulphate ions and free acidity from the protons are in forms which can be easily dissolved and transported in water. While the mine is operating this is not generally a problem as pumps keep the majority of the mine dry so most of the contaminants do not get mobilised. The problems tend to occur when the mine closes and the pumps are shut off. Water starts to fill up all the voids the mine created and the local water table rises back up in a process known as rebound. This creates ideal conditions for the oxidation products of iron to be dissolved and carried out of the mine. The minewater contaminants will continue to discharge for many years after the groundwater has fully rebounded.

Once a minewater encounters an oxidising environment, either within the mine or when it enters a receiving watercourse oxidation of ferrous iron (Fe^{2+}) to ferric iron (Fe^{3+}) can take place as below:

$4Fe^{2+} + 4H^{+}{}_{(aq)} + O_2 = 4Fe^{3+} + 2H_20$
Ferrous iron + protons + oxygen = ferric iron + water

This can then either undergo hydrolysis to the insoluble iron hydroxide or it can aid further pyrite oxidation within the mine by acting as an electron acceptor as in the reactions below:

$Fe^{3+} + 3H_2O = Fe(OH)_3 + 3H^{+}{}_{(aq)}$
Ferric iron + water = iron hydroxide + protons

$FeS_2 + 14Fe^{3+} + 8H_2O = 15Fe^{2+} + 2SO_4{}^{2-} + 16H^{+}{}_{(aq)}$
Pyrite + ferric iron + water = ferrous iron + sulphate + protons

The overall sequence of hydrolysis reactions is detailed below:

$4FeS_2 + 14H_2O + 15O_2 = 4Fe(OH)_3 + 8SO_4{}^{2-} + 16H^{+}{}_{(aq)}$
Pyrite + water + oxygen = iron hydroxide + sulphate + protons

The iron hydroxides (hydrous ferric oxides) formed in this process are insoluble and precipitate within discharge routes and the receiving watercourses. This causes the characteristic orange or ochreous staining that is visible in rivers below many minewater discharges.

As can be seen both the oxidation in the mine and the hydrolysis reaction give rise to a number of protons, or acidity, which can lead to significant impacts from the potential acidity and/or low pH of some minewater discharges. The generation of further contaminants including other metals in minewaters can occur due to the leaching effect of this generated acidity on the surrounding rock and release of trace metals in the pyrite.

Whether acidity is an issue or not is generally governed by the amount of available alkalinity in the minewater. The main sources for this alkalinity are from the weathering of carbonate minerals or reduction processes such as sulphate reduction.

1.3.2 Net acid or net alkaline minewaters

A common misconception with minewaters is that they are all acidic, hence the term Acid Mine Drainage (AMD). Many minewaters however, do not have acidic discharges due to the amount of alkalinity available within the groundwater. The minewaters can be classified into two types:
Net Alkaline- These discharges have an excess of total alkalinity over total acidity.
Net Acidic- Where a discharge has an excess of total acidity over total alkalinity.

Alkalinity relates to the capability of a solution to resist the lowering of pH and to neutralise strong acid (usually 1.6N H_2SO_4) to a specified end point (usually Bromocresol green-methyl red colour transition point at pH 4.5) (Howes and Sabine 1998).
Acidity is the capability of a solution to neutralise a strong base (e.g. NaOH) to a specified end point (usually pH 8.3). However it is not just the pH of a solution. Acidity is the

combination of the Hydrogen ion (proton) activity (indicated by pH) and the mineral acidity available from the capacity of metals such as iron, manganese and aluminium to undergo oxidation and hydrolysis reactions. Reactions such as those indicated in section 1.2.1 for iron release significant numbers of protons.

In the Pelenna valley the Whitworth A, B, Gwenffrwd and Garth Tonmawr minewaters are net-acidic. As such they have necessitated the use of RAPS cells in their designs to promote efficient iron removal, and counteract any problems likely to be caused by acidic discharges into the rivers. The Whitworth No1 discharge is net-alkaline and therefore is treated by mainly aerobic processes.

1.4 Impacts of Minewater Discharges on the Environment

As previously mentioned the main contaminants in these discharges are metals and acidity. The predominant metal contaminant, especially in the Pelenna valley minewaters, is iron. Other metals such as manganese and aluminium are frequently found in elevated concentrations in many other discharges. The combination of these metal contaminants and the pH of the discharges can have the following impacts on the receiving watercourses:

1.4.1 Visual impact

The formation and precipitation of iron oxides and hydroxides around discharge points from the mines and in the receiving watercourses is often the most obvious indication of an areas past mining history. Depending on the relative flows of the discharge and receiving watercourse and the iron loading from the mine, the orange ochre staining can be visible for a considerable distance downstream. The vivid orange staining caused from these processes will have a negative effect on people's perception of an area and their amenity use of it.

1.4.2 Water quality

The addition of contaminant bearing mine waters to otherwise un-impacted watercourses can have a significant impact on the quality of the receiving water. The effect will vary with the relative concentration and flow and therefore contaminant loadings of the discharge and the flow and chemical characteristics of the receiving water.

Minewater discharges have the ability to modify the pH and raise the levels of metals such as iron to a level where they exceed Environmental Quality Standards (EQS). These are guidelines based primarily on the Dangerous Substances Directive (76/464/EEC) and are aimed at protecting aquatic life. The current standards are for pH to be between 6-9, and a dissolved iron level of less than 1.0mg/l in 95% of samples.

Many watercourses are used as sources of water for human consumption, animal consumption and for extractions for industrial uses. A reduction in water quality can seriously affect these downstream users.

An example of this is the discharge at Ynysarwed in the Neath Valley, a site approximately 5km away from the Pelenna. Here the outbreak of minewater in the early 1990s entered the Neath Canal causing a major environmental incident and impact. Modifications to the canal downstream had to be made to protect a major industrial user of the water. The canal had to be dammed and the contaminated water discharged to the Neath estuary. Fresh water was then pumped back into the canal so that it was suitable for aquatic life and the downstream industrial uses (Edwards, 1994). In 2002 this pumping was still ongoing due to the scale of the project necessary in order to clean up the canal.

1.4.3 Impacts on biological quality

Minewaters have been shown to have a number of affects on the ecology of watercourses receiving contaminated discharges. Impacts can be seen in populations of invertebrates, fish, mammals and birds. Details of the main impacts are shown below:

Effects on invertebrate populations

Minewaters have been shown to cause a reduction in the diversity and abundance of invertebrate species downstream of the discharge point (Amisah and Cowx 2000, Critchlow-Watton 1998, Garate 2002, Koryak *et al.* 1972, Malmqvist and Hoffsten 1999, Moon and Lucostic 1979, Mori *et. al.* 1999, Sharp 2000, Soucek *et. al.* 2000, Von Reibnitz 2001, Warner 1970). The actual factors affecting the species vary, two of the main factors are the direct toxicity of the metals and pH, and substrate smothering by iron hydroxide precipitation.

Individual species and macroinvertebrate assemblages have been characterised as being good indicators of mine pollution due to their tolerance (or not) to the metal and pH effects. Species such as chironomids have been demonstrated by Amisah and Cowx (2000) and Gower *et al.* (1994) to exist at abandoned coal minewater impacted sites in the UK. Watanabe *et al.* (2000) identified their existence at metal mine contaminated sites in Japan. Malmqvist and Hoffsten (1999) identified certain stoneflies including *Amphinemura sulcicollis* and *Leuctra inermis* to be tolerant of metal minewater pollution in streams in central Sweden.

Certain species have been identified as being specifically intolerant of minewater pollution and as such are useful indicators of its effects. Mayflies especially appear to be intolerant of metal pollution, Nelson and Roline (1996) identified the mayfly *Rhithrogena hageni* as being intolerant and *Ameletus inopinatus, Ephemerella aurivilli* and *Heptagenia dalecarlica* have been found to be adversely effected by metal pollution by Malmqvist and Hoffsten (1999). Mori *et al.* (1999) found in their studies below a metal mine site in Corsica that the mayfly larvae *Ecdyonurus sp., Heptagenia sp., Baetis cyrneus, Ephemerella ignita* and *Habrophlebia fusca* were not found downstream of the mine site.

Effects on fisheries populations

Minewater discharges can reduce the diversity and numbers of fish in the receiving watercourses (Cannon and Kimmel, 1992). Amisah and Cowx (2000) demonstrated reduced fish numbers and limited reproduction of brown trout on the River Don in South Yorkshire. Sheehan and Knight (1985) found a complete absence of fish downstream of mine sites in California and abundant populations upstream.

The factors affecting the fish populations vary. Experiments by Edwards (1995) on the Blaenpelenna and Gwenffrwd rivers into the survival of trout eggs and caged trout parr found that the minewaters severely affected introduced fish. This work suggested that the smothering affects of iron, combined with the low pH in the watercourse were adversely affecting egg survival and causing a high mortality rate in the introduced fish. However background impacts caused by the acidified catchment were also controlling factors. The smothering of substrates by iron hydroxides and reduction in total biomass of benthic organisms was quoted by Koryak *et al.* (1972) as being a limiting factor on fish populations. Similarly, Scullion and Edwards (1980) suggested that the impoverished food supply, i.e. reduction in invertebrates, was responsible for the reduction in fish populations in the Taff Bargoed River.

Indirect effects

The impoverished food supply will have an impact on other species as well as fish. Included among these are riverine birds and mammals, such as dippers, kingfishers and otters. These would usually feed on the invertebrates and fish in the rivers. As a consequence of the reduced diversity and abundance of these they too can have reduced populations. Roberts (1996) surveyed the riverine birds on the Afon Pelenna and found that the area did not sustain a population of dippers even though it was estimated that the habitat was suitable for at least five pairs. The lack of prey species caused by the minewater pollution and acidification was attributed as being the major causes of this.

1.4.4 Amenity and recreation impacts

As already noted minewater discharges can have visual, chemical and biological impacts on watercourses. These can have serious effects on the amenity and recreational use of an area. Areas with rivers that are thought of as polluted will be less attractive for the general visitors and also less likely to be used for water based recreation activities. By reducing or excluding fish stocks then this will impair the development of the area as a fisheries resource. The associated impact of the lack of visitors will be felt both economically and socially by local hotels, guest houses, pubs, restaurants, shops and the general communities.

1.5 Treatment Options

There are two main options for treating minewater discharges; active or passive treatment. They vary in terms of effectiveness, cost, maintenance requirements and land requirements. Choice of treatment process will generally be made using a balance of these factors.

1.5.1 Active treatment

Active treatment will generally use a system of pumping and chemical dosing. The general approach is to use one or a mixture of the following (Younger, 2000):

- Aeration or an oxidising reagent to turn dissolved ferrous iron to insoluble ferric iron.
- 'Lime' dosing to raise pH and increase the reaction rate of iron hydroxide precipitation.
- Coagulation or flocculation using a chemical additive or mechanical means (i.e. centrifuge) to create larger particles that are more likely to precipitate.

1.5.2 Passive treatment

This would generally imply that treatment is carried out by natural geochemical processes that require no input of chemicals and a gravity feed distribution system. The main types of treatment process are outlined below (Younger, 2000):

- Settling Lagoons- use of simple ponds where precipitation is likely to be rapid. This allows the removal of a significant proportion of the dissolved iron load. The precipitate will be a relatively uncontaminated ochre and will therefore be more easily reused.

- Ochre Accretion Terrace (OAT)- Simple aeration spillway designed to oxidise iron.

- Aerobic Wetlands: These are most applicable to near-neutral net-alkaline discharges. The design is usually a shallow overland flow system; most are planted with reeds that serve to slow flow down. These wetlands work by allowing the oxidation, hydrolysis and precipitation of iron in a 'natural' environment.

- Anaerobic wetlands: The design of these is most suitable for net-acidic waters but can also treat sulphate rich net-alkaline discharges. They are created with thick compost substrates, which the contaminated water flows through or over. The systems operate by using bacterial sulphate reduction (BSR) to consume protons and generate bicarbonate alkalinity, this then raises and buffers the pH of the minewater. The BSR reaction is demonstrated by: $SO_4^{2-} + 2CH_2O \rightarrow H_2S + 2HCO_3^-$ (Rees *et al.* 2001). This bacterial reduction converts sulphate to sulphide, which will then either form hydrogen sulphide and be given off as a gas or react with ferrous iron to form insoluble sulphides, which are trapped within the substrate.

- Anoxic Limestone Drains (ALD): These are buried trenches filled with limestone. The anoxic minewater dissolves the limestone over time adding alkalinity to the discharge.

- Reducing and Alkalinity Producing Systems (RAPS): These are also known as compost wetlands. They incorporate a downward flow organic wetland over a buried bed of limestone. These operate by stripping oxygen from the minewater turning it anoxic prior to contact with the limestone to allow the addition of alkalinity. The processes occurring make it similar to a combination of an anaerobic wetland and an ALD.

2 METHODOLOGY

2.1 Wetland Treatment Performance

2.1.1 Water quality monitoring

Water quality samples were taken from the untreated minewaters and the outlet from each cell of each treatment system. River water quality was monitored at a number of sites including upstream and downstream of each discharge. Figure 2.1 shows the location of each sample point. Monitoring has been undertaken by the EAW at all sites from 1993 to 2001 and at the wetland sites since construction. Samples were taken using standard EA methods (Environment Agency, 1998a). The determinands routinely monitored are listed in R&D Project Record P2-181/PR Appendix 1. Field measurements for pH, conductivity, redox potential and dissolved oxygen was taken on a YSI 600XL multi-parameter meter and alkalinity was measured using a Palintest photometer 5000. The Environment Agency National Laboratory Service (NLS) in Llanelli undertook the analyses of the samples; the laboratory is accredited by the United Kingdom Accreditation Service (UKAS No. 1137). A brief overview of the methodologies used is outlined in R&D Project Record P2-181/PR Appendix 2.

Data gathered on the river water quality was analysed for any significant step changes following minewater treatment. A software package was used (TAPIR) to assess temporal trends. The Water Research Council (WRc) developed this, specifically for improved environmental monitoring by the Environment Agency (Wyatt *et. al.* 1998). Other water quality data analysis was undertaken using Minitab.v.13^{TM} statistical software.

On occasions continuous pH, temperature and conductivity data were collected from the watercourses. This was undertaken to assess the level of surface water acidification in the catchment and the buffering capacity of the wetland discharges. YSI 6920 continuous monitors were used for this purpose.

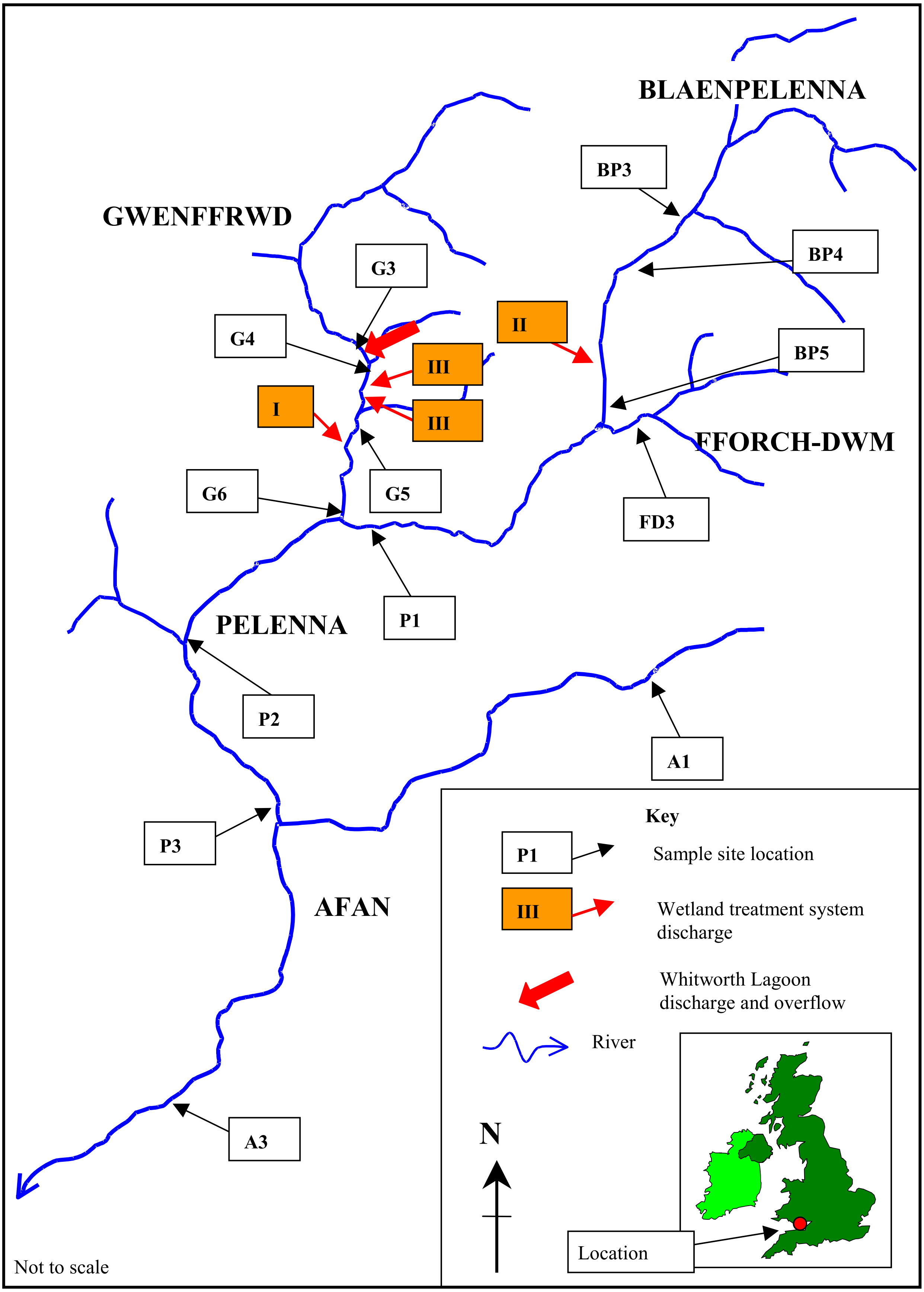


Figure 2.1 Wetland treatment systems and sample point locations

2.1.2 Hydrometric data collection

EAW Hydrometric Field Officers undertook flow monitoring in the wetlands and rivers using standard EA techniques (Environment Agency, 1998b). Monitoring was undertaken on a monthly basis along with the routine water quality monitoring. Continuous flow monitors were installed and operated by the hydrometric team using standard EA techniques (Environment Agency, 1998b). These were located on each of the minewater discharges and at the outlets from two of the treatment system.

The Hydrometric team provided rainfall data for the project from a gauge in the Fforch-Dwm catchment using standard EA methodology (Environment Agency, 1998b).

2.2 Internal Processes and their Sustainability

2.2.1 Sediment analysis

Sediment samples were taken from Whitworth No 1 and Whitworth A wetlands. Two main projects were undertaken as part of this R&D work. They were undertaken as MSc projects in the summers of 2000 and 2001. They were designed to advance previous work on the wetlands especially that of Wiseman (1997) and Rees (1998).

The first project (Thomas, 2001) sampled sediments using a 35mm diameter handheld metal piston corer. The cores were split into sections on site based on depth. The redox potential of each sample was identified on site. The samples were then sequentially extracted in Aberystwyth University for a range of metal fractions. These were water exchangeable, organically bound, non-crystalline oxide, crystalline oxide and sulphide bound iron, manganese and aluminium. The organic matter content and moisture content were also determined. From these results a picture of the spatial distribution of metals was developed. It also identified the range of iron precipitates occurring within the systems. This then allowed some inferences of the processes occurring within the systems to be made. Differential Thermal Analysis and Thermogravimetric Analysis were utilised on one sample to identify the type of iron precipitates forming.

The following project (Reilly, 2002) concentrated primarily on the Whitworth A RAPS cell. It involved taking a large number of cores from the RAPS cell. Various coring methods were attempted, including piston coring as used previously. However they were all found to disturb the sample too much. The chosen method was to use clear plastic tubing of 60mm internal diameter. Rough teeth were cut into the base to allow penetration through the substrate and plant roots. These were twisted into the substrate by hand and then extracted. By maintaining an airtight seal over the top of the tube with your hand it was possible to extract a substantially undisturbed core (Figure 2.2). The clear tubing allowed visual inspection of the core to check that it was satisfactory. The cores were then frozen on site with dry ice (solid CO_2) to avoid disturbance during transport. Prior to analysis the cores were sectioned while frozen with a saw. The sections were chosen with depth above and below the interface between oxidised iron and organic sediment. Analysis of the moisture content, organic matter content and total iron, manganese and aluminium was undertaken at the NLS in Llanelli.

Figure 2.2 Ochre and organic substrate boundary in a core from the Whitworth A RAPS.

2.2.2 Pore water analysis

Pore water samples were taken from a number of depths within the wetlands. This was undertaken in the summer of 2000 and reported in the MSc project of Thomas (2001). Samples were taken from the Whitworth No 1 and Whitworth A systems. Two types of sampler were used. The first, supplied by Aberystwyth University, utilised a plastic tube with porous ceramic cups on the base and sealed with rubber bungs on the top. Glass tubes were placed through the bungs and then extended with rubber hosing which was clamped to be airtight. A hand pump was used to create a vacuum within the sampler and pull water through the ceramic cup at the required depth. The samplers were allowed to equilibrate with the sediments and were then sampled by pumping the liquid from the interior of the sampler while it was *in situ*.

The second type consisted of plastic tubes approximately 40mm in diameter. The tubes were all the same length but had holes drilled at different depths. This allowed water entry from discrete zones within the sediments. The water was then pumped out of the samplers whilst *in situ*. For both methods samples were taken every two weeks and the water collected was analysed by the NLS in Llanelli for the determinands listed in R&D Project Record P2-181/PR Appendix 1.

2.2.3 Permeability tests

The permeability of cores from the Whitworth A RAPS was investigated during the summer of 2001. A permeameter was set up in the laboratory using five cores extracted from the wetlands by Reilly (2002) for his MSc project. The sample tube and frozen core was fitted to a U tube and head collection tube. The sample was allowed to defrost fully and was supported by gravel and metal mesh in the U tube. The head datum level was measured by filling the head collection tube and allowing it to drain to an equilibrium level. Then the sample tube was filled with water and the initial head above datum was noted. The apparatus was left for between 2-4 hours and the final head was recorded. The data was entered into a derivative of Darcy's equation (Equation 2.1, Fetter (1994)) to attain a value for hydraulic conductivity.

$$K = \frac{d_t^2 L}{d_c^2 t} \ln \left(\frac{h_0}{h}\right) \qquad \textbf{Equation 2.1}$$

Where

K is hydraulic conductivity (cm/s)
L is sample length (cm)
t is time (s)
d_t is the diameter of head collection tube (cm)
d_c is the diameter of the sample tube (cm)
h_0 is the initial head
h is the final head

The sample tube was then removed and the experiment repeated for the remaining four samples. A photograph of the apparatus in operation can be seen in Figure 2.3.

Figure 2.3 Permeameter in operation (Reilly, 2002)

2.2.4 Assessment of retention times

The Hydraulic Retention Times (HRT's) of the wetlands were investigated. Visible dyes (Fluorescein and Rhodamine) were utilised to assess preferential flow pathways and minimum retention times. Salt was used as a tracer by adding it to the inlet and then measuring the conductivity and salinity at the outlets on remote logging sondes. Peaks in the data could then be used to identify the average retention time of each system or cell.

2.2.5 Limestone and alkalinity budget calculations

Alkalinity was measured at the inlet and outlet of the RAPS cells. It is not however possible to calculate the use of limestone directly from the alkalinity generated through the system. This is because it makes no distinction between bacterially generated alkalinity and limestone generated alkalinity. It also does not allow for sinks of alkalinity within the system, for example by reaction with the free acidity created by iron oxidation on the surface of the wetland.

Calcium budget of limestone use

In order to calculate the limestone use from the monthly sampling data a calcium budget has been calculated. This uses a number of assumptions:

- The only addition of calcium is from limestone dissolution.
- All limestone is in the calcium form (i.e. $CaCO_3$). This is indicated from the design plans for the scheme (NPTCBC/EAW, 1999). Results from the monitoring of magnesium in the system indicate a slight removal of magnesium between the inlet and outlet of the RAPS (Mean 33.9 at inlet and 31.9 exiting the RAPS). As there is very little export of magnesium this would indicate that there is no $MgCO_3$ in the limestone.
- There is no removal of calcium within the cell. Again the results for magnesium demonstrate a very small removal and given that calcium would be expected to behave in a similar chemical manner it suggests that the amount of removal within the system is negligible. The calcium budget of Whitworth No 1 cells 1 and 4 and Garth Tonmawr cell 1 was calculated for comparison. These cells contain no limestone additions and therefore would give an indication if any net accumulation or loss of calcium were occurring. It was found that there were variations in the results from each cell, with some exporting Ca and some not. On the whole the levels of change throughout these systems were insignificant compared to the levels found in the cells containing limestone additions.

Initially the inlet and outlet loading of calcium was calculated, from this the addition of calcium by the RAPS was calculated. An average calcium addition was calculated for the system since construction. This can then be converted into a $CaCO_3$ equivalent. From design data (Younger, 1996) it is known that $1m^3$ of limestone with 30% void space would weigh 1102 kg. The surface area and depth and therefore volume of the limestone bed in m^3 are known. The initial mass of the limestone bed can then be calculated. Using this initial mass of limestone and the average removal rate the time taken for all the limestone to dissolve can be calculated. This obviously assumes (unrealistically) that there is no change in the reaction rate of limestone over time.

This method calculates the time taken for all the limestone to dissolve and obviously is not the most useful figure to calculate. At some point prior to this there will no longer be enough limestone left to generate the alkalinity necessary. It is this date that will be more important.

Alkalinity budget of limestone use

It was noted above that it was not possible to calculate an accurate alkalinity budget through the whole RAPS unit. However an attempt was made to calculate an alkalinity budget purely for the limestone bed of the RAPS. As part of this R&D project and the MSc of Thomas (2001) pore water samplers were installed in the organic layer of the RAPS unit in the Whitworth A system. The deeper samplers collected water just before it entered the limestone bed. This water had no alkalinity and a known pH and metal content. Therefore it could be concluded that all the alkalinity exiting the RAPS was created in the limestone bed as any entering in the minewater had been utilised by processes occurring within the organic substrate.

The water entering the limestone bed still had an acidity value and this was likely to react with and use up some of the alkalinity produced from the limestone. The acidity of the water entering the limestone was calculated (Watzlaf *et al.* 2002). The calculation used the assumption that all of the iron was in the Fe^{2+} (reduced) form at the low pH's observed. It was now possible to create an inlet versus outlet acidity budget for the limestone bed. It was found that the budget was negative, indicating that some alkalinity must be being used within the limestone to neutralise the incoming acidity. This alkalinity use was calculated by converting the acidity used within the system into alkalinity. When the figure for alkalinity use and the amount exiting the system were combined a value for alkalinity production is created.

Once the alkalinity production is known then a rate can be converted and applied to the initial mass of limestone in a similar matter to the calcium budget calculations. Again it makes the assumption that no rate changes occur over time. Like the method above it will calculate the time taken for all the limestone within the system to be utilised.

2.3 Environmental Benefits

2.3.1 Visual impact assessment

Using a key developed by Edwards and Maidens (1995) the impact of the ochreous discharges on each receiving watercourse was assessed. This gives a simple indication of the level of colouration and impact. The criteria assessed are explained below (Table 2.1):

Table 2.1 Visual impact assessment classification key

Classification system for the assessment of the visual impact of ferruginous discharges on river gravels.

Score 1 for Yes and 0 for No

1. Are there any visible ferruginous (orange) deposits?
2. Do these deposits cover >50% of the gravel substrate?
3. Is the gravel blanketed with orange ferruginous matter/bacterial slime, the majority of which can be easily washed or rubbed off?
4. Does disturbing (kicking) the gravel result in an opaque orange, rather than brown, cloud in the water.

Score	Category	Definition
0	No impact	No staining
1	Low impact	Limited staining
2	Moderate impact	Extensive staining
3+	High impact	Extensive blanketing

2.3.2 Invertebrate populations

Invertebrate populations were monitored twice yearly from 1993 until 2001 at the points shown in Figure 2.1. Sampling was undertaken in spring and autumn using a three-minute kick sample, followed by identification to species level in the laboratory. Standard EA methodology was followed (Environment Agency, 1999). In 2002 the population was only assessed in full in the spring with a bankside assessment being made in the summer due to time constraints within the project. The accuracy of sorting and identification was subject to in-house and external quality control. The external verification was undertaken at the Centre for Ecology and Hydrology (CEH) in Dorset. MSc students working with the EAW each summer undertook the in-house sorting and identification of the macroinvertebrates. The results presented here are based on the results of Boyd (1998), Critchlow-Watton (1999), Crocker (1997) Davies (1993), Garate (2002), Higham (1994), Rose (1996), Sharp (2000) and Von Reibnitz (2001).

Data generated were analysed using a number of biotic indices. These included:

- Total abundance of all macroinvertebrate individuals. This was useful to assess changes in the size of a population over time.
- Biological Monitoring Working Party (BMWP) Score, the BMWP score was designed to give a general indication of the biological conditions of rivers. Its primary use is as an indicator of organic pollution effects. Each macroinvertebrate family is given a score between one and ten, based upon its susceptibility to organic pollution (Metcalfe, 1989). The score of a site is then worked out by combining the scores from all the family groups identified.
- Average Score Per Taxon (ASPT), this was calculated by dividing the BMWP score by the number of taxa. It provides an indication of pollution within a watercourse.
- Combined Ephemeroptera, Plecoptera and Trichoptera (EPT) abundance. These three orders have been identified by previous researchers (see section 1.3.3) as being intolerant of minewater and metal pollution. Changes in the EPT abundance between sites and survey years will reflect the level of minewater contamination. Expressing the EPT abundance as a percentage of the total abundance has also been used to identified this.
- Abundance of the three orders Ephemeroptera, Plecoptera and Trichoptera has also been assessed individually. Again these have been assessed as a percentage of the total abundance.
- Numbers of EPT families and genera and Ephemeroptera families and genera. Identification of the changes in the numbers of families and genera over time will give an indication of how the diversity of the sites is changing.
- The multivariate classification technique TWINSPAN (Two-Way INdicator SPecies ANalysis) was also employed. TWINSPAN splits sites into groups that are essentially similar in taxonomic composition (Gauch, 1982). Indicator species that show a preference for each split or group are identified and the relationships between the site groupings and environmental variables, such as water quality, can be explored. The result of this analysis is a grouping together of sites that are either poorer or richer in terms of diversity of taxa. Severely minewater impacted sites will be grouped together as will good quality control sites. In between will be a range of groups, the number of which depends on the taxon compositions between sites. When TWINSPAN is applied to a dataset that covers a number of years it shows how sites change group membership throughout the years. This will give an indication of whether sites are increasing species diversity and approaching the taxa found in the control rivers.

2.3.3 Fisheries populations

Fish population data were gathered in the summers of 1993, 1994, 1996 and 1999 to 2002 at the sample points shown in Figure 2.1. Stretches of 50 m were electrofished using a quantitative method. The stretches were isolated up and downstream using nets and then fished a minimum of three times. Standard EA methodology was observed (Environment Agency, 1998c). Results were converted into fish densities per 100m^2 using the method of Carle and Strub (1978) for both fry (<1 year old) and parr (>1 year old). National Fisheries Classification Scheme scores (NFC) were calculated for the data (Mainstone, et. al. 1994). This system categorises the fish densities observed against a national dataset of fisheries of a similar type. The observed densities are then expressed as being within a certain percentage of sites for a given species group within the database. The system is classified in six parts as below:

- Grade A Excellent within top 20% of database
- Grade B Good 60-80%
- Grade C Fair 40-60%
- Grade D Fair 20-40%
- Grade E Poor Lower 20%
- Grade F Fishless

Predicted densities of fish were calculated using a habitat assessment scheme (HABSCORE), which measures and evaluates habitat features that influence salmonid distribution (WRc, 1999). Observations of the field sites are put into the system, which returns a predicted density for the habitat available assuming pristine water quality.

Estimates were made of the economic benefit of the observed increases in trout parr densities in the Pelenna catchment, based on the estimated subsequent increase in rod catches of adult sea trout in the River Afan. The market value of salmon and sea trout fisheries is largely a function of the annual declared rod catch. It has been estimated that the mean value of a rod caught salmon in Wales was approximately £8,000 at 1996 values (Radford et al, 1991) and that one salmon is equivalent in value to 2.5 sea trout (Evans, 1996). The increase in rod catches was calculated from observed trout parr densities (from this study), estimated survival and return rates (Kennedy, 1988; Scranney, 1998), the estimated proportion of fish returning within the angling season (Scranney, 1998) and the estimated exploitation rate obtained from studies on the River Tawe (Solomon, 1995). An allowance was also made for fish returning to freshwater to spawn on more than one occasion (Scranney, 1998).

2.3.4 Riverine bird surveys

Riverine bird populations were assessed using the British Trust for Ornithology (BTO) Waterways Bird Survey (WBS) methodology (Taylor and Murray, 1982). Three river stretches were surveyed on six separate occasions with no three sequential visits spanning less than ten days. Species were identified and as much extra information as possible was noted; including sex, juvenile birds and nest sites. The location of each siting was made using a handheld Global Positioning System (GPS). Analysis of territories was undertaken using the BTO WBS guidelines (Marchant, 1994). This identifies areas of the river that are occupied by separate specific groups of birds over a number of visits.

The 2001 survey commenced later in the year than the 1996 and 2002 surveys due to restricted access as a result of the Foot and Mouth epidemic. No access was gained to the polluted control stream (Afon Corrwg Fechan) during that survey year.

2.3.5 Vegetation surveys

Surveys of the vegetation species within the wetlands were made in the summers of 2000 and 2001. Plants were identified on site where possible with samples being identified in the laboratory where necessary.

3 RESULTS AND DISCUSSION

3.1 Wetland Treatment Performance

Monitoring of the wetlands has been undertaken since construction of the various schemes. The summary statistics for each determinand and each sample point are included in R&D Project Record P2-181/PR Appendix 3. Tables 3.1 to 3.4 contain a selection of the mean values obtained for each sample point in each system. Figures 3.1 to 3.10 are boxplots of the dissolved iron and pH data represented for each treatment system. Together these results demonstrate the changes occurring as water passes through the treatment systems. They will be discussed for the individual systems below.

3.1.1 Whitworth No 1

In this system the flow is split into four cells which each discharge separately. These four discharges then combine together at the outlet. From Table 3.1 it can be seen that there are low levels of manganese and aluminium in the discharge and the wetlands remove part of this contamination. There are reductions in sulphate, alkalinity and acidity as the minewater passes through the system. Iron levels are just above 20 mg/l and reduce to just over 3 mg/l with the differing cells performing at differing abilities. In order of achieving the lowest iron concentrations cell 4 is best followed by cell 1, then cell 3, and then cell 2. The two most efficient cells were operating aerobically and precipitating predominantly metal oxides as identified by Wiseman (1997) and Rees (1998). This is discussed further in section 3.2.1.

Figure 3.1 demonstrates the range of iron concentrations observed at each sample point and demonstrates that on the whole all the cells are operating to a fairly similar level. The small spread of the interquartile ranges show that the outlet concentrations of each cell and the final outlet are being achieved consistently. Figure 3.2 demonstrates the pH rising as the water passes through each cell with a similar performance achieved in each part. The combined outlet is slightly more alkaline then any of the four individual discharges and the reason for this is not known.

Table 3.1 Whitworth No 1 mean monitoring data results

Sample point	Flow[1]	pH	Mn total	Al total	Fe dissolved	Fe total	SO_4 total	Alkalinity (as$CaCO_3$)	Acidity (asCO_2)
Minewater	n/s[2]	6.3	1.8	0.41	21.8	23.3	336.3	72.5	38.5
D/s cell 1	n/s	6.7	1.2	0.10	3.2	3.9	319.1	61.6	15.8
D/s cell 2	n/s	6.6	1.4	0.10	3.4	5.5	330.3	66.2	17.9
D/s cell 3	n/s	6.6	1.1	0.07	3.1	4.2	330.3	73.9	18.7
D/s cell 4	n/s	6.6	0.9	0.06	2.7	3.1	323.3	70.5	19.7
Outlet	0.0059	6.9	1.1	0.09	3.1	3.8	288.3	64.5	14.0

Notes: [1] Flow in cumecs, all other results except pH in mg/l
[2] n/s indicates this determinand was not sampled at this point.
Monitoring data from post wetland construction 31/10/95-19/03/02

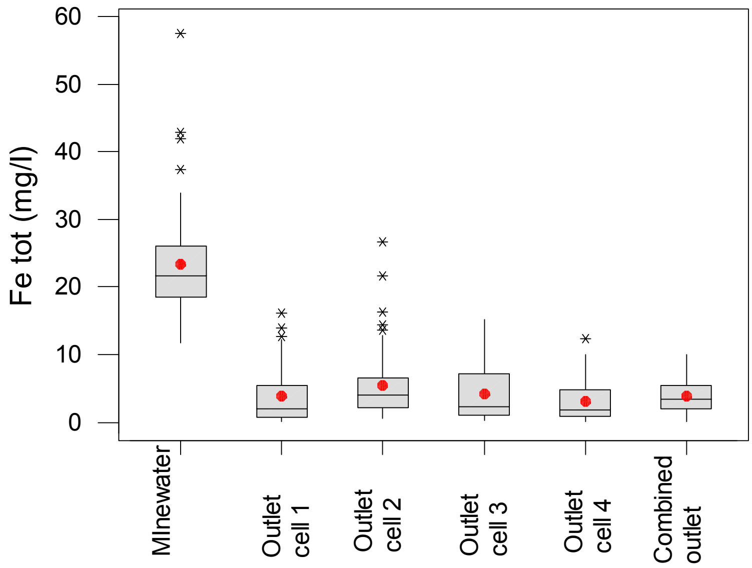


Figure 3.1 Changes in total iron though the Whitworth No 1 system (1995-2002)

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).

Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

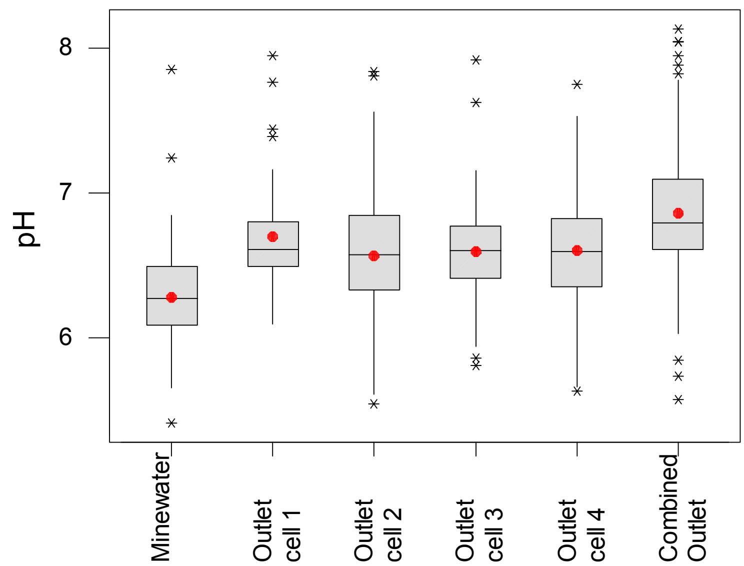


Figure 3.2 Changes in pH through the Whitworth No 1 system (1995-2002)

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).

Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

3.1.2 Garth Tonmawr

Table 3.2 contains a selection of mean results for monitoring work on the Garth Tonmawr wetlands. In this system the minewater flows from the adit through a natural wetland prior to entering cell 1. It then flows over the aerobic cell 1 and then down through the RAPS in cell 2 where it is sampled as it exits the RAPS (d/s cell 2). There is also a surface flow of water over the cell that is sampled (o/f cell 2). This does not therefore receive any alkalinity additions. The two flow routes can be seen in the photograph in Figure 3.3. The flow then goes across cell 3 and through the second RAPS cell (cell 4) in a similar way to cell 2 with a split flow. The water then exits the aerobic cell 5.

Figure 3.3 Flow pathways over and through cell 2 of the Garth Tonmawr system.
Note: Flow through the RAPS exits from the upwelling pipes an example of which is in the foreground. In the center of the wall is the overflow over the surface of the RAPS

Figure 3.4 demonstrates that iron levels dropped quickly through the system. There is a small change within the natural wetland but the majority of iron removal occurs within cell 1. This is an aerobic cell designed to promote the oxidation and deposition of metal oxides and hydroxides. The water then flows over and through cell 2, the RAPS, and from this point the iron levels are low and slightly fluctuating.

There is very little manganese or aluminium in the minewater and the system removes small amounts of these contaminants. Sulphate levels are reduced with passage through the system as a whole but increased after passage through the two RAPS cells (Table 3.2), the reason for this is unknown. Acidity is reduced on passage through the system with reduction occurring most obviously in the first and second cells. Alkalinity decreases by the end of the first cell due to the acid production with the oxidation and removal of the majority of the iron in this cell. This is also demonstrated by the drop in pH from 5.8 at the entrance to the cell to 4.1 at the exit (Figure 3.5). The water at this point can take one of two routes. The water that flows over the RAPS, effectively bypassing it continues to drop in terms of alkalinity with the oxidation of more iron and the pH drops to 3.5. The iron content also drops in the water that flows through the RAPS but concentrations are greater than in the water flowing over the surface. However the alkalinity rises significantly due to limestone dissolution and this significantly raises the pH to 6.5. The water has further additions of alkalinity from the RAPS in cell 4 and small amounts are utilized to counteract further acid generation. By the discharge point there is a pH of 6.8 and 40mg/l of alkalinity remaining with very little free acidity.

Table 3.2 Garth Tonmawr mean monitoring data results

Sample point	Flow[1]	PH	Mn total	Al total	Fe dissolved	Fe total	SO_4 total	Alkalinity (as$CaCO_3$)	Acidity (asCO_2)
Minewater	0.024	5.6	0.69	0.40	31.7	33.3	271.7	30.8	63.7
Inlet cell 1	n/s[2]	5.8	0.68	0.31	28.2	30.1	265.3	30.7	70.2
D/s cell 1 (aerobic)	n/s	4.1	0.66	0.22	8.1	8.6	248.9	8.3	32.6
D/s cell 2 (RAPS)	n/s	6.5	0.59	0.06	2.7	3.0	267.8	76.1	10.3
O/f cell 2 (aerobic)	n/s	3.5	0.71	0.29	1.0	1.1	258.4	0.8	29.1
D/s cell 3 (aerobic)	n/s	5.9	0.62	0.15	1.9	2.4	263.4	27.7	9.5
D/s cell 4 (RAPS)	n/s	7.0	0.40	0.02	0.5	0.7	270.0	65.3	5.9
O/f cell 4 (aerobic)	n/s	6.1	0.46	0.14	0.1	0.2	247.5	30.5	7.6
Outlet (aerobic)	0.020	6.8	0.50	0.09	1.0	1.3	255.2	40.4	4.7

Notes: [1] Flow in cumecs, all other results except pH in mg/l
[2] n/s indicates this determinand was not sampled at this point
Monitoring data from post wetland construction 30/03/99-19/03/02

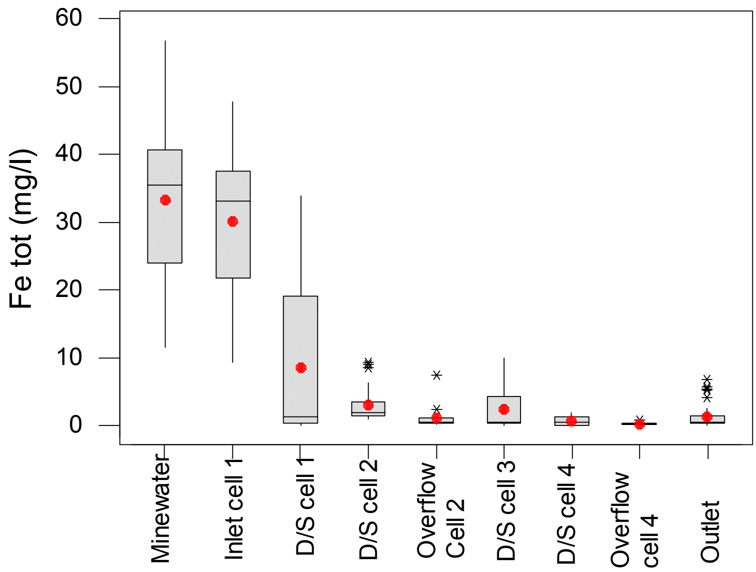


Figure 3.4 **Changes in total iron through the Garth Tonmawr system (1999-2002)**

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).

Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

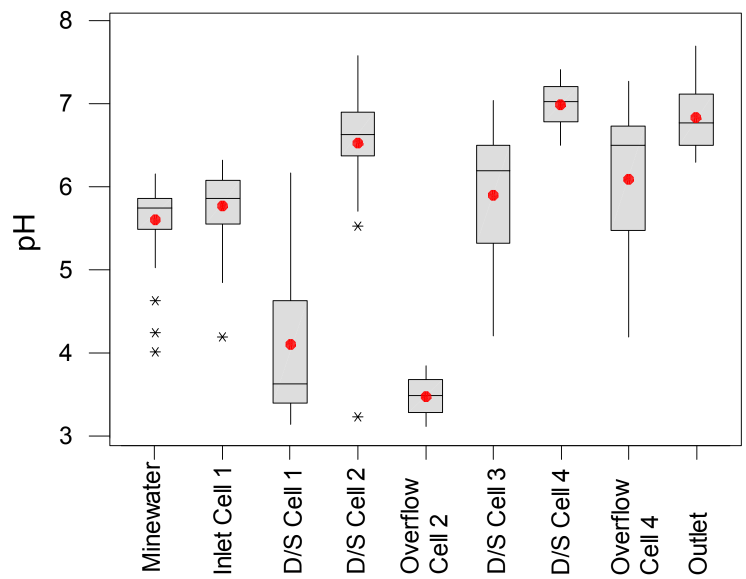


Figure 3.5 **Changes in pH through the Garth Tonmawr system (1999-2002)**

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).

Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

3.1.3 Whitworth A

Mean data from the Whitworth A system is presented in Table 3.3. In this system all the flow entering the system flows initially through a RAPS and then into an aerobic cell prior to discharge. There are low levels of all metals in the minewater apart from iron. Iron removal is shown in Figure 3.6. It can be seen that the RAPS cell removes a significant majority of the iron (91 %). This is not how the system was designed to operate. The original idea was for the RAPS to add alkalinity and remove small amounts of iron prior to iron oxidation and settlement within the aerobic cell. However, the system has spillways for the water entering the RAPS and this oxygenates the minewater promoting iron oxide and hydroxide (iron ochre) precipitation on the surface of the RAPS.

Bullock (1999) studied the performance of this treatment system and also concluded that the majority of iron removal (~68%) was occurring within the RAPS. She suggested that due to the high pH and oxygenation of the standing surface water the majority of this is occurring as iron oxide and hydroxide precipitation on the surface of the cell. Reilly (2001) identified a mean depth of 21.6cm of ochre across the RAPS and the potential implications of this will be discussed in section 3.2.2. A photograph of this ochre accretion on the surface of the RAPS is shown in Figure 3.8. By the end of this project no problems had been observed due to the system operating in this way. Acidity levels are significantly reduced and elevated alkalinity levels are observed both below the RAPS and at the final outlet. Sulphate levels are reduced slightly throughout the system and the pH rises significantly downstream of the RAPS with further increases below at the final outlet as shown in Figure 3.7. The iron removal rate over this period has been consistently very high.

Table 3.3 Whitworth A mean monitoring data results

Sample point	Flow[1]	pH	Mn total	Al total	Fe dissolved	Fe total	SO_4 total	Alkalinity (as$CaCO_3$)	Acidity (asCO_2)
Minewater	0.0075	6.1	0.81	0.02	58.6	59.6	321.5	64.9	95.0
D/S RAPS[3]	n/s[2]	7.0	0.94	0.05	6.6	7.4	322.8	107.0	11.5
Outlet	N/s	7.3	0.50	0.02	1.2	2.0	296.9	110.6	6.1

Notes: [1] Flow in cumecs, all other results except pH in mg/l
[2] n/s indicates that this determinand was not sampled at this point
[3] D/S RAPS indicates downstream of reducing and alkalinity producing cell.
Monitoring data from post wetland construction 22/04/98-19/03/02

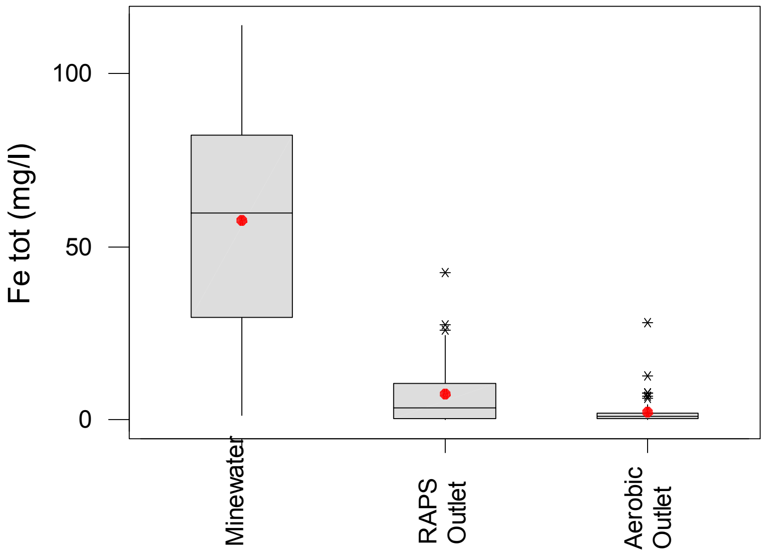


Figure 3.6 Changes in total iron through the Whitworth A system (1998-2002)

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).

Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

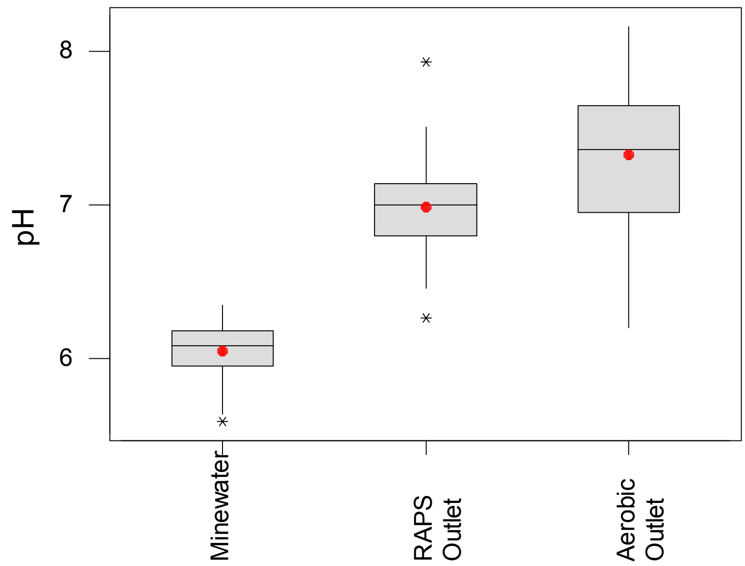


Figure 3.7 Changes in pH through the Whitworth A system (1998-2002)

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit:Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).

Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

Figure 3.8 Iron oxide and hydroxide accumulation on the surface of the Whitworth A RAPS.

3.1.4 Whitworth B and Gwenffrwd

Table 3.4 contains the mean results for the system treating the combined flows of the Whitworth B and Gwenffrwd minewaters. The Whitworth B minewater flows into the RAPS while the majority of the Gwenffrwd minewater flows down the Ochre Accretion Terrace (OAT). These two flows combine and flow through settlement and aerobic cells. There are low levels of manganese and aluminium in the minewaters and roughly half of the loads of these elements were removed by the system. Iron is also in relatively low concentrations and this is removed throughout the system as shown in Figure 3.9. It is relatively difficult to assess where most of the treatment occurs due to the low concentrations and the mixed flow sources and routes. The OAT it is a short spillway feature that primarily oxidizes the water and doesn't remove much iron. The RAPS significantly adds alkalinity and removes iron, especially the dissolved iron. The pH changes demonstrated in Figure 3.10 show that the pH increased after the OAT. It then increased again following the RAPS with the discharge maintaining a pH just below 7.

Table 3.4 Whitworth B and Gwenffrwd mean monitoring data results

Sample point	Flow[1]	pH	Mn total	Al total	Fe dissolved	Fe total	SO_4 total	Alkalinity (as$CaCO_3$)	Acidity (asCO_2)
Whitworth B minewater	0.018	5.6	0.79	0.73	6.8	7.0	112.1	19.3	33.6
Gwenffrwd Minewater	0.0097	6.3	0.20	0.38	1.22	1.44	44.9	17.0	10.7
OAT	n/s	6.8	0.34	0.55	2.4	2.5	68.1	14.3	8.5
RAPS	n/s	7.0	0.45	0.05	0.7	3.5	120.0	94.6	3.7
OAT & RAPS	n/s	7.2	0.63	0.19	1.3	1.4	112.4	64.9	5.1
Settlement Pond	n/s	7.2	0.55	0.18	0.6	1.2	102.0	74.8	3.2
Aerobic	n/s	7.1	0.30	0.14	0.4	0.9	91.7	71.9	3.8

Notes: [1] Flow in cumecs, all other results except pH in mg/l
[2] n/s indicates that this determinand was not sampled at this location
Monitoring data from minewater rerouting into system 01/04/01-19/03/02

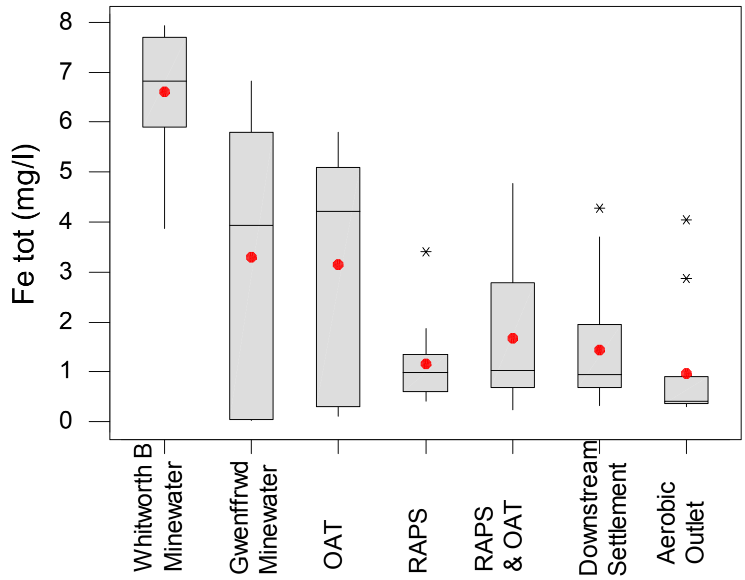


Figure 3.9 Changes in dissolved iron through the Whitworth B and Gwenffrwd system (2001-2002)

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).
Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

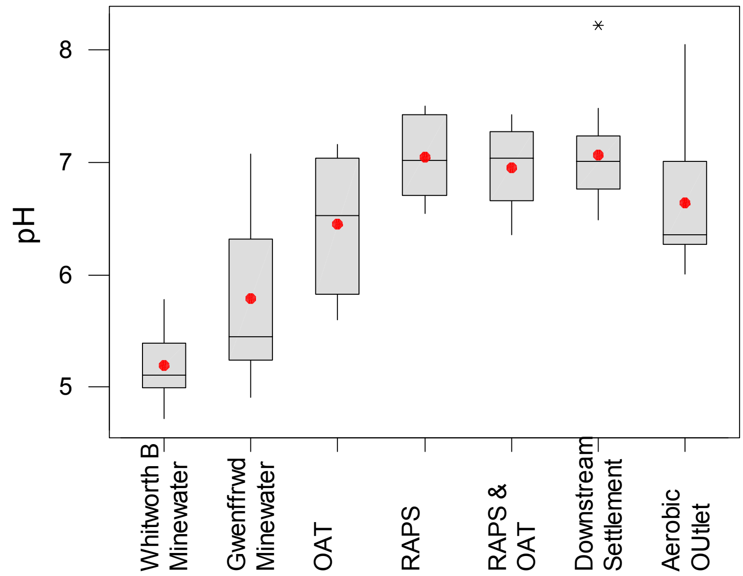


Figure 3.10 Changes in pH through the Whitworth B and Gwenffrwd system (2001-2002)

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).
Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

3.1.5 Flow and loading characteristics

The minewaters in the Pelenna valley have been monitored for a number of years. Generally flow data was taken as monthly spot samples alongside the water quality samples. The minewaters and certain points in the treatment systems have had continuous flow data recorded for a number of years. This allowed correlations to be made between the flows and chemical parameters of the minewaters. This has been undertaken for flow, iron loading, iron content, pH, acidity and alkalinity. Generally few variables correlated well, indicating a variable flow and chemistry regime in all the mine systems. There were only three statistically significant relationships found, these were:
A positive relationship ($p<0.001$) between iron loading and flow in Whitworth No 1 and Whitworth A minewaters.
A positive relationship ($p<0.001$) between iron loading and both dissolved and total iron content (mg/l) in Whitworth A minewater.
A weak negative relationship ($p< 0.005$) between alkalinity and flow in the Garth Tonmawr system.

Whitworth No 1 and Garth Tonmawr systems have continuous flow recorders at the inlets and outlets of the systems. Using this data it is possible to create a water budget for the two systems. The mean flows in and out of the systems and mean water loss are shown in Table 3.5. It can be seen that both the systems use or lose water within the wetlands. TAPIR analysis on the data did not identify any significant trends, seasonal or otherwise in the flows or water usage. It is not known whether all the water use is due to evapotranspiration processes or whether some is lost from the systems by leakages. It seems more likely that it represents loss from evapotranspiration throughout the year, as there is more lost from Garth Tonmawr that has the greater surface area of wetland.

Table 3.5 Water gain/loss through Whitworth No 1 and Garth Tonmawr wetlands

Wetland System	Mean flow in (l/s)	Mean flow out (l/s)	Mean water loss (95% CI's)
Whitworth No 1 (96-00)	5.499	4.18	1.308 (+/-0.1)
Garth Tonmawr (99-02)	20.296	17.71	3.039 (+/- 0.21)

3.1.6 Contaminant removal efficiency

Table 3.6 contains calculated iron loadings and removal efficiencies along with acidity removal efficiency. It can be seen that the ranges of total iron loadings to the watercourse are low, compared to the minewater loadings, varying from 1.1 to 2.6 kg/day. The percentage removal of this loading varies (Figure 3.11). The original demonstration system, Whitworth No 1, has been removing 83% of the incoming iron and it can be seen from the boxplot that its performance has been fairly consistent with an interquartile range primarily between 78-92%. It is not possible to accurately assess which of the four cells performs best, as the individual flows through each cell have not always been known. Therefore the loading could not always be calculated.

The Whitworth B and Gwenffwrd system had a similar percent removal performance to Whitworth No 1, however this is calculated from a smaller dataset and is probably not so much a reflection of the true performance but more a factor of its relatively low loading. If the removal rate in g/day/m^2 is used then it can be seen that this system has by far the lowest removal rate (Figure 3.12). It is likely that this does not reflect poor performance of the system but more the fact it is so oversized for its current loading.

Table 3.6 Iron and acidity removal performance for the wetland systems

Wetland	Minewater total iron loading (kg/day)	Outlet total iron loading (kg/day)	Percent removal of loading	Removal rate (g/day/m^2)	Acidity removal rate (g/day/m^2)
Whitworth No 1	9.3	1.5	83.2	8.1	9.5
Garth Tonmawr	52.8	2.6	95.3	7.9	15.4
Whitworth A	38.8	1.1	96.0	5.8	9.3
Whitworth B & Gwenffrwd	6.92	2.1	82.4	0.9	6.0

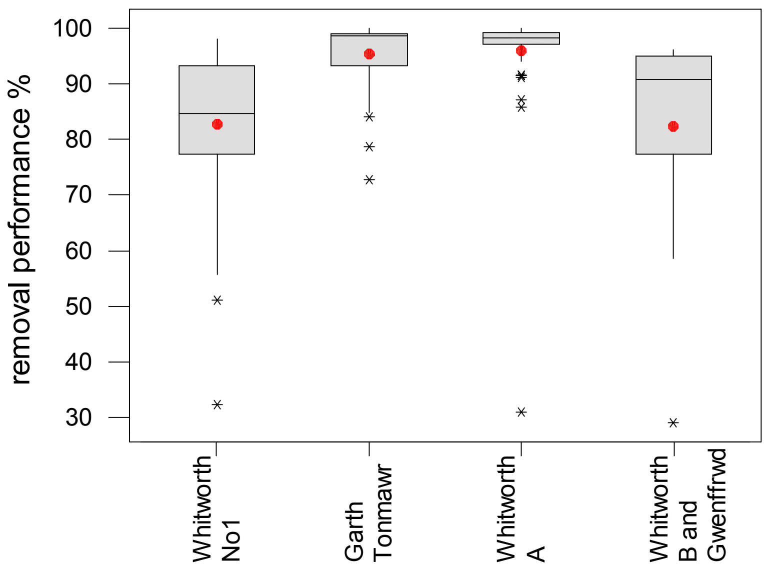


Figure 3.11 Percentage removal of iron loading

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).

Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

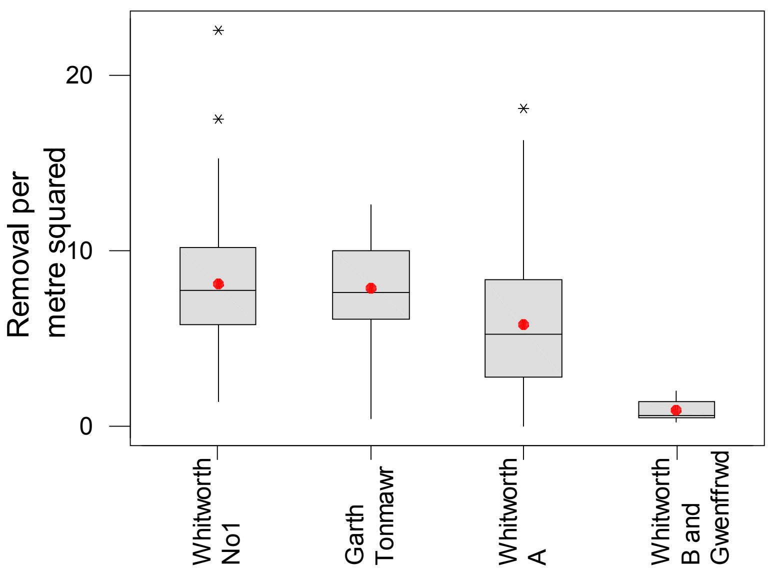


Figure 3.12 Removal of iron loading per metre squared of wetland

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).

Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

The Garth Tonmawr system removes 95% of its iron loading and the Whitworth A system has a similar performance of 96%. The Garth Tonmawr system has performed best in terms of removal per square meter out of these two. Its perfromance is at a level just under that of Whitworth No 1. Figure 3.13 demonstrates the iron loading at the inlet and outlet of the Garth Tonmawr wetlands over time and the percentage removal. It can be seen that this system has coped with a fluctuating iron loading and except for the first winter of operation its performance has been very consistent. Figures 3.14 and 3.15 respectively show similar data for the performance of the Whitworth No 1 and Whitworth A systems. The Garth Tonmawr system removed acidity at a greater rate per meter squared of wetland compared to the other three systems. It has also not suffered from as many operational and maintenance issues as the other systems which is a subject that is covered further in section 3.1.7.

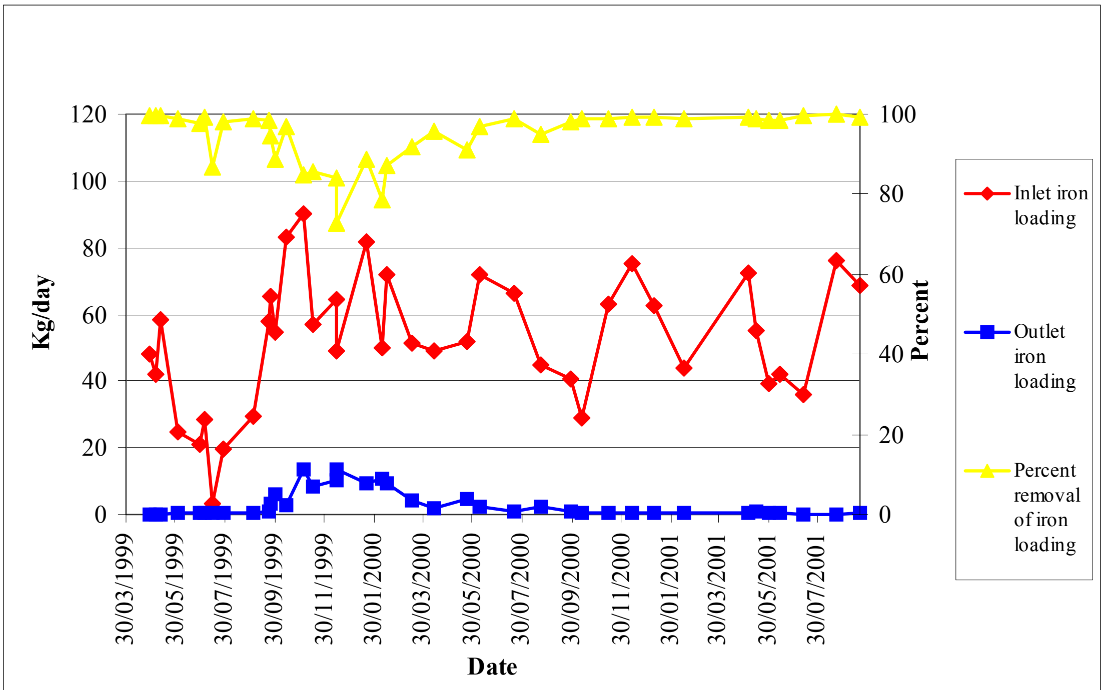


Figure 3.13 Iron loading and percentage removal of the Garth Tonmawr system.

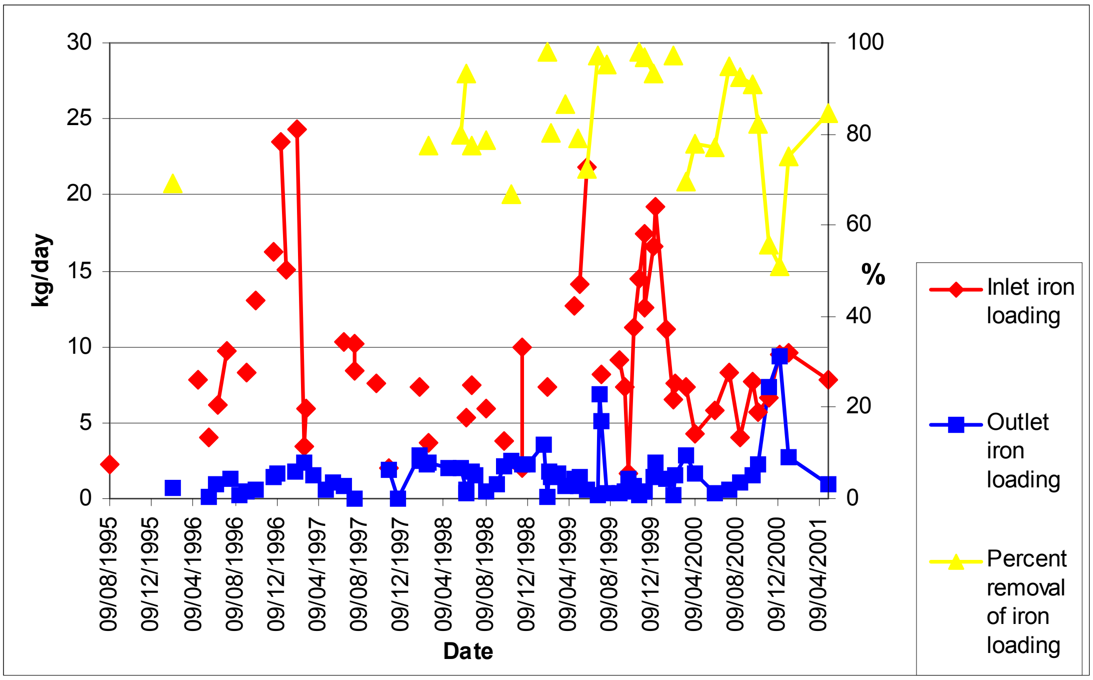


Figure 3.14 Iron loading and percentage removal of the Whitworth No 1 system.

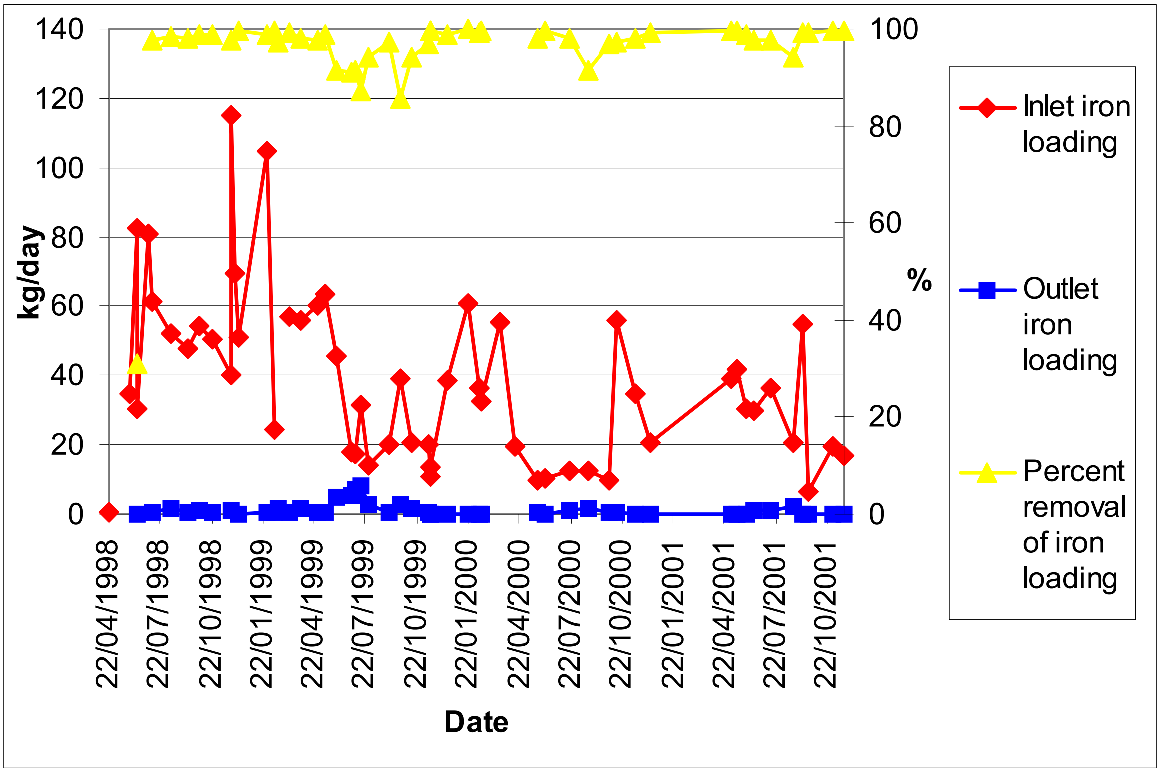


Figure 3.15 Iron loading and percentage removal of the Whitworth A system.

Figure 3.16 demonstrates a boxplot of the percent removal of iron within the Whitworth A RAPS unit and the Whitworth A system as a whole. The RAPS cell is removing approximately 91% of the iron loading from the minewater. The average minewater loading is 38kg/day and the loading at the exit of the RAPS is 3.5kg/day. This gives an indication of the quantity of iron that is being deposited within and on the RAPS. It the RAPS unit is taken on its own it is having a removal rate of 17.8 g/day/m^2 and discharging 7.3 mg/l of total iron.

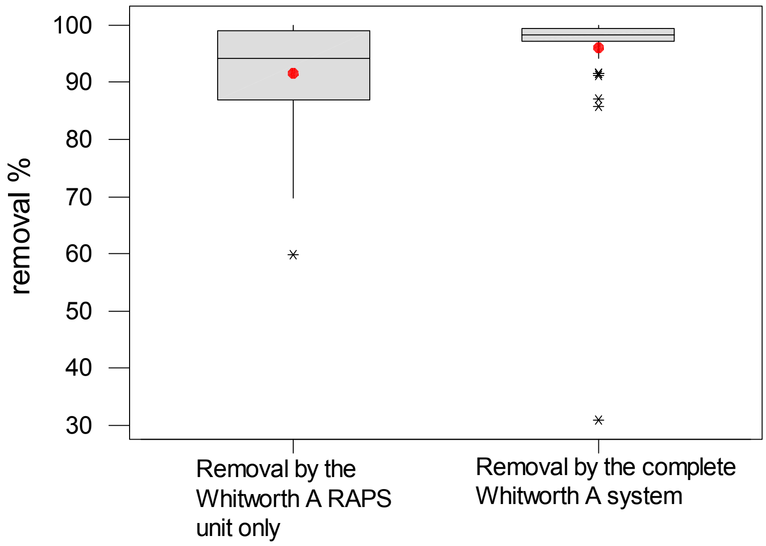


Figure 3.16 Removal performance of the Whitworth A RAPS unit and the complete Whitworth A system.

The TAPIR analysis identifies linear, step and seasonal trends in datasets. There were no significant trends identified in the performance of any of the systems over time. This indicates that the systems appear to be working consistently well. No periods of reduced performance or complete system failure would be expected in the near future if the current trends continue.

3.1.7 Operational issues with the wetlands

During the course of this project many operational issues have been noted with regard to the wetlands. These are primarily to do with the design and operation of the systems. The following sections contain a summary of the problems encountered within the wetland systems. It is hoped that the experiences encountered during this R&D project will lead to some advice that is useful for the design of future systems.

Whitworth No 1

This system has been operating for the longest on the Pelenna. Shortly after construction the initial distribution system (perforated plastic piping) was replaced with an open channel system to stop it blocking so frequently. The main problems encountered in this system have been with the distribution of flow between the four cells. It is a common occurrence (every couple of months) for the inlet and outlet structures to block and cells to either flood or dry up.

The inlet structure consists of a series of weirs and level control structures within a large closed chamber. These structures are overly complicated for the task necessary and slow the flow down allowing ochre to build up. This build up blocks the pipework entering the wetland cells. The flows still go through the treatment system but frequently one or more cells receive no flow. As the chamber is a confined space it requires specially trained teams to enter and clean it out which has implications for the cost and frequency of maintenance. It would be advisable to have the inlet structures as open as possible and with easy access for regular clearing of any ochre buildups. Areas of slack flow should be avoided if possible as this may allow the system to be more self-cleaning.

The outlet structures are again level control structures (Figure 3.17). The flow out of these is often impeded which backs water up in the cells. This frequently reaches the point where it affects the inlet distribution channels. It is not obvious whether it is the outlet flow control structure itself or the pipes that bring water to it that are blocking. However it would appear again that the level control structures are over engineered for this use. The control structures are again placed within a chamber that is a confined space. This limits the ability to access and maintain these structures.

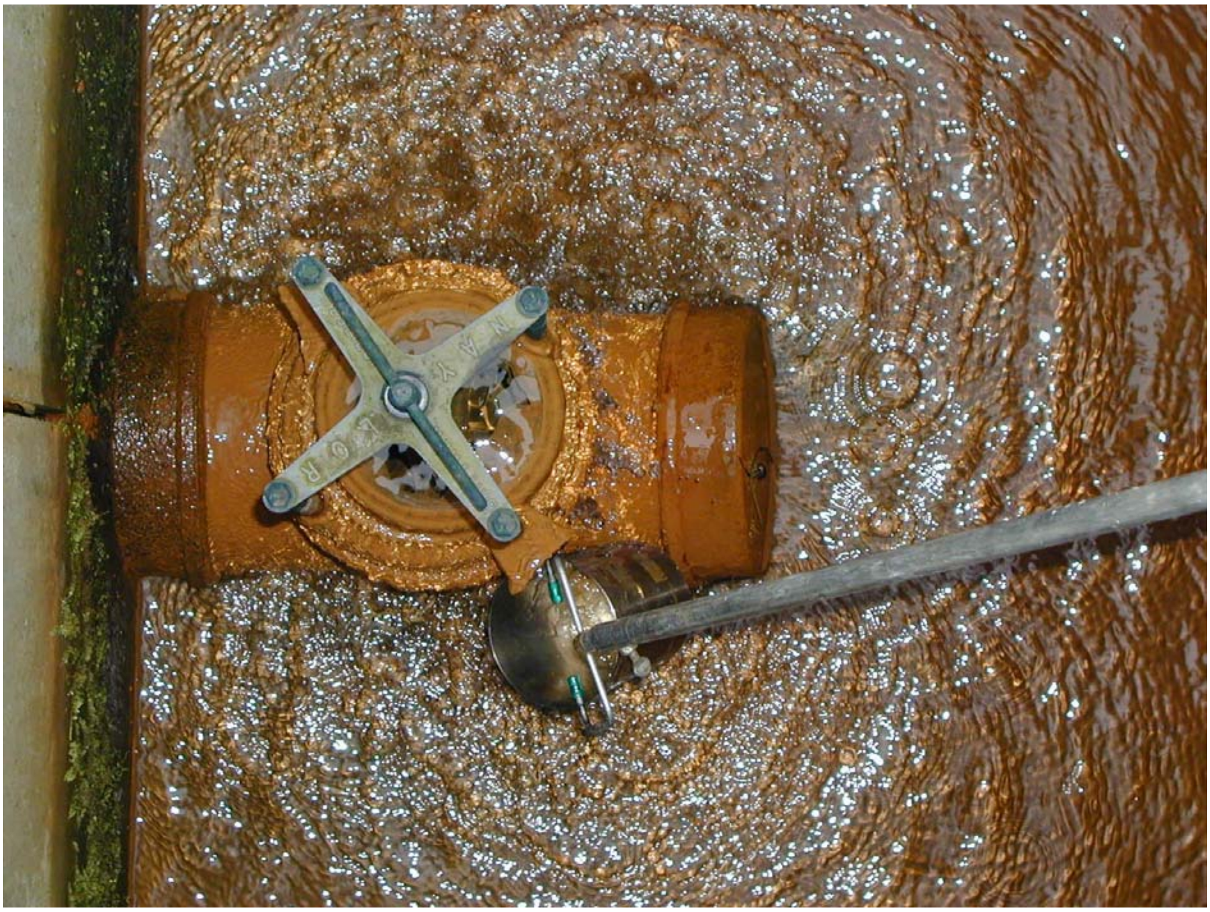


Figure 3.17 Sampling of an outlet level control structure used in the Whitworth No 1 wetland

From a sampling perspective these structures are difficult to obtain samples from at low flows. They are designed to allow water to trickle over the top edge. It would be a great advantage if a notch were provided at one point in the rim to provide a flow that is easy to sample. Again similar overall advice to the inlet structures is relevant. Having a simple flow control structure which is open and accessible would be more beneficial.

It must be noted that while inherent design features frequently cause sections of the system to block, it is a lack of routine maintenance that is exacerbating these problems. If a routine inspection and maintenance programme were in place then these problems could be cleared quickly. This would ensure the performance of the system could be optimized.

The physical integrity of the system has been noted as deteriorating. The cell walls are formed from blocks of concrete sealed with a rubber sealant. This is starting to leak in places but is not yet affecting the performance of the wetland.

Garth Tonmawr

The first problem encountered in this system was a reduction in the downward flow through the RAPS cells. This caused the water in cells 2 and 4 to build up to a depth great enough to overflow the retaining wall and bypass the cell. Specific overflow points were constructed that channeled the flow over in one place rather than allowing it to seep over the whole wall. This should have limited the potential damage from the flow and did allow it to be sampled. The actual reason for the reduction in downward flow is not known. It could be due to the build up of ochre on the surface, a characteristic of the substrate used or a blocking of the pipework in the base of the bed. A proportion of the flow still passes through the RAPS. To date (summer 2002) this has been creating enough alkalinity to buffer the acidity of the minewater. Iron removal has been very good despite these overflow problems. It is therefore thought that no problems should occur if the system continues to operate this way.

In the spring and summer of 2002 the natural wetland that the minewater enters prior to entering the treatment system has been overflowing. This has been caused by the build up of ochreous sediments and the amount of vegetation growth. This overflow has been affecting the river causing localized iron staining. Simply clearing out some of the sediments and plants can rectify this problem.

Generally the system has been operating very efficiently with few problems. It has suffered from a couple of instances of physical damage. This is more of a cosmetic issue rather than a performance one as it is the facing brickwork on the walls that has collapsed. This occurred when a bridge upstream of the wetland blocked with debris under high flow conditions. Water from the river flooded into the systems and overwhelmed the walls causing two sections of facing brickwork to collapse (Figure 3.18)

Figure 3.18 Physical damage of the brickwork at Garth Tonmawr

Whitworth A

The Whitworth A system has been working very well when the minewater is flowing through it. However the inlet structure is very prone to blocking. This consists of a weir arrangement that allows the minewater to be piped into the system. During high minewater events, excess flows bypass the system. During normal flow conditions all the flow should enter the wetlands. Ochre has been building up within the inlet structures and especially in the pipe that leads to the wetlands. This has frequently reduced the flow to the point where the majority of it bypasses the system. It then discharges effectively untreated into the Nant Gwenffrwd. These significant blockages occurred most dramatically in the spring and summer of 2000 and 2002.

There are two main reasons for this ochre buildup becoming a problem: The first is the design of the system. Having a long submerged pipe makes clearing any blockages more difficult and inspection of the degree of ochre buildup impossible. The second problem is again a lack of routine inspection and maintenance of the inlet structures. Frequent clearing of the ochre from the inlet structure and clearing of the pipe would be beneficial. It would ensure that the whole minewater flow entered the system more frequently and therefore optimize the performance of the system.

It would be recommended that structures taking water from its point of emergence to a treatment scheme should be open channels where possible. This will allow much easier maintenance and inspection and should be less prone to blocking. This approach was utilized successfully to transfer the Whitworth B minewater as described in the next section.

In 2001 the bund on the aerobic cells of the phase III wetlands was lined with a plastic liner. This was in response to erosion and flow distribution problems. The bund had originally been unlined and as such was permeable to water.

Whitworth B and Gwenffrwd

Shortly after construction of the phase III wetlands the underground flow of the Gwenffrwd minewater shifted and came out of the Whitworth B adit. This then bypassed the treatment system and discharged effectively untreated into the Nant Gwenffrwd. Work undertaken in March 2001 rerouted this minewater flow by way of a mostly open channel into the combined Whitworth B and Gwenffrwd leg of the phase III system. Since this point there have been no problems with this system.

3.2 Internal Processes and their Sustainability

3.2.1 Whitworth No 1

This system has been and still is the location for many studies into the internal processes occurring within passive treatment schemes. Due to it being the oldest of the Pelenna wetlands it has received more in depth analysis than the later phases. During the course of this project there have been two MSc students working with the EAW in the summers of 2000 and 2001. They investigated the internal wetland processes and sustainability at Whitworth No 1 and Whitworth A wetlands (Reilly 2002, Thomas 2001). Various other projects have investigated the Whitworth No 1 wetlands including (Davies 2002, Hicks 1997, Hussey 1997, Rees 1998, Wiseman 1997). A PhD is currently underway at the University of Leeds investigating the internal processes and sustainability of the Whitworth No 1 wetlands (Emma King (PhD student) personal communication).

Iron deposition chemistry

The Whitworth No 1 wetlands were designed with different substrates and plants in each cell and engineered to have differing flow types and redox environments. This would then have promoted either oxidation or reduction iron removal processes in each cell. Work within this R&D project presented in Thomas (2001) has investigated the buildup of iron within each cell and attempted to assess the removal processes occurring. This built on the findings of Rees (1998) and Wiseman (1997). The studies are all in agreement that the cells are not behaving as designed but the redox environment and therefore removal processes are being governed primarily by substrate type. The cohesive mushroom compost in cells 2 and 3 is actively encouraging a reducing anaerobic environment to form. By contrast the larger particles of the bark mulch have allowed more air ingress into the sediments. The redox profiles of cells 1 and 4 measured on site in cores taken by Wiseman (1997) and Thomas (2001) were primarily indicative of aerobic oxidising environments. Table 3.7 contains the mean redox potentials for each cell.

Table 3.7 Mean Redox potential of the four Whitworth No 1 cells in 1997 and 2000.

Cell	Wiseman (1997)	Thomas (2000)
1	50	436
2	-140	23
3	-112	-193
4	220	364

In all the sediment cores taken from the Whitworth No 1 cells the majority of iron was found at the surface. Figure 3.19 demonstrates the results from Wiseman (1997) that indicate the relative distribution of iron fractions with depth. Rees (1998), Thomas (2001) and Wiseman (1997) used sequential extraction methods on the samples to identify the types of iron that were being deposited. Figure 3.20 contains the results from Wiseman (1997) for the relative proportions of each iron fraction found in each cell. Figure 3.21 contains similar data from Thomas (2001) for purely the organically bound and non-crystalline and crystalline oxides. All three researchers found that iron oxide and hydroxide deposition was the main removal process. This is obviously more likely to occur on the surface of the sediment where there is more interaction with the air. This matches the observed findings in the field of ochre deposition primarily on the surface of all the cells.

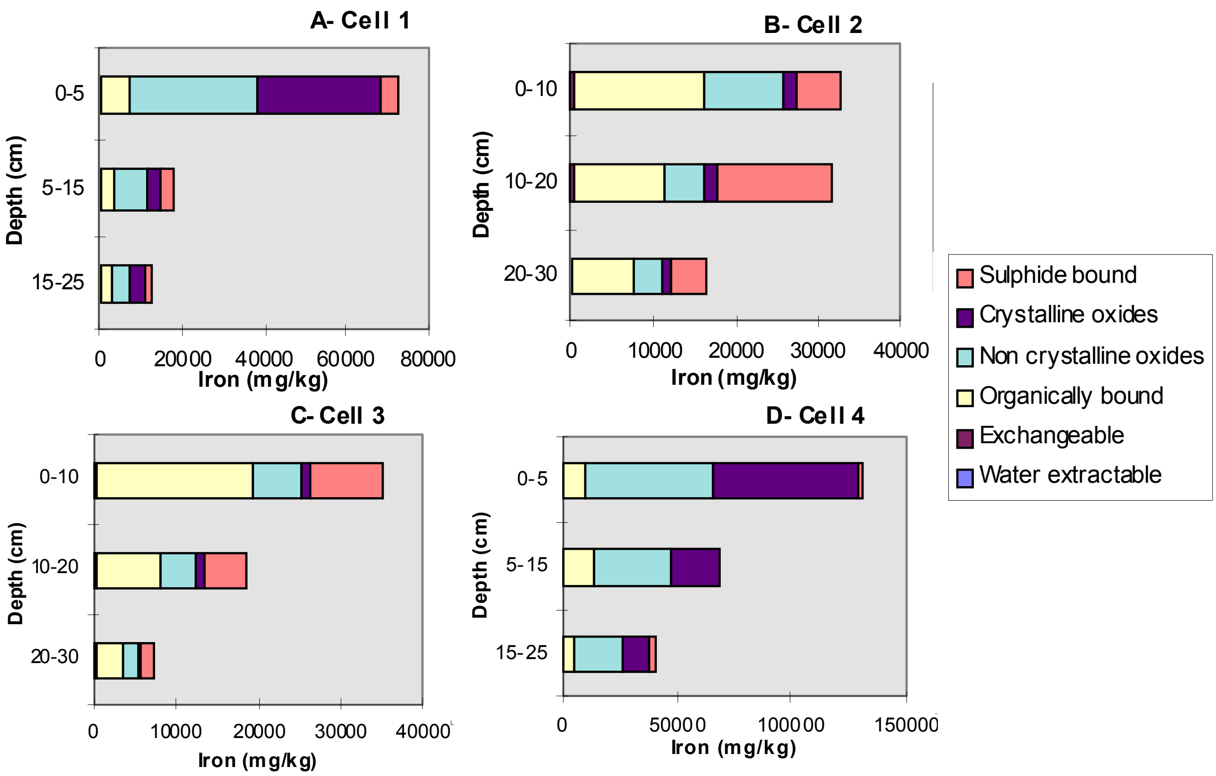


Figure 3.19 (a-d) Variations in the amount of iron with depth in the Whitworth No 1 wetlands (Wiseman, 1997)

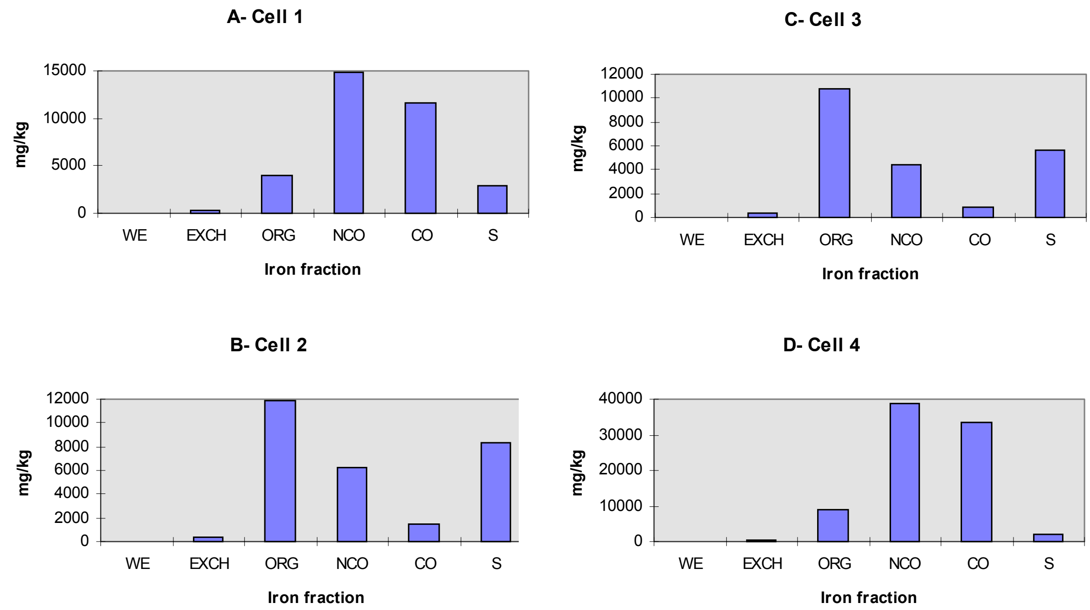


Figure 3.20 (a-d) **Mass of each iron fraction observed in each cell of the Whitworth No 1 wetlands (Wiseman, 1997)**

Note: WE = Water extractable; EXCH = Exchangeable iron: ORG = Organically bound iron; NCO = Non crystalline oxide bound iron; CO = Crystalline oxide bound iron; S = Sulphide bound iron.

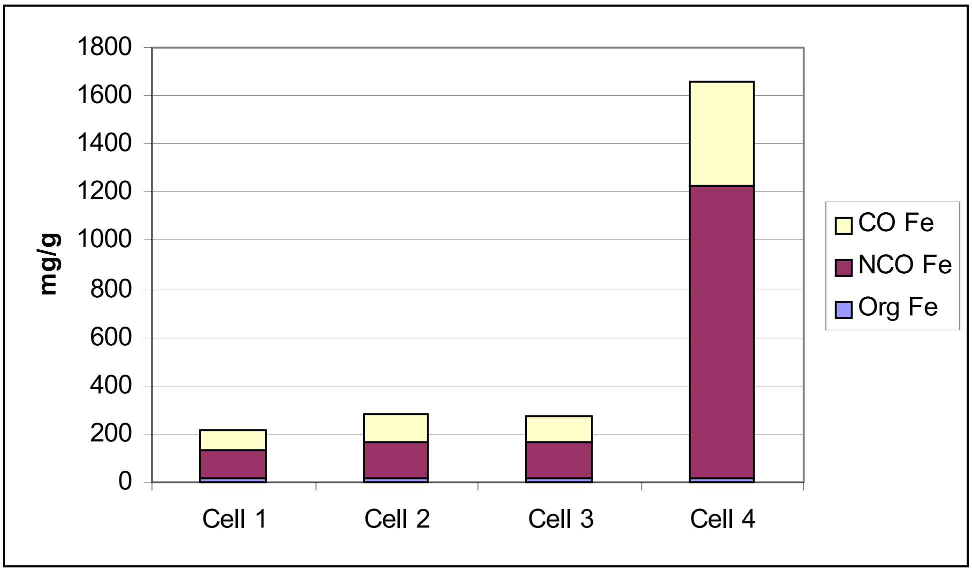


Figure 3.21 **Mass of each iron fraction observed in each cell of the Whitworth No 1 wetlands (Thomas, 2001)**

Bacterial sulphide reduction

Wiseman (1997) identified greater iron sulphide precipitation within cells 2 and 3 that were the primarily anaerobic environments formed by the mushroom compost. Rees (1998) who identified more sulphide fractions in these cells then confirmed this. He investigated this further looking at the extent of BSR within the system. By investigating the pore water profiles and specifically changes in sulphate concentrations with depth he found that the sulphate was being reduced at depth in cells 2 and 3. The mushroom compost appeared to provide a more useful source of organic carbon for bacterial activity than the bark mulch. Both substrates had similar amounts of organic carbon, but the mushroom compost had a higher fraction of labile organic carbon compounds such as cellulose and lactic acid that decay relatively easily. From these findings it was concluded that mushroom compost is more able to support BSR than the bark mulch and would therefore encourage greater iron sulphide precipitation.

Rees (1998) raised concerns over the longevity of the BSR process due to a reduction in the availability of an organic carbon food source for the reducing bacteria. He attempted to calculate how long it may last and came out with a result of 40 years, which he suggested was a massive overestimate due to a number of large assumptions. No attempt to recalculate this has been made, as there are too many unknown variables, especially the rate of renewal of organic matter with vegetation dieback each year. It has been noted that with a lack of BSR iron sulphide will not form. This however is not likely to have a major effect on treatment performance as iron oxidation was identified to be the major removal process (Rees 1998, Thomas 2001, Wiseman 1997). As this occurs mainly on the surface of the wetland the available space will be the limiting factor on iron removal.

Rate of iron accumulation

In 1997 Wiseman identified between 5-10 cm of ochre on the surface of the wetlands. In 1998 Rees identified an average depth of ~10cm. From these a rate of just over 3cm of ochre build up a year can be inferred. By 2001 this would have caused 20cm of ochre to be deposited. However from the cores taken by Thomas in 2001 the average depth was 10-15 cm of ochre which would suggest a rate of build up at least half of that suggested previously. The main reason for this is likely to be compaction of the ochre as it accumulates. This obviously occurs but is difficult to quantify. So assuming a build up rate with compaction of 1.5 cm a year the lifespan of the wetlands can be calculated. There is currently (2002) approximately 30 cm of space from the current substrate surface to the level of the inlet structures. This would therefore take 20 years to fill. It must be noted that this calculation does not allow for any organic matter that may build up if the rate of accumulation of plant debris exceeds the rate of organic substrate breakdown. It should also be noted that in order to achieve this level of sediment buildup the water surface in the cells would have to be maintained at the sediment surface level. This would then avoid the water backing up into the inlet chambers and causing ochre accretion and blockages.

Internal process change

The above calculation also assumes that the other processes within the Whitworth No 1 wetland remain as they are now. If there is a reduction in iron or acidity removal the system could become ineffective at treating the discharge. Rees (1998) noted that with a reduction in BSR due to a lack of useable organic carbon there is potential for remobilisation of sulphide bound iron as it becomes anaerobically oxidised by reaction with Fe^{3+}. Rees calculated tentatively from the available dataset what stage in the process the system was at and concluded that in 1998 cell 3 was likely to have crossed into sulphide oxidation. There exists a large volume of monitoring data following this date and no problems in terms of declining performance have yet been encountered. It would seem likely that insufficient iron is becoming remobilised to affect the discharge, or that it is being reprecipitated as oxides and hydroxides. Again if this is the case we return to the physical space being the limiting factor on lifespan.

The other removal process of note is that of acidity. Whitworth No 1 minewater is net alkaline and as such is unlikely to cause an acidic discharge following iron oxidation and removal. To date the pH of the discharge has been greater than that of the inlet. This has been suggested as being a balance of bacterially generated alkalinity and limestone dissolution from within the bark mulch (Rees 1998, Thomas 2001). With a reduction in BSR rates alkalinity generation could decrease however as noted previously no time frame for this is readily available. Rees (1998) analysed the inorganic carbon content within the mushroom compost substrate as this inorganic carbon is totally derived from a limestone source. From this he calculated that the small amount of limestone in the substrate of cell 3 would last a maximum of 9 years, and even this is likely to be an overestimate. This work is being re-assessed as part of a PhD project at the University of Leeds (Emma King, personal communication). It is believed that if there were no limestone derived alkalinity generation then there would not be significant pH effects on the discharge. This can be inferred from the fact that the bark mulch cells do not contain limestone and do not have discharges significantly different from cells 2 and 3. Therefore if there are no foreseen changes to the pH discharge regime then again the physical space limitations will be the main limiting factor. As noted above this would give a tentative estimated lifespan of 20 years from now or 27 years total lifespan. The scheme would therefore be filled with sediments and need substrate removal in 2022.

Physical integrity

Obviously the projected lifespan mentioned in the previous section gives no consideration to non-process based factors such as physical damage to the systems. It has been observed that the inlet distribution and discharge structures frequently block with ochreous deposits. This should not be an issue as long as they are regularly cleared and do not solidify to produce permanent blockages. It has also been observed in 2002 that in at least one place the rubber sealant between the concrete blocks of the cell walls has broken. This should not affect the treatment quality at its current location but may have structural implications if more areas were affected. However, inspection and repair of the physical integrity of the system if necessary may prevent more costly work in the future.

Vegetation development

The vegetation in the Whitworth No 1 system has changed from the monocultures of cat's-tail *Typha latifolia* and soft rush *Juncus effusus* that were originally planted. Jointed rush (*Juncus articulatus*), compact rush (*Juncus conglomeratus*) and bistort (*Polygonum amphibium)* are the main species that have developed within the cells. These have colonised most of the cells in Whitworth No 1, not however at the expense of the planted vegetation.

3.2.2 Whitworth A

The Whitworth A system was designed to add alkalinity in the RAPS and then precipitate iron by oxidation within the aerobic cell. The alkalinity would raise the pH and promote the oxidation process. It would also counteract the acidity produced by the oxidation reactions. However it has been noted from the water quality data that the majority of iron removal is occurring within the RAPS unit (91%). From visual observations of the RAPS it appears to be removing iron by oxide and hydroxide precipitation on the surface of the system (see Figure 3.8 in section 3.1.3). It was thought possible that a significant proportion could have been forming as iron sulphides within the organic substrate. However Bullock (1999) and Reilly (2002) have both identified little change in the sulphate concentration throughout the RAPS. This would be expected to drop if sufficient sulphides were forming. Results in section 3.1.3 of this report also confirm this finding.

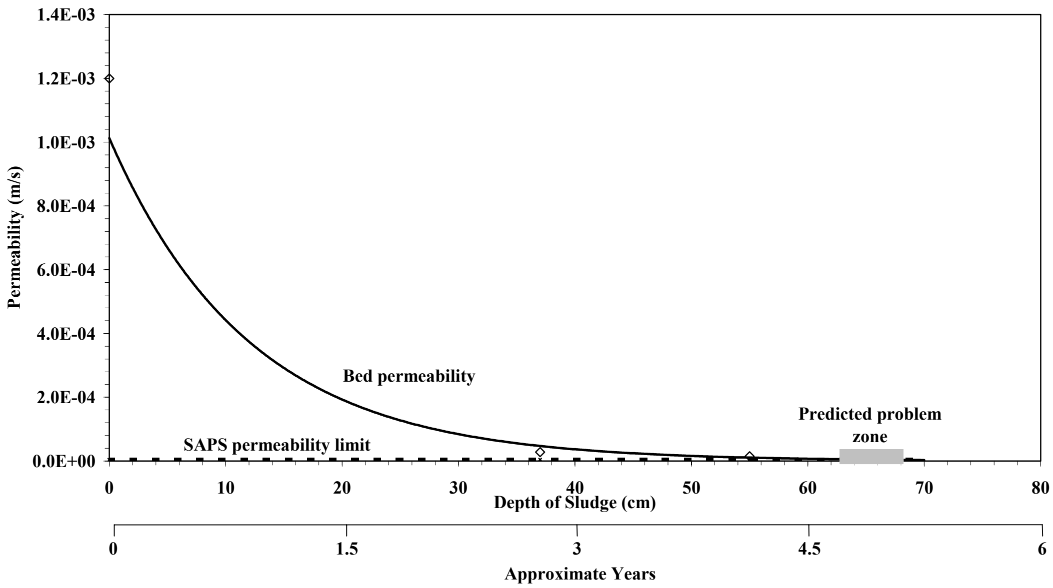
Iron deposition: Permeability effects and the longevity of Whitworth A RAPS

The majority of work done on the processes occurring in Whitworth A has concentrated upon the unplanned volume of iron oxide and hydroxides precipitating on the surface of the RAPS. This is important both because it is the dominant removal process as well as being the biggest concern to long term performance and treatment lifespan. Dey and Williams (2000) from Cardiff University initially highlighted and studied the potential negative effects of this ochre accumulation.

The build up of iron oxide and hydroxide precipitates on the surface of the RAPS cell has the potential to negatively affect its performance and especially lifespan in two main ways:

- The ochre will be compacted as its depth increases, with increased compaction the permeability and hydraulic conductivity should decrease. If the permeability falls to the stage where downward flow through the RAPS is impeded then the cell will overtop through the overflow structures. This will obviously then reduce residence and treatment time in the system as a whole and minimize the alkalinity production from the RAPS.
- If the permeability and hydraulic conductivity do not increase to a point where they impede downward flow then the system can continue to accrete ochre. This will continue until the depth is great enough for the water pooled on top of the ochre to overflow the system. This again would reduce the residence and treatment time of the wetlands and reduce alkalinity generation.

Dey and Williams (2000) created a falling head permeability model of a section of the system in their lab. They used ochre collected from the wetlands over an organic and limestone substrate and assessed how the permeability would change with increased ochre depth. Their work created an equation for bed conductivity changing with increased ochre depth. Using Darcy's equation they determined the permeability requirement of the RAPS to permit an 8l/s flow. When these two are plotted a cross over or end period is indicated as is shown in Figure 3.22. This is at approximately 65cm of ochre depth. They calculated how long this would take to be accumulated based on their observations of 20cm ochre depth accumulation after 1.5 years. This gave the worst case scenario figure of 4.5 to 5 years time before the permeability would be too great for downwards flow (Figure 3.22). It must be noted that they presented this work as preliminary findings and effectively 'work in progress' and indicated that more work needed to be done.



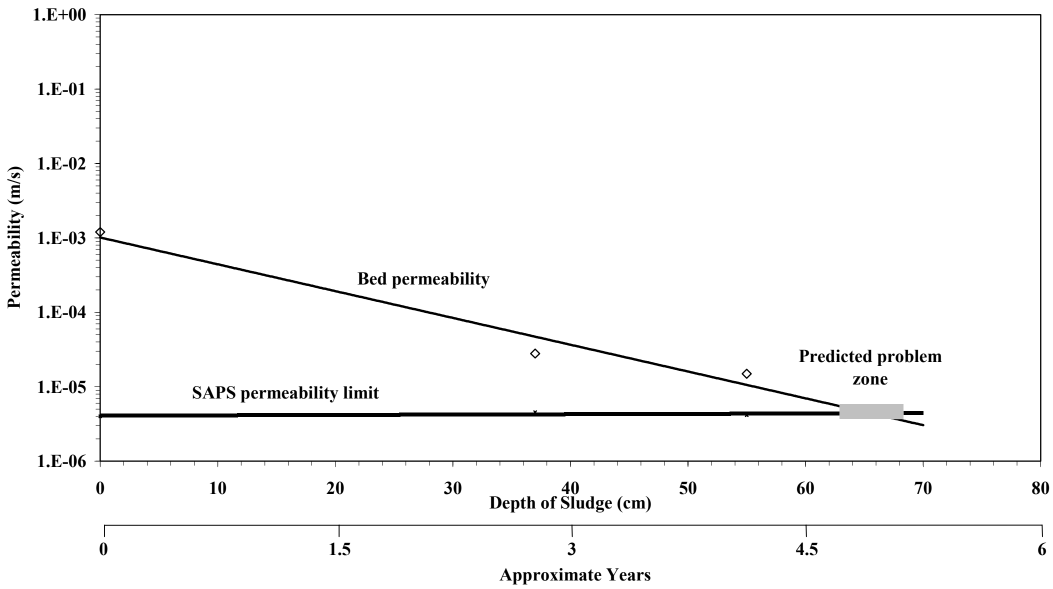


Figure 3.22 Approximate prediction of RAPS life using decreasing permeability data on linear and logarithmic scales (Dey and Williams, 2000)

Following the findings of Dey and Williams (2000) a further investigation of Whitworth A RAPS was made during 2001 as part of the R&D project. It was undertaken as part of the MSc project of Reilly (2002). A set of 18 cores were extracted from the RAPS unit. Five of these were used to assess how permeability changes with increasing depth of the ochre layer. These studies were carried out using a falling head permeameter described by Fetter (1994). Hydraulic conductivity (k) was calculated using Darcy's equation. Figure 3.23 shows the distribution of permeability (hydraulic conductivity) of the cores against the depth of ochreous material in the core. The depth of the ochreous material was used instead of the total depth of the core as visual inspection showed that the organic material had not been significantly broken down and remained as large fractions of wood and bark. Permeability studies carried out on organic material have shown such material to be relatively permeable.

It was, therefore, considered that the permeability of the cores would be largely dependent on the ochreous material.

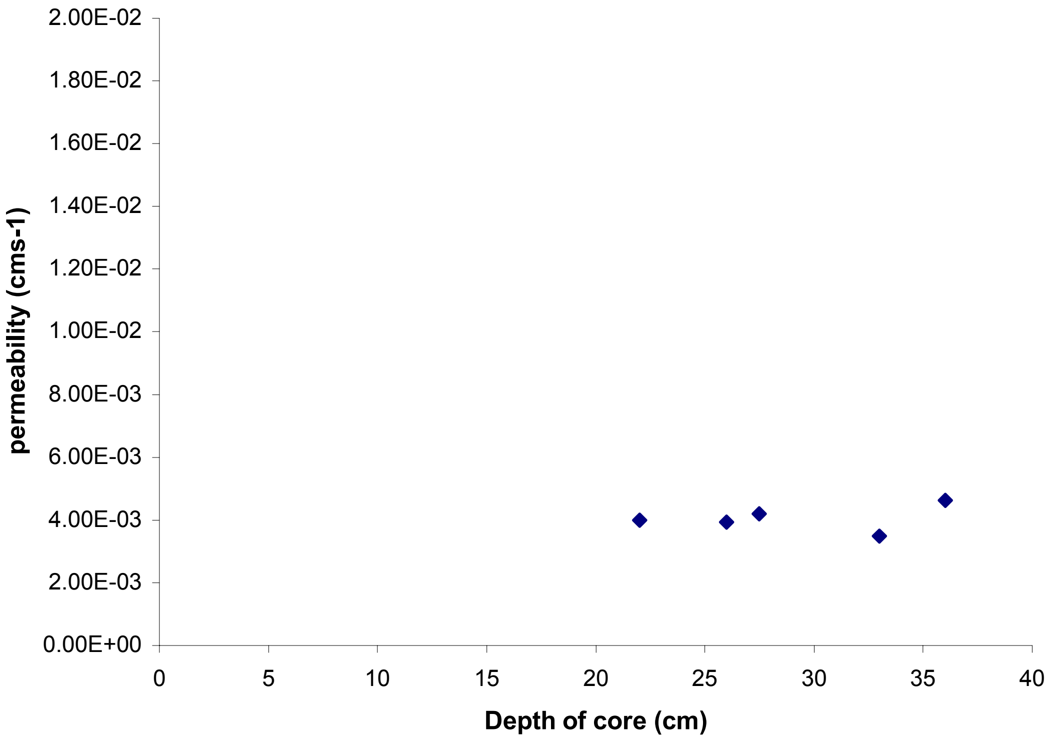


Figure 3.23 Permeability variations with changing depth of ochre in the core (Reilly, 2002)

The relationship between the two variables (r^2 =0.06, P=0.69) is not significant. This therefore indicates that the permeability does not significantly change with increasing depth of ochre. Dey and Williams (2000) found a negative relationship between permeability and sample depth in their permeability study using ochre collected from the Whitworth A RAPS

A negative power curve would be expected for the relationship between permeability and ochre depth in the Whitworth A RAPS cell. When the cell commenced operation any ochre would have been porous and permeable as no compaction processes were in operation. As the ochre built up compaction processes would have resulted in reduced porosity and permeability of the cell. The amount of compaction will increase with increased ochre depth to a point where the sediment cannot be easily compacted any further by the weight of ochre alone. This will result in a stabilisation of the permeability of the ochreous material. The relationship between permeability (hydraulic conductivity) and depth of ochre in the Whitworth A RAPS is illustrated in Figure 3.24. The figure uses a value for virgin substrate obtained by Dey and Williams (2000). This is combined with the results of the permeability studies and produces a statistically significant negative power regression (r^2 = 0.982, $P < 0.001$).

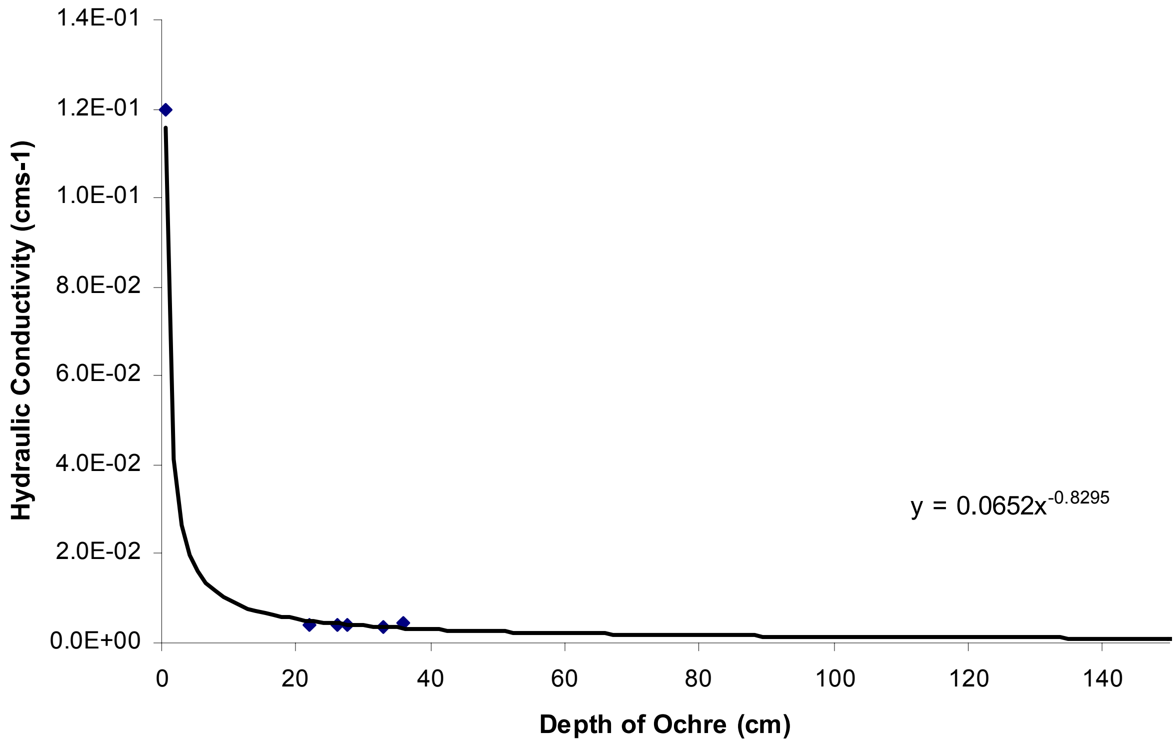


Figure 3.24 Regression relationship between depth of ochre and hydraulic conductivity (Reilly, 2002)

Utilising this permeability work Reilly (2001) investigated how long it would take for the permeability to develop to the point where downward flow were likely to be impeded. Dey and Williams (2000) identified a bed permeability limit for the RAPS of the order $4x10^{-6}$ to $6x10^{-6}$. From Figure 3.24 and the regression equation it contains it can be seen that it is unlikely that the hydraulic conductivity will reach this permeability limit. It has also been suggested that older ochre could crystallise and this may increase its permeability (Ben Rees, personal communication). Therefore it is possible that the Whitworth A RAPS could continue to build up ochre until the physical space available is utilised.

Iron deposition: physical accumulation and the longevity of Whitworth A RAPS

The overflow pipe at the Whitworth A RAPS will be breached as a result of one of two scenarios:

- The permeability of the bed will be reduced to such an extent that a large head will form and breach the overflow.
- The permeability will not be significantly reduced and a small head will breach the overflow pipe as the cell reaches its iron retention capacity.

Based on their preliminary work, Dey and Williams (2000) calculated that as a worse case scenario it would take 4.5 to 5 years time for the 65cm of ochre to build up before the permeability would be too great for downwards flow (Figure 3.22). Again it must be noted that they presented this work as preliminary findings and were planning more studies. They identified a bed permeability limit for the RAPS of the order $4x10^{-6}$ to $6x10^{-6}$. As mentioned above it is unlikely that the hydraulic conductivity will reach this permeability limit.

Therefore it is perhaps less likely that the first scenario will occur. Obviously the second scenario mentioned would be preferable, as the longevity of the system will be maximised.

It is possible to attempt to identify when this depth of ochre may build up. The average depth of ochre in the cores in 2001 was 21.6cm. The remaining capacity of the wetland in August 2001 was 82.5cm. The total capacity of the system was thus 104.1cm and the system will fail when the depth of ochre plus the hydraulic head reaches this value. Using Darcy's equation and the regression line in figure 3.24 the hydraulic head can be calculated for a given depth of ochre. Thus the depth of ochre when the system fails can be established. Table 3.8 shows the depth of ochre, the hydraulic head, and the sum of these variables. As the discharge rate of the minewater is not constant the data in Table 3.8 was calculated using the mean minewater discharge, and the upper and lower 95% confidence intervals for minewater discharge. The highlighted results in Table 3.8 show that the Whitworth A RAPS will fail when the depth of ochre is between 79cm (upper confidence limit) and 84cm (lower confidence limit).

Table 3.8 The critical depth of ochre in the Whitworth A RAPS for the mean and 95 percentiles of the flow (Reilly, 2002)

		Mean Flow		**Lower 95% confidence interval**		**Upper 95% confidence interval**	
Ochre (cm)	Hydraulic conductivity (cms-1)	Head	Head Ochre	+ Head	Head Ochre	+ Head	Head + Ochre
25	0.00451	2.70	27.70	2.25	27.25	3.15	28.15
30	0.00388	3.76	33.76	3.14	33.14	4.39	34.39
35	0.00342	4.98	39.98	4.15	39.15	5.81	40.81
40	0.00306	6.36	46.36	5.30	45.30	7.43	47.43
45	0.00277	7.91	52.91	6.59	51.59	9.23	54.23
50	0.00254	9.58	59.58	7.98	57.98	11.18	61.18
55	0.00235	11.39	66.39	9.49	64.49	13.30	68.30
60	0.00218	13.40	73.40	11.16	71.16	15.63	75.63
65	0.00204	15.51	80.51	12.92	77.92	18.10	83.10
70	0.00192	17.75	87.75	14.79	84.79	20.71	90.71
75	0.00182	20.06	95.06	16.71	91.71	23.41	98.41
76	0.0018	20.55	96.55	17.12	93.12	23.98	99.98
77	0.00178	21.06	98.06	17.54	94.54	24.57	101.57
78	0.00176	21.57	99.57	17.97	95.97	25.18	103.18
79	0.00174	22.10	101.10	18.41	97.41	25.79	104.79
80	0.00172	22.64	102.64	18.86	98.86	26.42	106.42
81	0.0017	23.19	104.19	19.32	100.32	27.07	108.07
82	0.00169	23.62	105.62	19.68	101.68	27.56	109.56
83	0.00167	24.19	107.19	20.16	103.16	28.23	111.23
84	0.00165	24.78	108.78	20.65	104.65	28.92	112.92
85	0.00164	25.23	110.23	21.02	106.02	29.44	114.44
90	0.00156	28.08	118.08	23.40	113.40	32.77	122.77
95	0.00149	31.04	126.04	25.86	120.86	36.22	131.22
100	0.00143	34.04	134.04	28.36	128.36	39.72	139.72

It can be assumed that the vast majority of material in the ochreous layer was composed of iron compounds. Thus, it is these iron compounds that determine the permeability of the Whitworth A RAPS cell. Figure 3.25 shows a significant correlation ($r^2 = 0.44$, $P = 0.003$)

between the depth of the core sample and the mass of iron in that sample. Given the prediction that an addition of 79-84 cm of ochre may result in the failure of the cell. Then figure 3.25 can be used to calculate that this depth of ochre will have a mass of between 232.9 kgm^{-2} to 249.7kgm^{2}. Thus, the Whitworth A RAPS can accommodate a further 274 to 301 tonnes of iron.

The volume and mass of the cores was measured and the concentration of iron within these had been identified. Using this information the mass of iron in each sediment core taken from the RAPS was then calculated. Utilising this and the area of the wetland it was possible to work out the mass of iron in the ochre layer. The ochre layer had a mass of 64.37 Tonnes (Reilly, 2002). This had developed by the summer of 2001 in 3.33 years of operation. Thus, at the same rate of iron accumulation it will take a further 14.2 to 15.6 years for the Whitworth A RAPS cell to reach capacity.

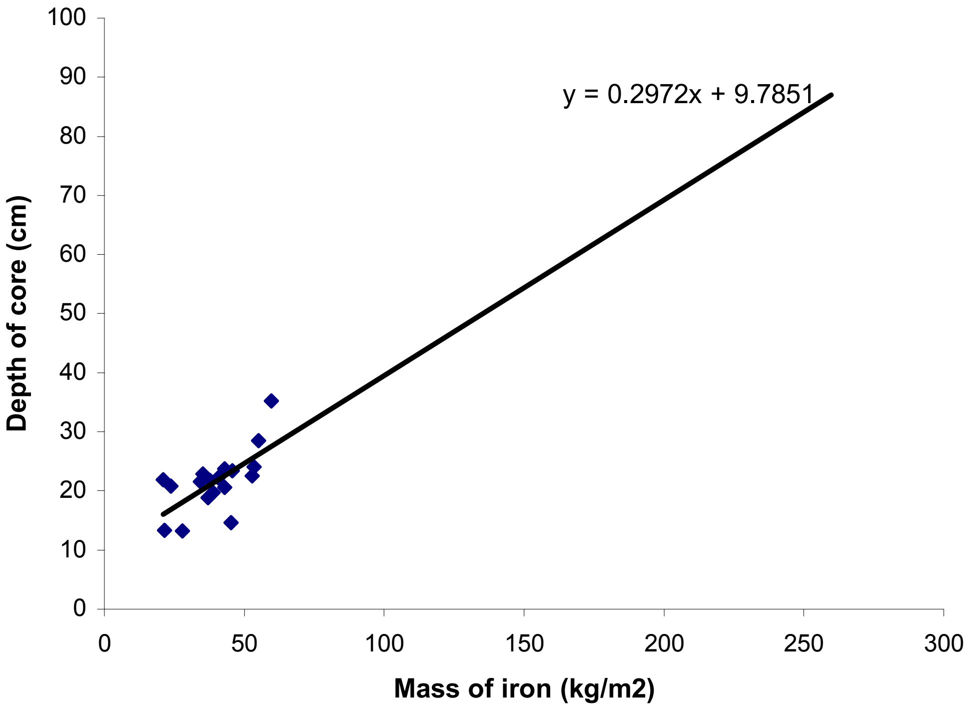


Figure 3.25 Correlation between depth of ochre in the core and mass of iron in the core (Reilly, 2002)

This result calculated by Reilly (2002) makes the assumption that ochre accumulates at the same rate but does not allow for further compaction of the ochre with extra depth of buildup. The timescale does initially appear to differ from that of Dey and Williams (2000). They did stress the preliminary nature of their work but interestingly it is their rate of accretion that appears to be questionable. Based on the rate they had assumed, utilizing the available data at that time, then when Reilly sampled there should have been approximately 45cm of ochre in the system. The lack of this depth of ochre can be attributed to compaction of the ochreous material as neither the iron loading or the iron removal performance of the RAPS have changed over this time. If we use the rate of 21cm in 3.33 years that was observed by Reilly then it would take between 10 and 11 years to reach the ochre depth calculated by Dey and Williams (2000) to exceed the RAPS permeability limit.

Both the methods calculated above do not differ greatly in terms of the depth of ochre that can accumulate before problems are encountered (65 to 84cm) or the likely timescale (10-15 years). Both methods contain assumptions but do appear to provide a good insight into the potential lifespan of the system. If either physical space or permeability were to be the limiting factor then the amount of ochre that can be accumulated by the cell would appear to be fairly similar before problems occur. This would give a date range of 2010-2016 for the system to fill up. It would potentially be possible to remove some or all of this ochre either by a pumped mechanism (of a type suitable for sludge wastes) or by use of a mechanical digger.

Either of the ochre removal methods would only be worthwhile without replacing the rest of the substrate as long as the organic and limestone layers are still performing adequately. The organic material in the compost layer may break down significantly. This may reach the point where there is not enough organic carbon available for any bacterial processes that may be necessary. There is little capacity for organic matter replenishment through natural plant die back as there is not much vegetation within the system. However no assessment of this has been made as part of this project. An assessment of the use and therefore potential lifespan of the limestone bed has been attempted.

Use of limestone and longevity of alkalinity production

A calcium budget was used to assess the use of limestone within the Whitworth A RAPS. The volume of the limestone layer is 912.5m^3. Assuming that 1m^3 of limestone with 30% voids weighs 1 102 kg (Younger, 1996) then the mass of limestone in the bed is 1 005 575 kg $CaCO_3$.

A mean rate of removal of $CaCO_3$ and 95% confidence intervals of 27 720 kg/year (+/- 5 450 kg/year) was calculated from the budget of calcium through the RAPS.

The lifespan was then calculated as 36.3 years (95% confidence interval range 30.2 to 45.2 years). This will be the time taken to use all the limestone. It gives an end date of 2034 and a date range of complete limestone usage of 2028 to 2043. Due to the size of the confidence intervals this gives a large date range for complete limestone usage. It is however satisfying that the lower confidence interval date is still indicating a long lifespan for the limestone.

The alkalinity budget of the limestone bed of the RAPS was calculated. This used the pore water data gathered for this project by Thomas (2001) and the monthly monitoring data from the RAPS exit. The average pH of the water exiting the organic substrate and entering the limestone was 3.5 and thus there was no alkalinity. The acidity of the incoming water was calculated from measured data. This was utilized to create an acidity budget purely for the limestone layer and converted to an amount of alkalinity use within this layer. The total production of alkalinity was then known and converted to an amount of $CaCO_3$ use. The mean yearly rate and 95% confidence intervals for $CaCO_3$ export from the RAPS calculated by this method was 27 945 kg $CaCO_3$/year (+/- 8 165 kg $CaCO_3$/year).

Applied to the initial mass of limestone it gives a lifespan for the entire limestone bed of 36.0 years (confidence interval range of 27.8 to 50.8 years). The end date of limestone use calculated by this method is therefore 2034 with a range of 2025 to 2048. This is obviously a large date range and little significance can be gained from it. Again similarly to the previous calculation it does imply that the limestone should last for a significant timespan.

The two dates calculated for complete limestone use are very close to each other. And the ranges while large do have lower confidence intervals that are well into the predicted 30-year lifespan of the systems. However they do assume that the rates of dissolution remain constant, which is unlikely. At some point before the limestone is completely utilized the rate of limestone dissolution to create alkalinity would diminish. It will reach a point where it will not buffer all the acidity in the minewater. This is the date that is important in terms of treatment lifespan and is currently unknown.

Vegetation development

The Whitworth A RAPS was not originally planted with any vegetation species. There has been a small development of certain species, mainly jointed rush (*Juncus articulatus*), compact rush (*Juncus conglomeratus*) and soft rush *Juncus effusus*. Due presumably to the rate of ochre buildup in the cell a full vegetative cover has not developed. The aerobic cell was created by bunding around an existing wetland area and as such has a diverse variety of plant species from the locality.

3.2.3 Whitworth B and Gwenffrwd

This set of wetlands has never been studied in any detail. For the majority of its operational life to date it has been receiving uncontaminated flow. Following engineering works in March 2001 the Whitworth B and Gwenffrwd minewaters have both been flowing through this system. The loading of iron has been very low and there is not an obvious ochre layer on the RAPS. It is difficult to suggest when any problems may occur with this system but it is likely that it will continue to operate efficiently longer than Whitworth A, the other treatment leg of phase III, due to the relatively low iron loading.

3.2.4 Garth Tonmawr

This wetland system has again received little attention in terms of work on its internal processes and sedimentation. It has been noted that there were problems with downward flow through the RAPS cells within a year of construction. This created deep water on the surface of all the cells, which made access to the sediments for sampling very difficult. Ochre has accumulated primarily across the aerobic cell 1, but a layer (~5cm deep in 2001) has been deposited on the surface of the RAPS cells. It is not clear why a large proportion of the minewater flows over the RAPS surface rather than through it. It could be an effect of differing substrate characteristics or minewater chemistry. These may make the flow down through this RAPS system more difficult than in Whitworth A. The proportion of flow bypassing the RAPS appears to have been fairly constant. To date, given this volume of overflow, there has still been enough alkalinity generated to counteract the acidity present within the minewater. The most likely scenario for failure of this treatment system would be if downward flows through the RAPS were to be completely impeded and no alkalinity production occurred.

The limestone is likely to last longer than anticipated as it has less flow passing through it than anticipated. As the volume of flow passing through these beds is not known it is not possible to calculate a calcium budget as was done for Whitworth A RAPS.

It can be seen that certain species have effectively colonised all the cells in this system, notably, jointed rush (*Juncus articulatus*), compact rush (*Juncus conglomeratus*) and bistort (*Polygonum amphibium)*. From the rapid vegetation growth demonstrated in the Garth Tonmawr system it can be seen that it is perhaps not always necessary to plant the wetlands as rapid colonisation by local species can occur. However planted vegetation will immediately provide benefits by slowing flow and facilitating deposition of suspended pollutants, as well as being a source of organic matter. Local species can then move into the wetland as demonstrated within the Whitworth No 1 system. The decision whether to plant a system or not is therefore probably best made on a site specific basis based on proximity to source vegetation, the need for flow reduction/organic inputs and cost.

3.3 Environmental Benefits

3.3.1 Water quality improvements in the rivers

The changes in water quality following minewater treatment were assessed at a number of points up and downstream of the treatment systems. The results will be discussed in three sections, the two main tributaries (Nant Gwenffrwd and Blaenpelenna) and the Pelenna from the confluence down.

Nant Gwenffrwd

The first treatment scheme to be constructed on the Nant Gwenffrwd was Whitworth No 1 in 1995. The water quality data from the downstream sample point (G6) does not show any changes in dissolved iron or pH data following treatment (Figure 3.26). The EAW trend identification software (TAPIR) was used. This identifies linear, step and seasonal changes in long term data sets. The lack of any statistically identifiable change in water quality following treatment is due to the combined loading of the three much more contaminating discharges upstream. These other minewaters were treated in phase III of the project in 1998. It can be seen that the software identifies a step change in the dissolved iron concentration at the date where the Phase III (Whitworth A, B and Gwenffrwd) wetlands were constructed (Figure 3.26).

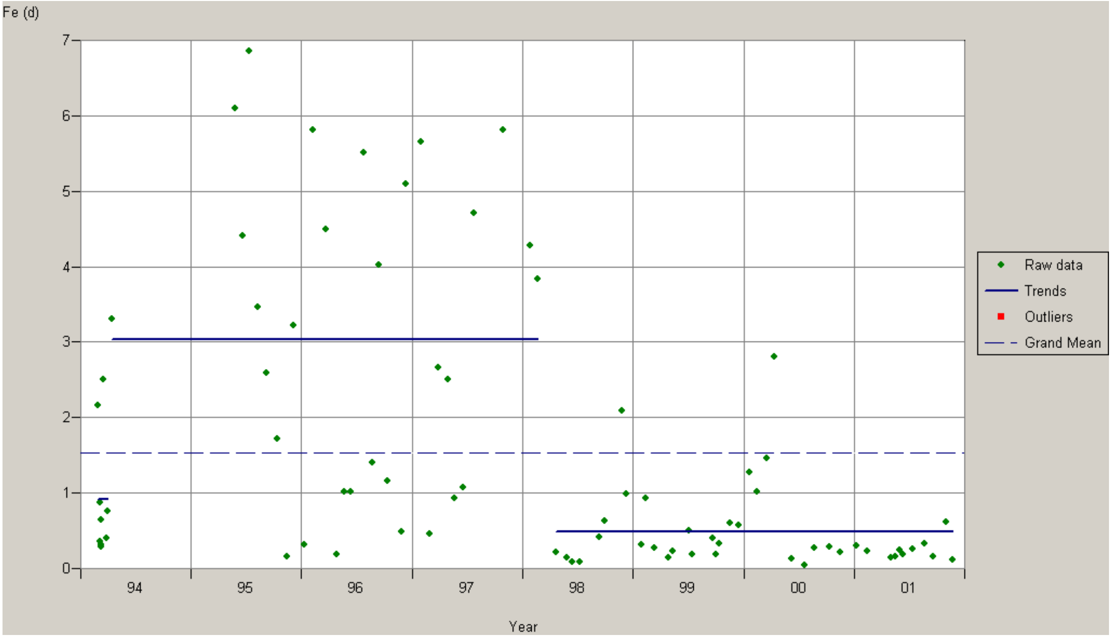


Figure 3.26 TAPIR analysis results for dissolved iron at site G6.

R&D Project Record P2-181/PR Appendix 4 contains the summary data for each sample point pre and post treatment of the minewaters. Photographs of the sample sites on the Nant Gwenffrwd are also included in Appendix 4. They show how the river looked in 1996, 2001 and 2002. Table 3.9 contains an extract of this data with the mean dissolved iron and pH values for each site on the Pelenna pre and post treatment. Figures 3.27 and 3.28 show the dissolved iron and pH results respectively for each sample point on the Gwenffrwd pre and post treatment. Figure 3.27 for dissolved iron demonstrates that there has been no change in the upstream iron concentration over time (site G3). For all the sites impacted by minewaters there has been a major decrease in the dissolved iron concentrations within the watercourse. Importantly the mean dissolved iron concentration is below the EQS level of 1 mg/l at all sample points within the river. However from Figure 3.27 it can be seen that there are a number of results above this level following treatment. These will be discussed further in this section.

Table 3.9 Mean dissolved iron and pH in the Gwenffrwd pre and post treatment.

Site		Mean dissolved iron	Mean pH
G3	Pre Treatment	0.09	6.08
	Post Treatment	0.21	6.45
G4	Pre Treatment	5.35	5.61
	Post Treatment	0.89	6.35
G5	Pre Treatment	4.57	5.52
	Post Treatment	0.75	6.69
G6	Pre Treatment	2.51	5.65
	Post Treatment	0.47	6.97

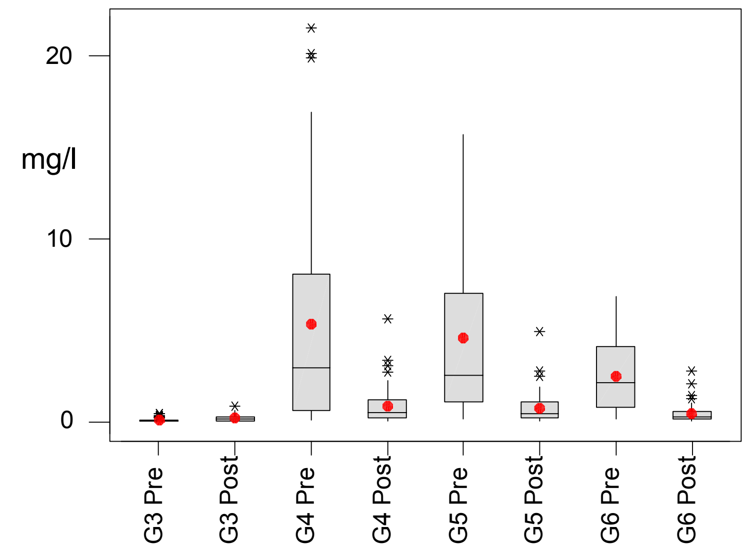


Figure 3.27 Boxplot of the dissolved iron concentration in the Gwenffrwd at all sample points pre and post treatment.

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).

Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

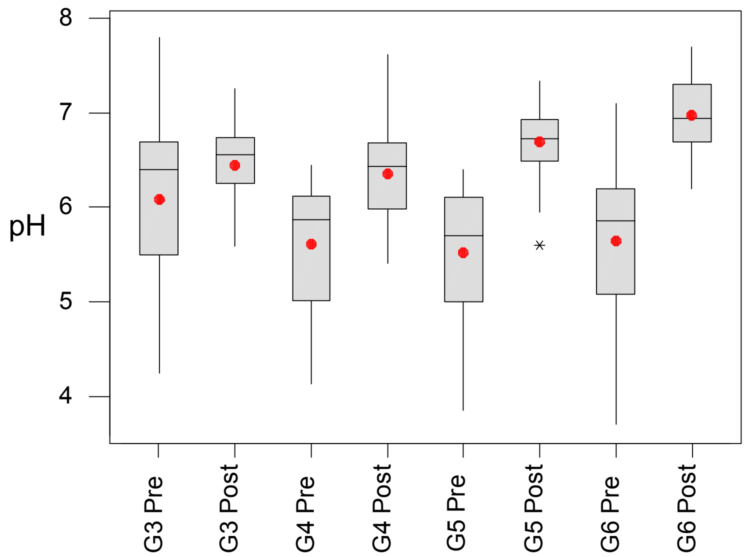


Figure 3.28 Boxplot of the pH in the Gwenffrwd at all sample points pre and post treatment.

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit:Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).

Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

Looking at the mean pH results in Table 3.9 and the distribution of data in Figure 3.28 the changes in pH following treatment can be seen. At the upstream sample point (G3) the pH appears to have increased with a less acidic mean value and a smaller spread of results. Figure 3.29 demonstrates the TAPIR analysis output for this site and shows a reduction in the minimum pH values observed over time and a linear trend of increasing pH. The likely reason for this is that there have been significant reductions in local source emissions of acid rain producing contaminants over the last few years. This has occurred due to the cleaning of major local industrial process emissions and the closure of other local industries (M Broom, PIR/RSR Officer, Personal Communication). A step change (P<0.05) was identified in 1997. The reason for this date being identified is unknown. These changes in pH are still not as great as those seen at the sites below the minewaters. At all these points the pH has risen significantly with greater increases seen at the lower sites G5 and G6.

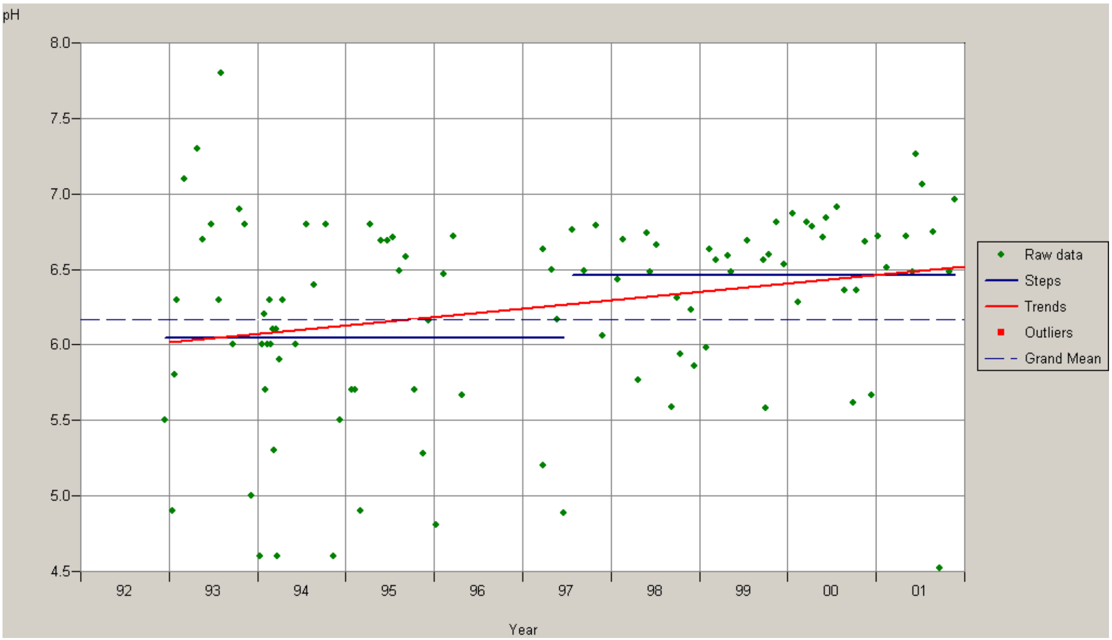


Figure 3.29 TAPIR analysis output for pH at site G3 on the Nant Gwenffrwd

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).
Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

Figure 3.30 demonstrates the changes in pH and dissolved iron at site G4 over time. This point is immediately downstream of Whitworth Lagoon, the point at which Whitworth A and B minewaters originally discharged to the Gwenffrwd. The treated effluents from the Phase III wetlands now discharge downstream of this point. However this location does have the potential to receive contaminating discharges if the minewaters overflow their inlet structures.

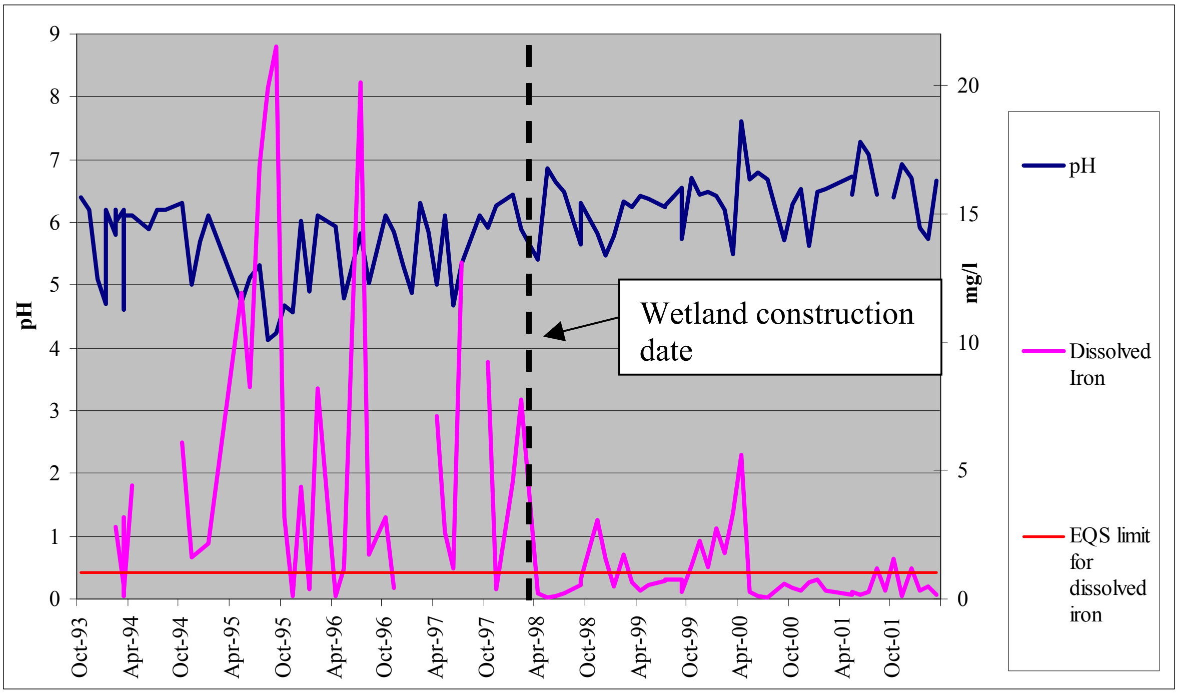


Figure 3.30 Dissolved iron concentration and pH changes over time at site G4.

From the graph it can be seen that the pH has been rising slightly and it was higher following the minewater treatment system coming on line. The TAPIR output for this site (Figure 3.31) shows increasing pH over time, again with an unexplained step change in 1997. The dissolved iron levels were very variable and frequently well above the EQS level of 1 mg/l before minewater treatment. Following treatment the levels dropped quickly to be below 1mg/l but there have been occasions when higher levels have again been observed. There were two main peaks in iron concentration above 1 mg/l observed following the treatment systems coming on line. The first of these is potentially as a result of the underground diversion of the Gwenffrwd minewater, which caused it to discharge from Whitworth Lagoon into the Nant Gwenffrwd and bypass the treatment system.

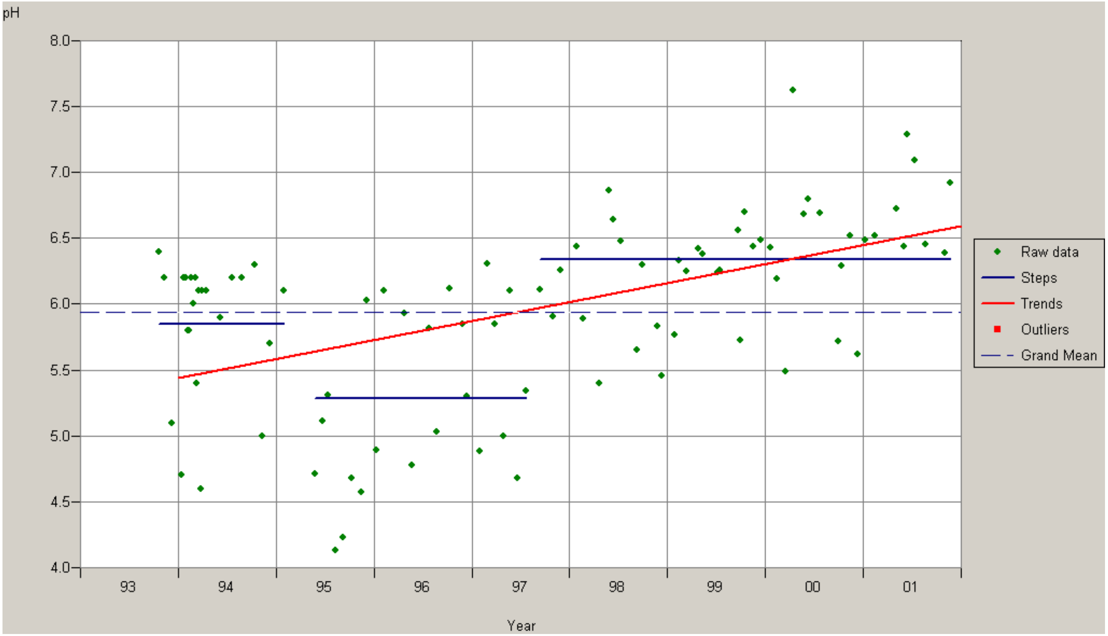


Figure 3.31 TAPIR output for pH at site G4 on the Nant Gwenffrwd

The second was due to the blockage of the inlet structure and inlet pipes that took Whitworth A minewater into the treatment system. This started to block over the winter of 1999 and was completely blocked in the spring of 2000. At this point all the minewater discharged virtually untreated into the Gwenffrwd through the Whitworth Lagoon. A corresponding drop in pH can also be seen from the data at this time indicating the increased acidity loading from the raw minewater. Once this blockage had been cleared the dissolved iron level quickly returned to below 1 mg/l. A series of small peaks in the iron data exists at the end of 2001 through to the start of 2002. At this point the inlet structures and pipework were starting to block again. This was causing a proportion of the minewater to continuously overflow the system without receiving treatment. The implications of these overflows on the river ecology and its recovery following treatment are discussed in sections 3.3.2, 3.3.3 and 3.3.4.

Figure 3.32 demonstrates a similar graph of the dissolved iron and pH results from the next sample point down (G5). This is downstream of the discharges from the Phase III (Whitworth A, B and Gwenffrwd) wetlands but upstream of the Whitworth No 1 discharge. Looking initially at the dissolved iron concentration the picture is essentially the same as at the point upstream. This indicates the effectiveness of the treatment systems when they are receiving the minewaters. They treat the iron in the discharges to a level that has no noticeable effect on the concentration in the watercourse. This concentration is well below the EQS limit. However the same peaks in iron that are at the upstream site can be seen at the point. At site G6, 1km downstream of the minewater input, the impact of these overflows was still evident as peaks over 1 mg/l. The peaks were visible still in data from the site P2 at Efail Fach. This site is 2.8km downstream on the Pelenna but here the levels only just exceeded the EQS on one occasion following minewater treatment.

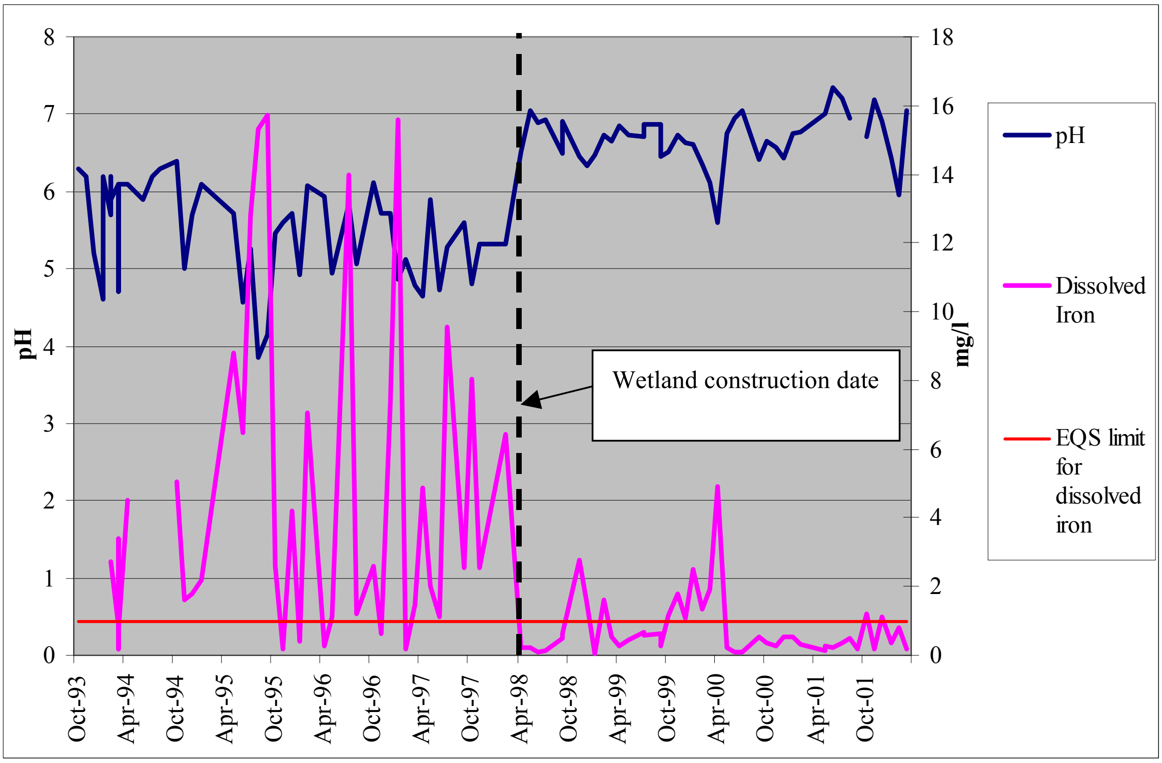


Figure 3.32 Dissolved iron concentration and pH changes over time at site G5.

The pH change following treatment at G5 is much greater than that observed at the upstream point (G4). It is believed that this is due to the discharges from the Phase III wetlands. These contain a substantial amount of alkalinity generated by the two RAPS units and this appears to be having a substantial buffering capacity in the watercourse thus raising the pH. This effect is also evident in the boxplot in Figure 3.28 where the pH following minewater treatment is higher further downstream.

Figures 3.33 and 3.34 contain a two month set of continuous data gathered from sites G3 and G6 respectively. This was gathered in January and February 2002 using YSI 6920 remote sondes logging pH, conductivity and temperature every 15 minutes. The determinand of primary interest is pH as this can be seen to vary considerably in relation to rain storm events. The pH can be seen to drop quickly repeatedly in Figures 3.33 and 3.34 as a response to rainfall. The increased runoff causes an acidic flush in the river in part by mobilising acidic airborne pollutants deposited on the conifer forest plantations. The water quality then recovers slowly over time to return to a much less acidic base level. In Figure 3.19 the pH upstream of the minewaters can be seen to drop to a low of 4.09, with a mean of 5.59 and a max of 6.46. Figure 3.20 downstream of the RAPS does not drop as low, min 4.66 and a mean of 5.88 with a max of 6.66 over this period. The RAPS discharge does seem to offer some buffering to the most acidic peaks which are more damaging to the ecology and it also appears to reduce the range of variation within the pH's observed.

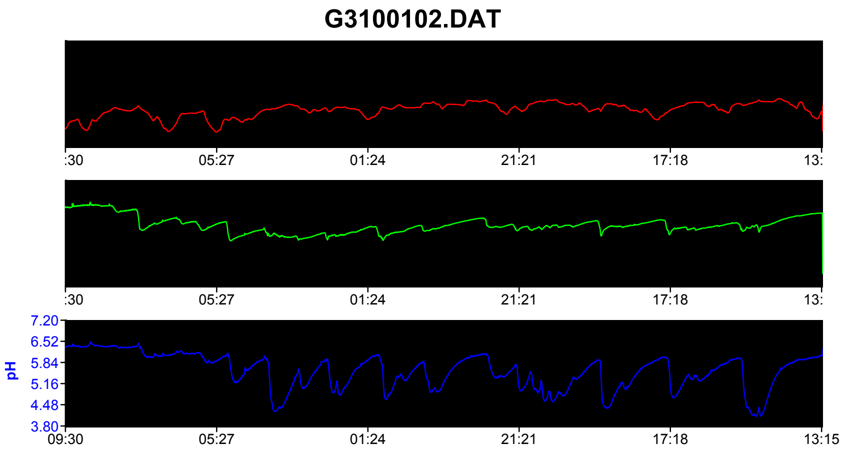


Figure 3.33 Continuous monitoring dataset from site G3 (upstream of RAPS discharges)

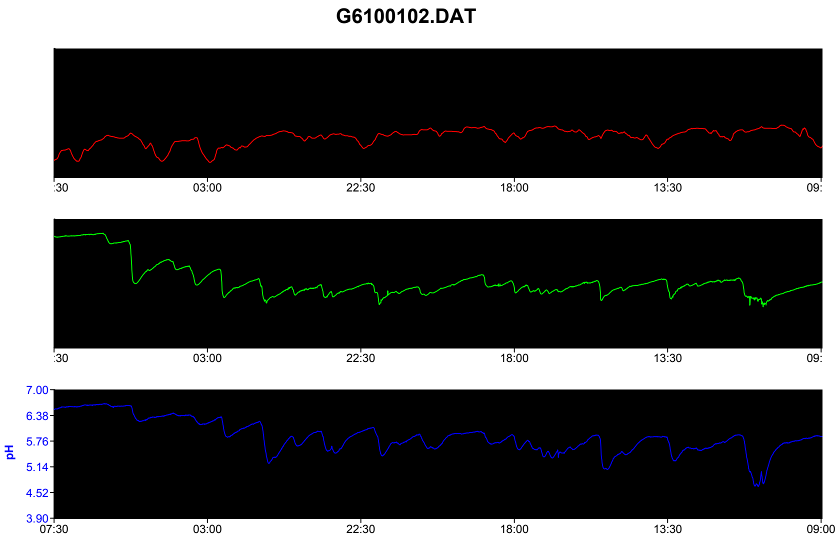


Figure 3.34 Continuous monitoring dataset from site G6 (downstream of RAPS discharges)

Blaenpelenna

The data from monitoring sites on the Blaenpelenna were inputted into the TAPIR software to identify any trends in iron and pH concentrations over time. An example of one of the outputs for the site below the treatment system (BP5) is shown in Figure 3.35. The software identified a step change in the iron concentration at the end of 1998. This corresponds to a date just before the treatment system was commissioned. Variability in the iron concentrations observed before this date can be attributed to the slight discrepancy in dates identified. Table 3.10 contains the mean dissolved iron and pH values from all the sites on the Blaenpelenna. A full set of summary data for all these points is included in R&D Project Record P2-181/PR Appendix 4 along with photographs before and after treatment at each sample point. From Table 3.10 it can be seen that the minewater discharge had an effect on the Blaenpelenna at site BP5 approximately 500m downstream of its discharge but by site P1, 2.5km downstream, there is little effect on the dissolved iron, but still an effect on pH. This can also be seen in the boxplots of dissolved iron and pH at all sample points on the Blaenpelenna before and after treatment (Figures 3.36 and 3.37).

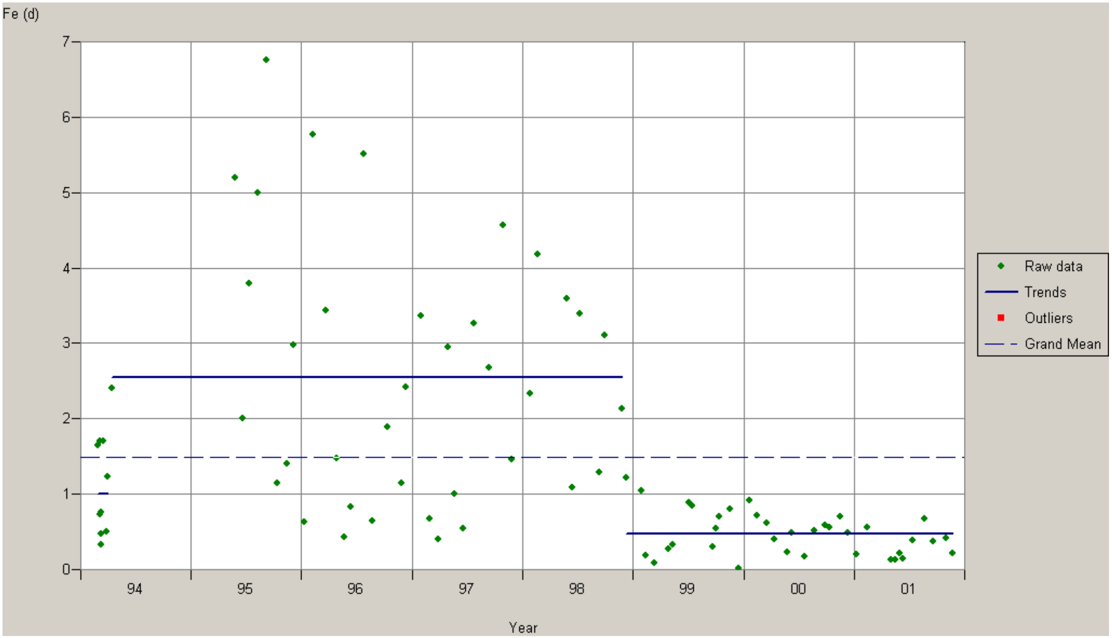


Figure 3.35 TAPIR output for dissolved iron at site BP5

Table 3.10 Mean dissolved iron and pH in the Blaenpelenna pre and post treatment.

Site		Mean dissolved iron	Mean pH
BP3	Pre Treatment	0.21	6.05
	Post Treatment	0.30	6.17
BP4	Pre Treatment	0.21	6.36
	Post Treatment	0.30	6.58
BP5	Pre Treatment	2.11	6.09
	Post Treatment	0.45	6.63
P1	Pre Treatment	0.30	6.48
	Post Treatment	0.16	6.95

Figure 3.36 demonstrates the effectiveness of the minewater treatment as it has reduced the dissolved iron in the watercourse downstream (site BP5) to a level that is very similar to the rest of the river. The pH has also risen following treatment at this site (Figure 3.37) as it has downstream also. There are two upstream points sampled on this river and whilst iron has not changed much over time pH appears to have risen slightly. This could again be attributed to the changes in airborne acid contaminant loading to the catchment as was suggested may be occurring on the Nant Gwenffrwd. In between the two upstream sample points is another minewater discharge called Middle Mine. It discharges down a watercourse and by the time this enters the Blaenpelenna it has a pH of 6.7 with an iron content of 0.94mg/l total iron. This is not having an effect on the downstream sample point BP4.

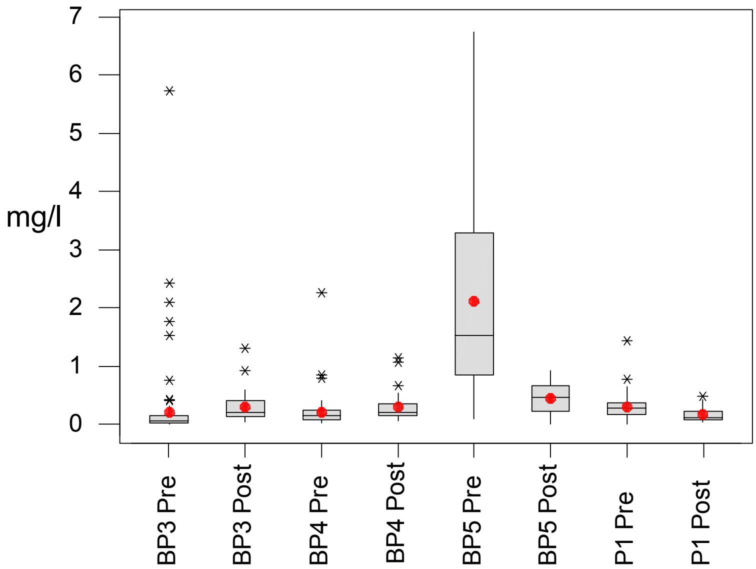


Figure 3.36 Boxplot of the dissolved iron in the Blaenpelenna at all sample points pre and post treatment.

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).
Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

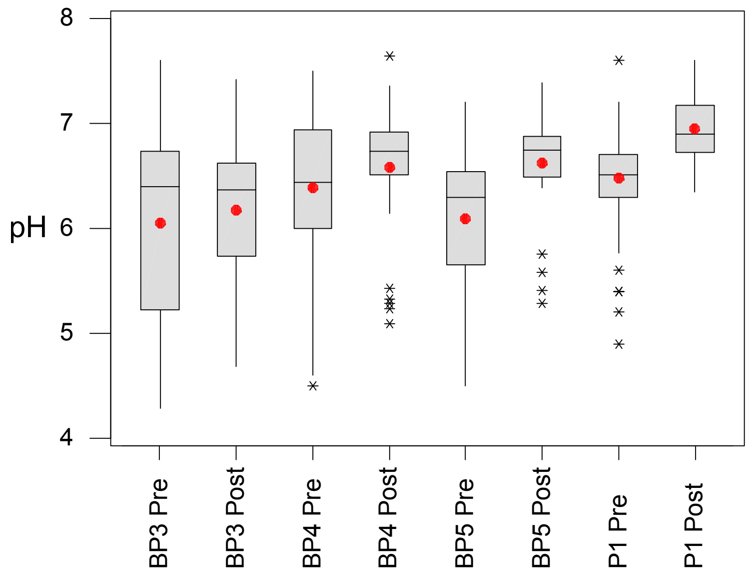


Figure 3.37 Boxplot of pH in the Blaenpelenna at all sample points pre and post treatment

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).
Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

Figure 3.38 contains a graph of the pH and dissolved iron at site BP5 downstream of Garth Tonmawr treatment system. It is also downstream of any overflows should the minewater flow be too great for the treatment system. There has been an increase in pH following treatment from a mean of 6.06 to 6.63. More significantly is a large decrease in the concentration of dissolved iron. This was previously fairly high and variable and since treatment has always been maintained below the EQS of 1 mg/l.

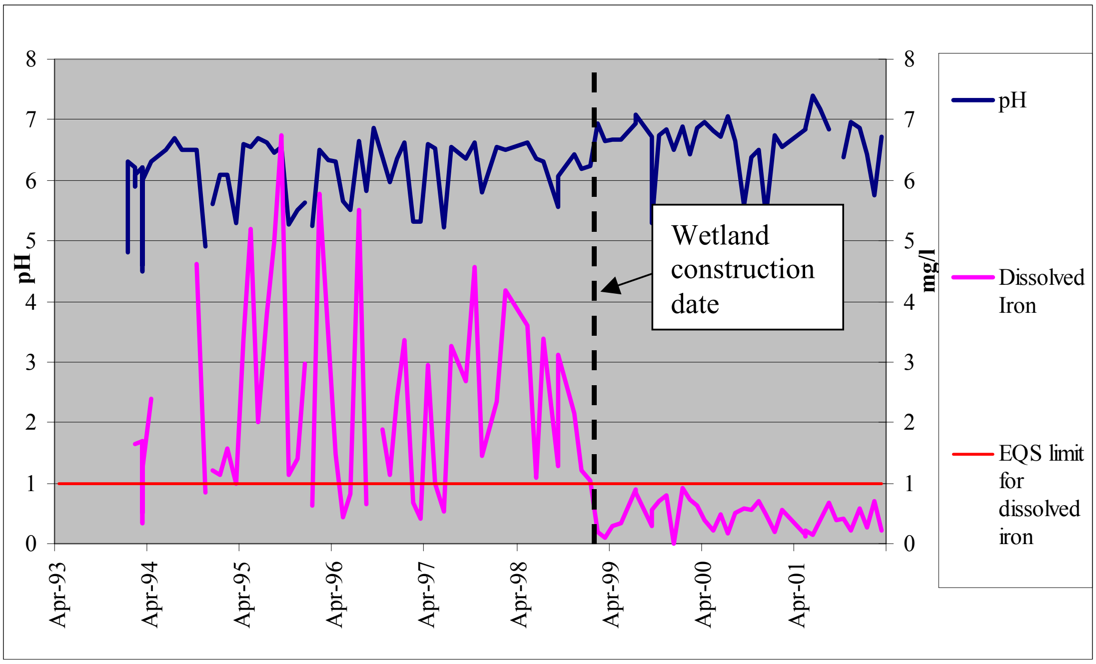


Figure 3.38 Dissolved iron concentration and pH changes over time at site BP5.

Pelenna

Figures 3.39 and 3.40 demonstrate the pH and dissolved iron changes over time on the Pelenna. Figure 3.39 is site P2 at Efail Fach, 2.8km downstream of the Phase III treatment system. Figure 3.40 is site P3 at Pontrhydyfen, 5km downstream of the Phase III treatment system. At site P2 the dissolved iron concentration occasionally went above 1mg/l but there was still a significant visible orange staining of the watercourse. R&D Project Record P2-181/PR Appendix 4 contains photographs of these points before and after minewater treatment. It would appear based on the results from the Blaenpelenna sites, especially P1 that the majority of this iron was coming from the Gwenffrwd. Therefore the iron was from Whitworth A, B and the Gwenffrwd minewaters. Following March 1998, the date of treatment of these minewaters, the dissolved iron has been below the EQS except for one of the peaks identified previously in the Gwenffrwd data in Figures 3.30 and 3.32. However both the trends in iron and pH are difficult to identify due to the break in data when this point was not sampled.

At site P3 (Figure 3.40) the dissolved iron has never been above 1 mg/l. The pH at this point has risen slightly with a mean pre Phase III construction value of 6.77 and a mean post construction value of 7.09. However it is difficult to attribute this solely to minewater treatment at this distance from the discharges due to the observed improvements in the pH of the water in the catchment upstream.

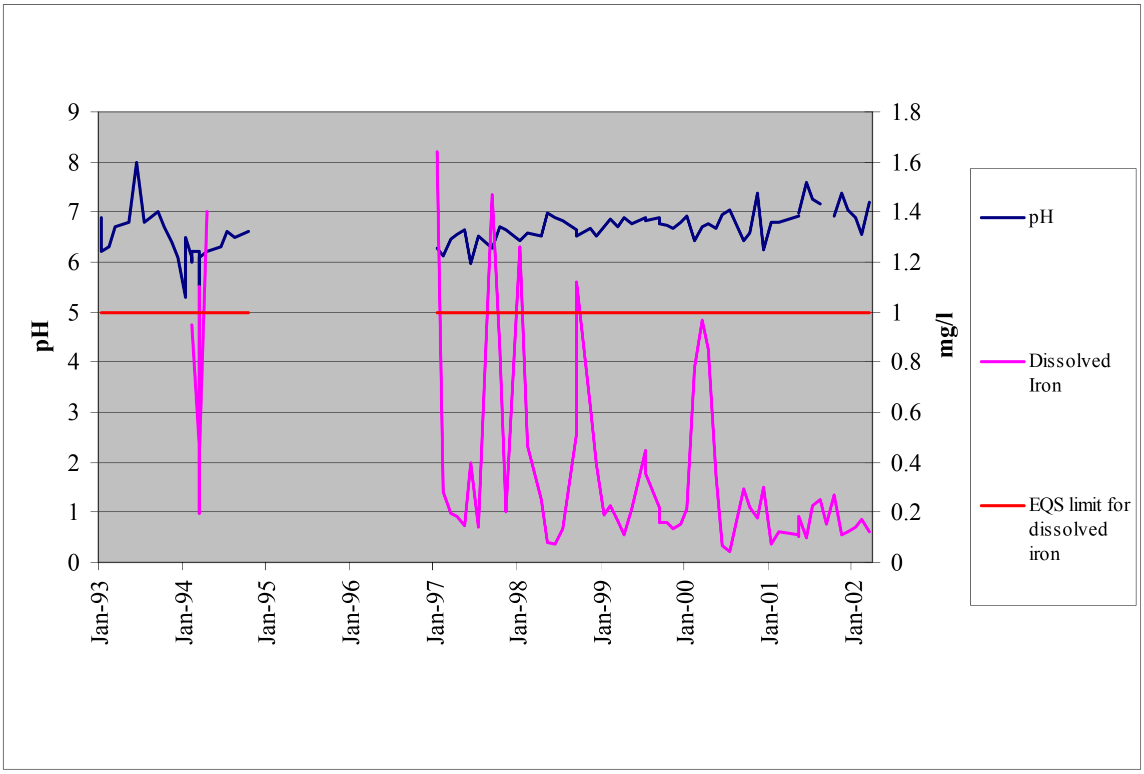


Figure 3.39 Dissolved iron and pH at site P2 on the Pelenna (at Efail Fach)

Note: Site P2 was not sampled from late 1994 till early 1997.

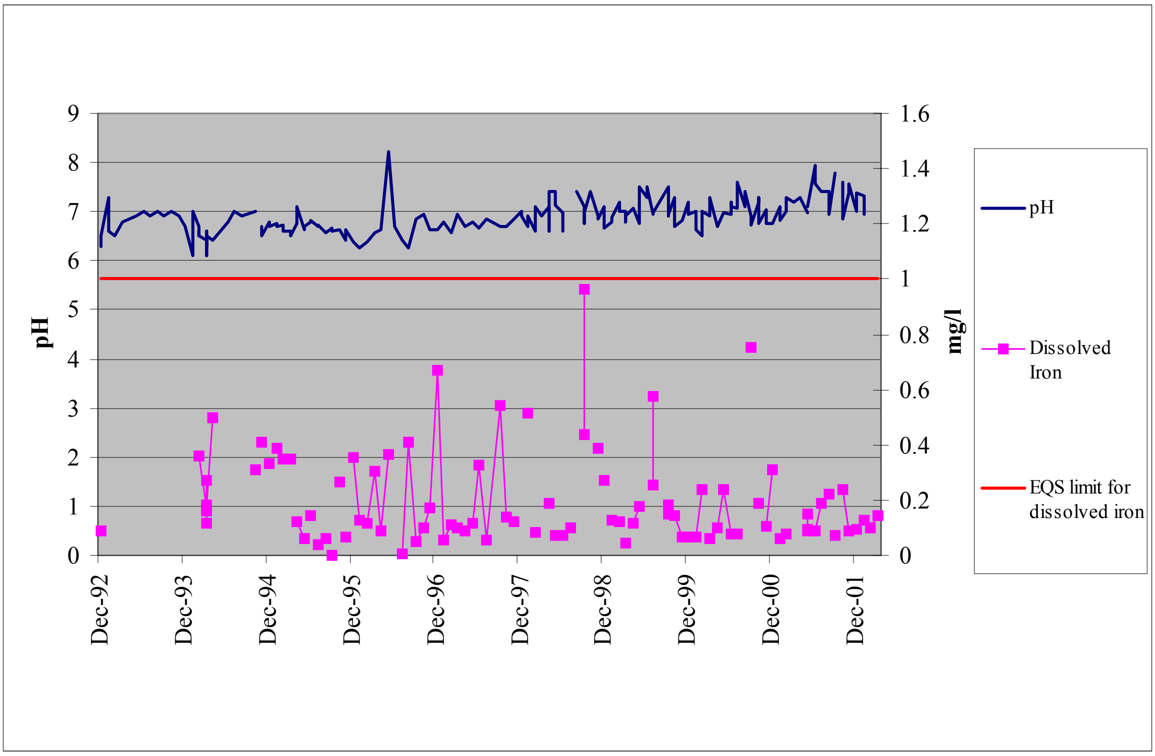


Figure 3.40 Dissolved iron and pH at site P3 on the Pelenna (at Pontrhydyfen)

3.3.2 Visual impact assessment key

The Visual Impact aSsessment key (VIS) was used at the Pelenna and a number of other minewater sample sites across South Wales. Its aim was to develop a key that identified changes in the visual impacts from mines in the watercourses. On the Pelenna it was utilized to assess changes over time following treatment. Figure 3.41 contains a map of the changes in percentage distribution of the VIS results each year for each sample point on the Pelenna. From this it can be seen that the upstream sites and the Fforch Dwm were almost always clear of any visual staining. On the Gwenffrwd below the minewaters the results are a bit more complex. The visual impact is reduced following the water quality improvements caused by the Phase III treatment coming on line. However there have still been a number of higher results seen especially in 2000. This is another indication of the overflows that impacted the Gwenffrwd especially in 2000. Figures 3.42 and 3.43 demonstrate the visual changes at site G4 from 1995 to 2001 that were achieved by stopping the minewater from entering the river upstream of here. Figures 3.44 demonstrates the visual impact of the 2002 overflows on this stretch of river.

The Blaenpelenna demonstrates little change over time at site BP5 below the minewaters until 2001. The recovery demonstrated at this and the other points can be seen in the photographs of before and after treatment in R&D Project Record P2-181/PR Appendix 4.Further down just above the confluence with the Pelenna the VIS key demonstrated recovery following treatment. However this site again appeared to be much worse in 2000. On the Pelenna downstream of the confluence at Efail Fach recovery following minewater treatment can again be seen. The impact of the 2000 overflows is quite obvious, as it is downstream at P3 also. The distinct improvements in terms of a reduction in ochre staining at Efail Fach (P2) can be seen in the 1995 pre-treatment photograph in Figure 3.45 and the 2001 post-treatment photograph in Figure 3.46.

Figure 3.47 contains the range of dissolved iron levels observed for each VIS score for all the sample points on the Pelenna. It does give an indication of the level of staining that may be expected with a given value or range of iron content. There is however a large overlap between the ranges identified. Figure 3.48 demonstrates the mean dissolved iron level observed at each sample point in South Wales where this key has been used. This has been plotted against the most frequently observed (mode) VIS score for that site. This gives a better representation of the range of iron concentrations that may be expected to lead to certain levels of staining. It allows for a more long term view of the effects of the dissolved iron rather than the previous figure which is concerned purely with the actual results from spot samples.

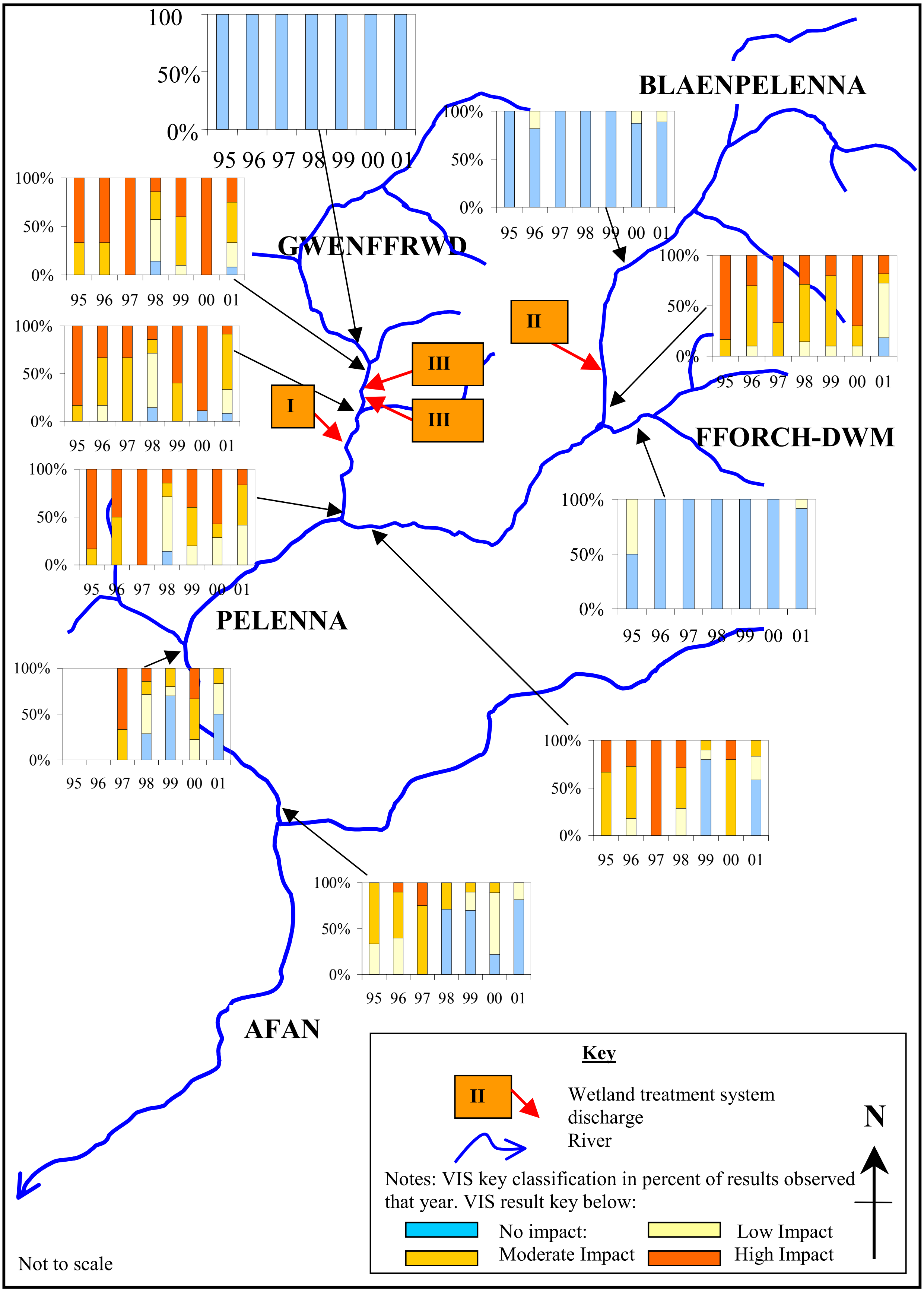


Figure 3.41 Percentage distribution of visual impact assessment results on the Pelenna

Figure 3.42 Visual impact at site G4 on the Nant Gwenffrwd in 1995.

Figure 3.43 Visual impact at site G4 on the Nant Gwenffrwd in 2001.

Figure 3.44 Visual impact at site G4 on the Nant Gwenffrwd in 2002.

Figure 3.45 Visual impact at site P2 on the Pelenna in 1995.

Figure 3.46 Visual impact at site P2 on the Pelenna in 2001.

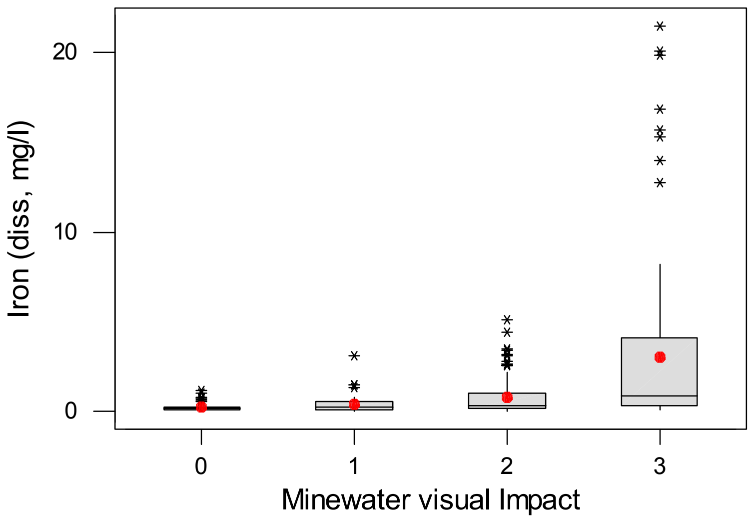


Figure 3.47 Range of dissolved iron levels observed for each visual impact score on the Pelenna and tributaries.

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).

Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

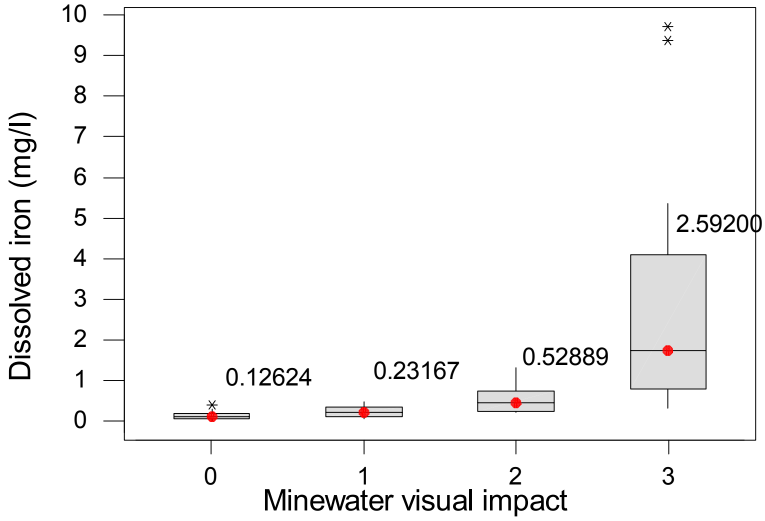


Figure 3.48 Range of mean dissolved iron content observed for the mode (most frequent) visual impact score at each available minewater sample point in South Wales.

Note: Boxes indicate interquartile range with the median value as an intersect line. The mean is indicated by a red dot and the associated number. The whiskers extend from the top and bottom of the box to the adjacent values. The adjacent values are the lowest and highest observations that are still inside the region defined by the following limits: Lower Limit: Q1 - 1.5 (Q3 - Q1); Upper Limit: Q3 + 1.5 (Q3 - Q1).
Outliers are points outside of the lower and upper limits and are plotted with asterisks (*).

3.3.2 Macroinvertebrate populations

The results of the monitoring of the macroinvertebrate populations are presented for the spring and then the summer data for each section of the river system. The two periods of monitoring are presented separately due to the differing populations encountered. The environmental conditions and stresses existing in the catchment also frequently varied throughout the seasons. A range of analyses and biotic indices were utilised on the dataset, as listed in the methods section 2.3.2. The results are first presented and discussed in terms of changes in abundance and then issues of diversity are covered.

Afan and Fforch-Dwm spring monitoring

There are two control areas assessed in the catchment. The first is the Fforch-Dwm in the headwaters. This acts as a control to the smaller sections of the Nant Gwenffrwd and Blaenpelenna. Under optimal conditions these sites should contain a similar population to it. There are two sites on the Afan that can act as controls. These are more similar in nature to the two sites on the main River Pelenna. They should provide a guide as to what the populations at these points could compromise.

The changes in total macroinvertebrate abundance are represented in Figure 3.49. The sites on the Afan and Fforch-Dwm have been variable over time with no obvious trends. Figure 3.50 contains the combined Ephemeroptera, Plecoptera and Trichoptera (EPT) abundances at these three sites. The changes in EPT abundance are similar to the variations in total abundance. The EPT populations at the two Afan sites are predominantly composed of

Ephemeroptera (mayflies). At the upstream control site on the Fforch-Dwm the population is a more even mix of Ephemeroptera and Plecoptera with a small number of Trichoptera.

TWINSPAN analysis was undertaken to investigate these changes in population diversity over time. The dataset for all sites from 1993 to 2001 was classified into three groups that were essentially similar in taxonomic composition. These groupings represent the range of populations found from good quality larger sites on the Afan to the small minewater and acidification impacted streams. Table 3.11 shows how the mean BMWP, number of Taxa and total abundance at sites varies between the TWINSPAN groups. Group 1 are the better quality group and the biological quality, diversity and abundance decrease through the group numbers. Figures 3.51 and 3.52 respectively represent the range of BMWP scores and $\log_{10}$ total abundance's observed for the three TWINSPAN groups. The BMWP classification does vary with each grouping but the group 1 and 2 BMWP scores do overlap markedly. This indicates that the BMWP scoring system is better for differentiating the lower group characteristics. The total abundance variations between the three groups demonstrate that each grouping has a fairly distinct range of abundances.

The TWINSPAN results for each year in spring are plotted graphically in Figure 3.53. It can be seen that the sites on the Afan have been almost exclusively in group 1. This is the population that the sites lower on the Pelenna may be expected to approach. The Fforch-Dwm has had group 2 membership over time and this is indicative of the population structure the sites higher in the catchment may approach, following successful remediation.

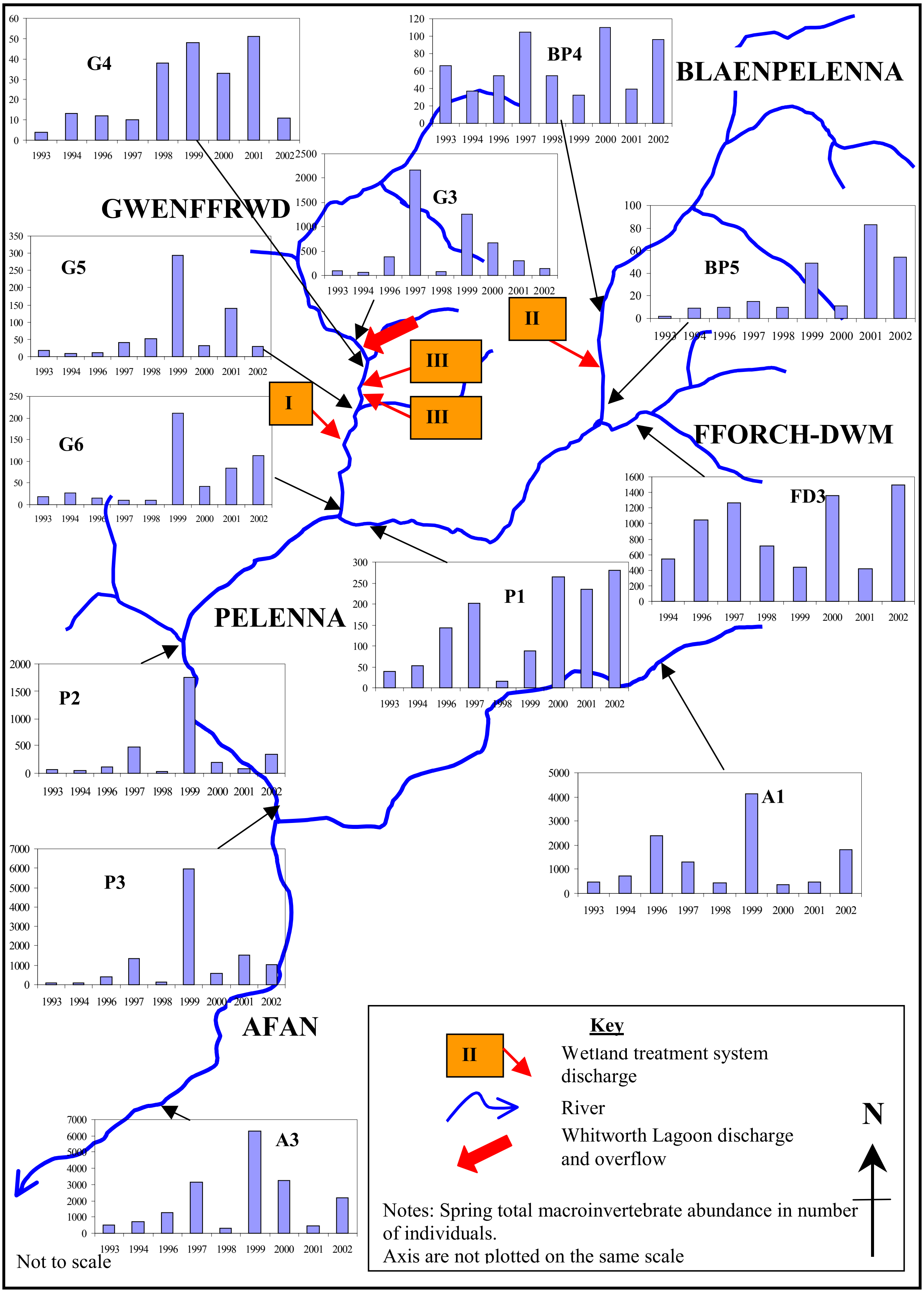


Figure 3.49 Spring total macroinvertebrate abundance

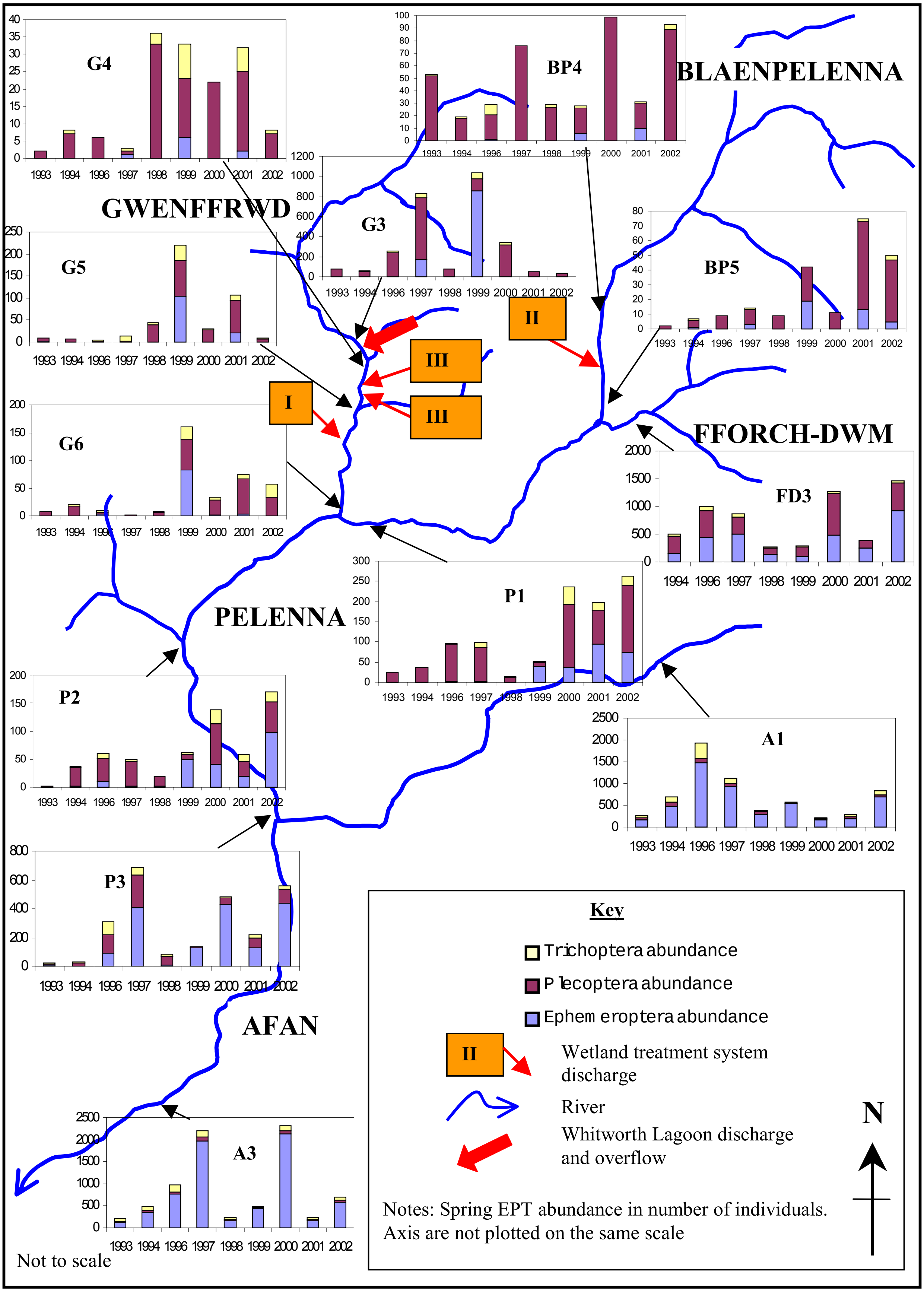


Figure 3.50 Spring Ephemeroptera, Plecoptera and Trichoptera (EPT) abundance

Table 3.11 Biotic indices data for the spring TWINSPAN classification groups

	TWINSPAN group	BMWP[1]	Number of taxa[2]	Total abundance[3]
Spring	1	112.6	17.7	1499
	2	94.9	14,2	541
	3	48.5	7.7	251

Notes: [1]Mean Biological Monitoring Working Party (BMWP) score
[2]Mean number of taxa
[3]Mean Total abundance

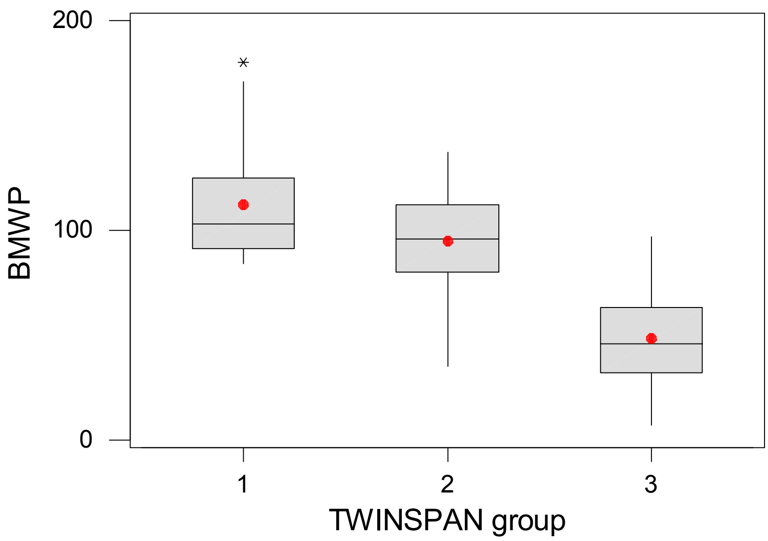


Figure 3.51 Variations in the BMWP score of the three spring TWINSPAN groups

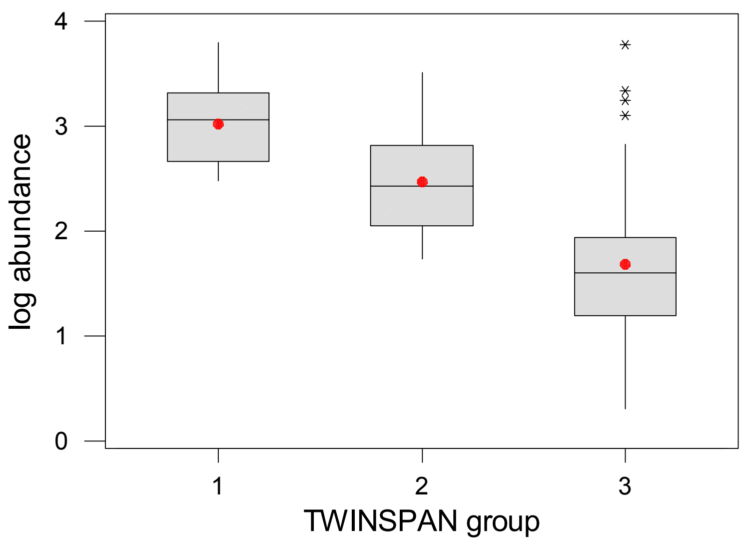


Figure 3.52 Variations in $\log_{10}$ total abundance between the three spring TWINSPAN groups

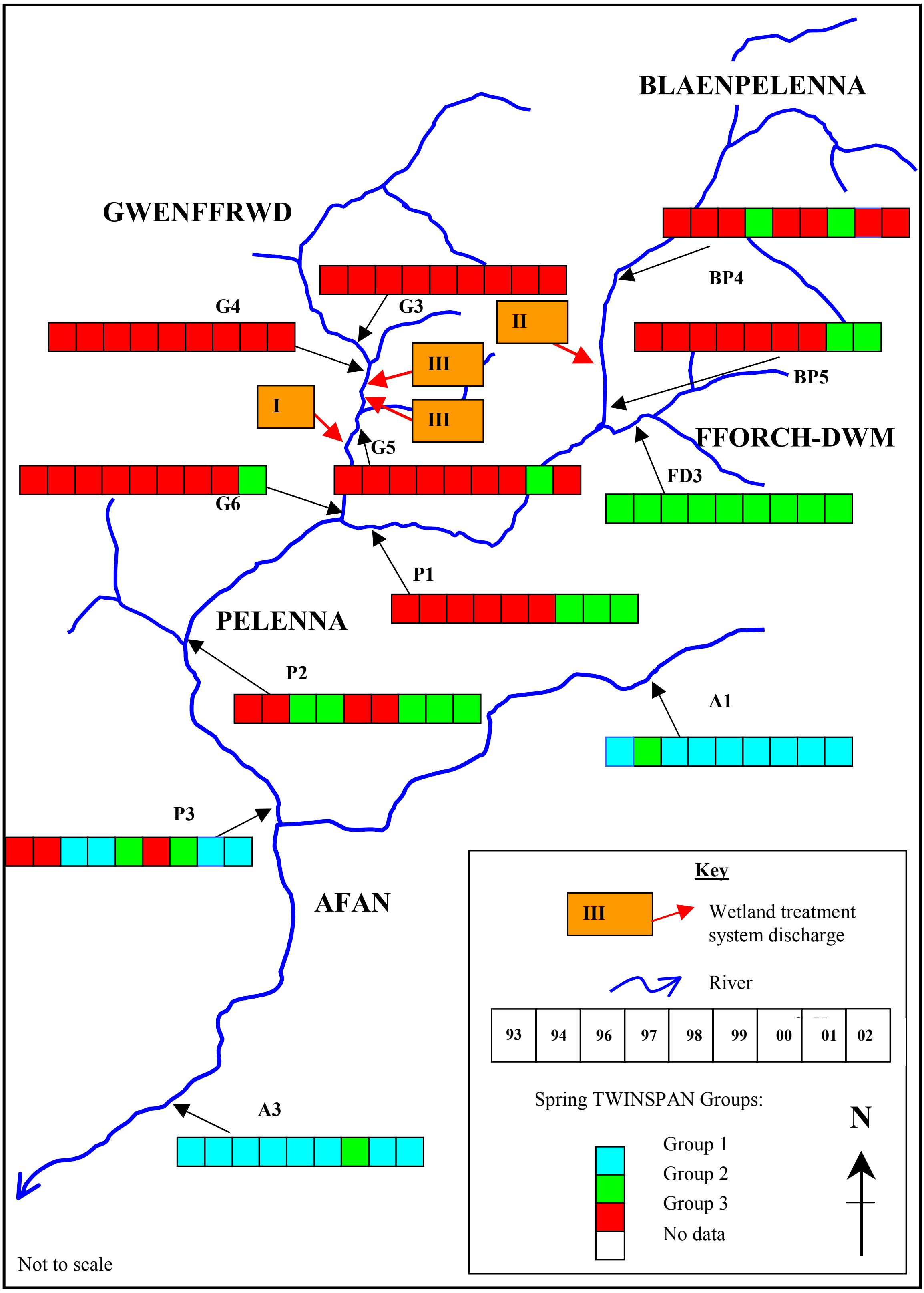


Figure 3.53 Spring TWINSPAN results

Nant Gwenffrwd spring monitoring

The total abundance of macroinvertebrates on the Nant Gwenffrwd is shown in Figure 3.49. Site G3 upstream of the minewaters has varied over time and there are no clear trends in population change. The sites further down (G4, G5 and G6) all demonstrate an improvement in the population abundance in 1999 immediately after minewater treatment. It can be seen that the abundance decreases in 2000 and 2002 at sites G4 and G5 and just in 2000 at site G6. This can be attributed to the impacts of the Whitworth A overflows that were occurring in the spring of these two years.

Figure 3.50 contains the EPT abundance's and presents a similar picture of overall EPT population changes over time. It can be seen that the Ephemeroptera, which are noticeably intolerant of minewater, are especially prevalent in 1999 following minewater treatment. They are then completely removed from the populations of sites G4, G5 and G6 in 2000 and 2002. They did recover again in 2001 once the 2000 overflows had been rectified only to be lost again the next year. Investigating this further Figure 3.54 contains the total abundance at G4 and the mean dissolved iron concentration over the previous 6 months. It can be seen that as mean dissolved iron increases in 2000 and 2002 the total abundance falls. Figure 3.55 contains a similar graph for the Ephemeroptera abundance. When the iron concentration falls following minewater treatment in 1999 more Ephemeroptera exist. Then with increased iron in 2000 the population is removed again. The following year with reduced iron they return again only to be removed by the impacts of the 2002 overflows.

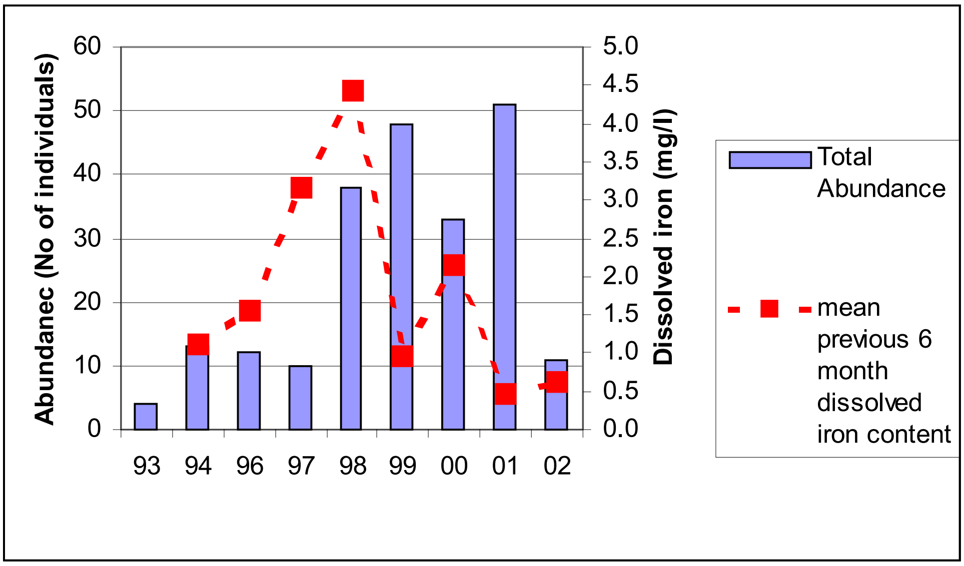


Figure 3.54 Variations in spring total macroinvertebrate abundance and mean dissolved iron concentration in the previous 6 months at site G4 on the Nant Gwenffrwd.

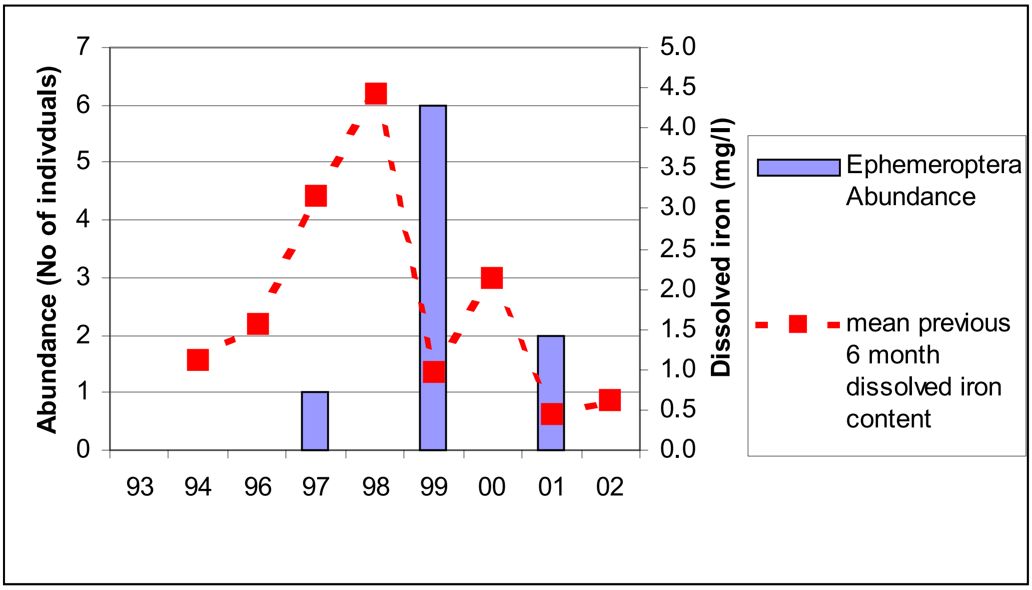


Figure 3.55 Variations in spring Ephemeroptera abundance and mean dissolved iron concentration in the previous 6 months at site G4 on the Nant Gwenffrwd

The spring TWINSPAN results for these sites are shown in Figure 3.51. The upstream site (G3) is in group 3 and has not varied over time. At the three sites downstream the pre treatment conditions were exclusively group 3 impoverished populations. Only twice have populations approached the group 2 levels to be similar to the Fforch-Dwm. This was in 2002 at G5 and 2001 at G6. These Gwenffrwd sites have remained impoverished even though abundance has increased, as there is no healthy upstream population to supply the area. Diversification will rely on migration in from healthy populations downstream. At the time the spring samples are taken there is a more obvious effect of the natural background acidification in the catchment.

Blaenpelenna spring monitoring

The upstream site on the Blaenpelenna has had a varied total abundance over time with no obvious trends (Figure 3.49). At the site downstream of the minewater there was an increase in total abundance of invertebrates following minewater treatment in 1999. Further down on the river at P1 there were increased population numbers from 2000 onwards. A similar pattern in EPT abundance is obvious in Figure 3.50. The abundance of the Plecoptera order increases slightly following treatment and large increases in Ephemeroptera were observed. At both BP5 and P1 there were almost no Ephemeroptera existing prior to minewater treatment. At BP5 below the Garth Tonmawr discharge there was a significant reduction in total and EPT abundance in 2000. Figure 3.56 demonstrates the changes in total abundance and the mean dissolved iron concentration over the previous six months. It can be seen from this that the mean iron concentrations have been low and stable from 1999 onwards and provide no clue as to what may have reduced the population in 2000. A similar graph for the Ephemeroptera abundance is presented in Figure 3.57 again the iron is low and stable post treatment but a complete reduction in Ephemeroptera is seen in 2000.

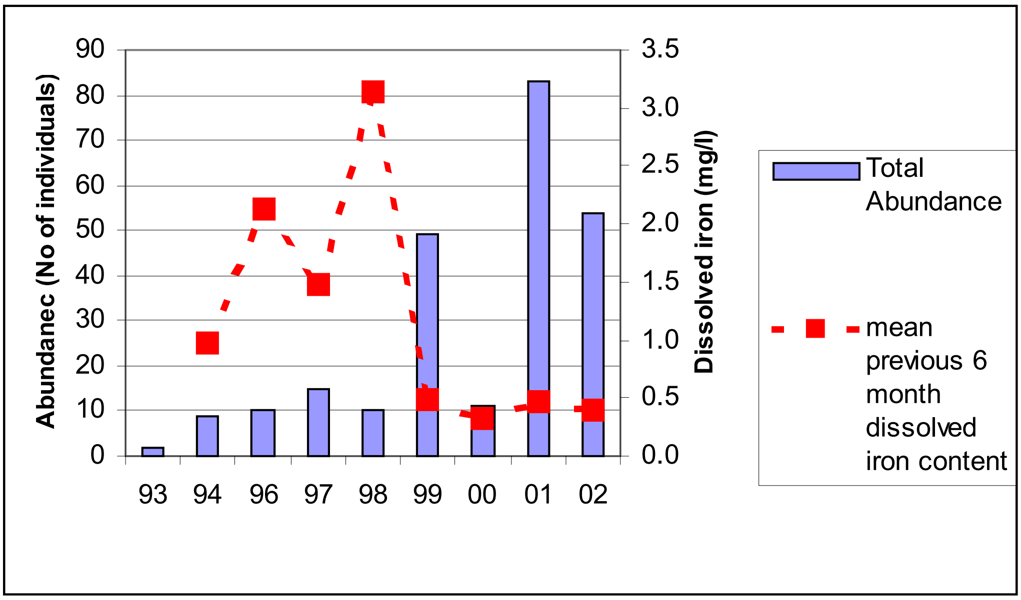


Figure 3.56 **Variations in spring total macroinvertebrate abundance and mean dissolved iron concentration in the previous 6 months at site BP5 on the Blaenpelenna**

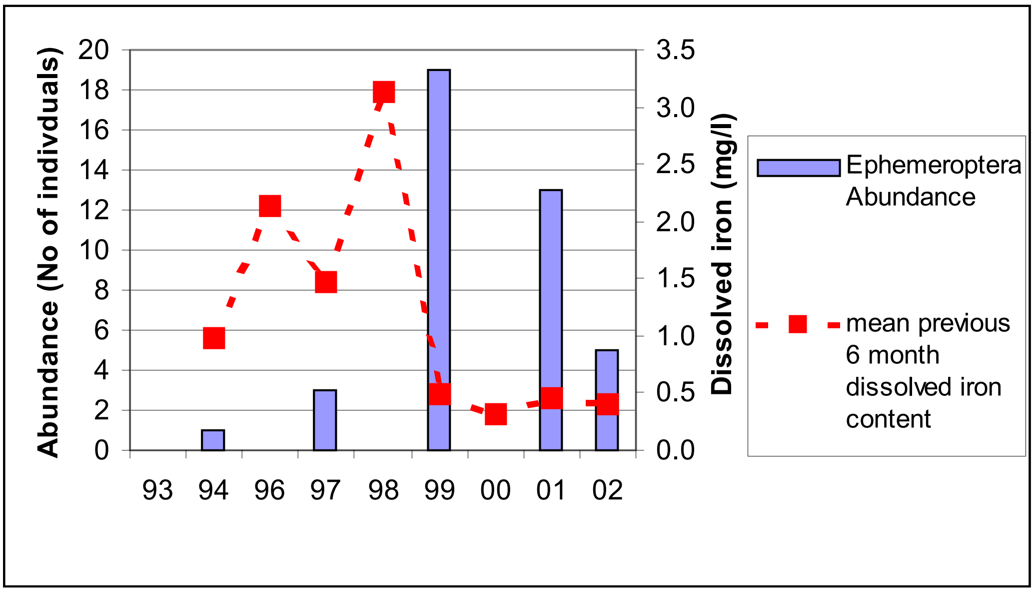


Figure 3.57 **Variations in spring Ephemeroptera abundance and mean dissolved iron concentration in the previous 6 months at site BP5 on the Blaenpelenna**

Pelenna spring monitoring

The total abundance at the two sites on the Pelenna is difficult to interpret due to the extremely large abundance found in 1999 (Figure 3.49). However the general trend appears to be for some increases from 1999 onwards. Figure 3.50 contains the EPT abundance's at these points. At Efail Fach (P2) the population does improve, especially in 2000 and 2002. From 1999 onwards the increase in Ephemeroptera is especially noticeable. Further down above the confluence with the Afan (P3) there is more variability over time but again marked increases in the relative proportion of Ephemeroptera to the other two orders have been found since 1999.

Investigating the TWINSPAN results at this point (Figure 3.53). It can be seen that both sites were in the Group 3 category in 1993 and 1994. They have then moved up through the groups. From 2000 to 2002 the site at Efail Fach (P2) was in group 2 and therefore had a population similar to that found on the upstream control site. P3 achieved group 1 membership in 2001 and 2002 and its population was therefore similar to that of the two sites on the Afan.

Afan and Fforch-Dwm summer monitoring

The two control areas will have the potential to be representative of differing areas of the catchment as discussed in the spring section above. The total macroinvertebrate abundance is presented in Figure 3.58. From this it can be seen that while there have been variations in the size of the populations on the Afan and Fforch-Dwm there are no obvious trends over time.

The TWINSPAN analysis was undertaken on the summer dataset as a method of identifying changes in macroinvertebrate diversity at the monitoring points over time. Four TWINSPAN groups were identified that represented sites which were essentially similar in taxonomic composition. The relationship between these four groups and a selection of the other biotic indices is presented in Table 3.12. It can be seen that the BMWP, number of taxa and total abundance all decrease with passage down through the groups. The relationship between these four groups and the BMWP and abundance data is shown in the boxplots in Figures 3.59 and 3.60. It can be seen that both the BMWP and abundance are significantly lower in the more impoverished groups 3 and 4. Groups 1 and 2 are much more similar when viewed in terms of the BMWP scoring population and the total macroinvertebrate abundance.

The TWINSPAN group membership results for all the sites and years in the summer are presented in Figure 3.61. From this it can be seen that the two sites on the Afan had populations which fell within the top group (group 1) during each sample year. This indicates the range of organisms that the lower sites on the Pelenna may be expected to reach. The Fforch-Dwm has had group 2 membership for the whole monitoring period. This indicates the population which the sites on the Nant Gwenffrwd and Blaenpelenna could be expected to reach.

Nelson and Roline (1996) identified that the number of taxa in the Ephemeroptera order increased following minewater treatment along with Plecoptera and Trichoptera. Malmqvist and Hoffsten (1999) and Soucek et al. (2000) identified Ephemeroptera as being intolerant to elevated metal concentrations. EPT and Ephemeroptera (mayfly) abundance was investigated further at the sites in this study. The changes in EPT populations are shown in Figure 3.62. From this it can be seen that the distribution is similar to that of total abundance and does not demonstrate any changes over time. The Ephemeroptera abundance is presented as the lower section of the stacked bar charts in Figure 3.62. Again the populations in the three control sites are variable over time with no discernable trends.

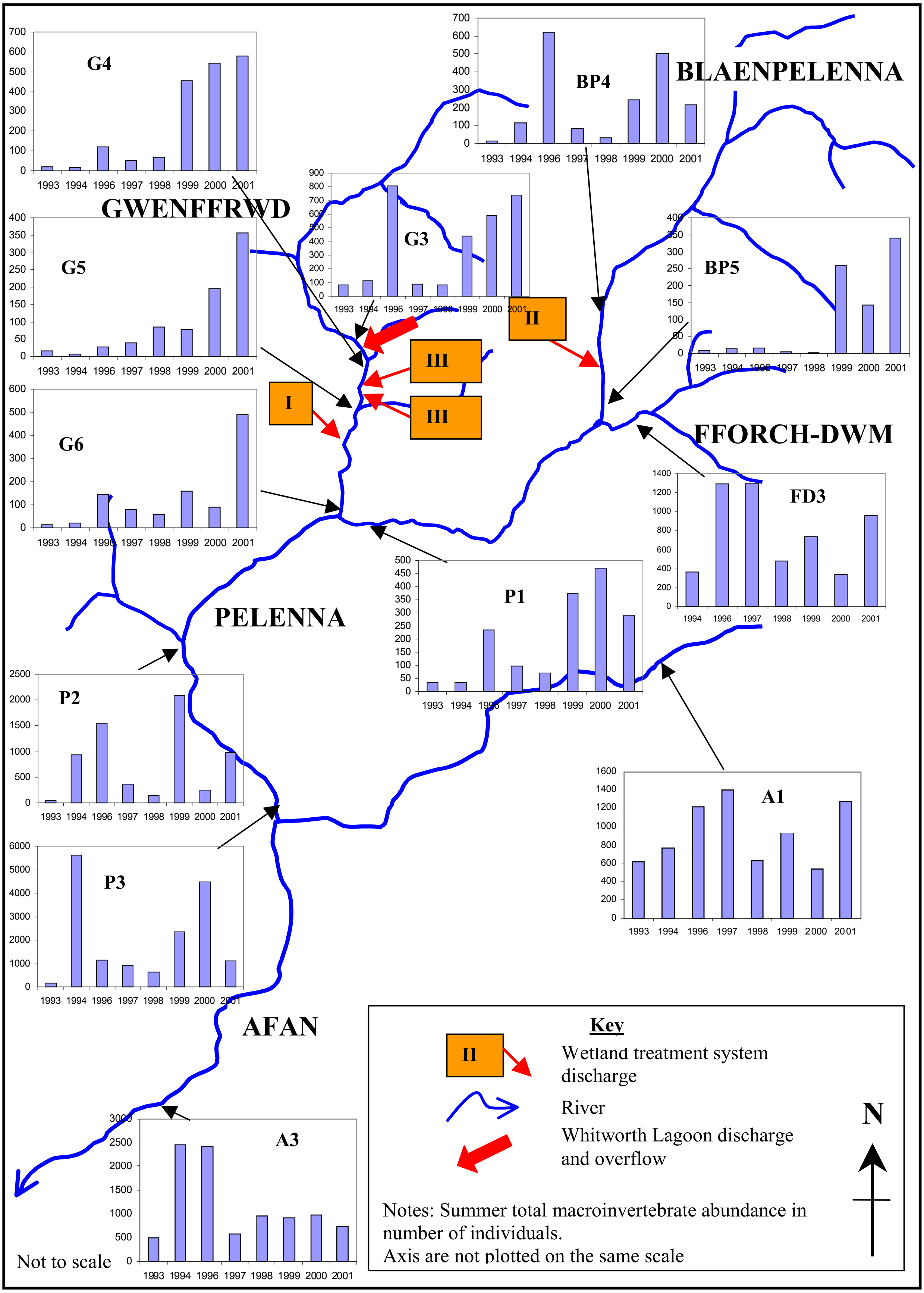


Figure 3.58 Summer total macroinvertebrate abundance.

Table 3.12 Biotic indices data for the summer TWINSPAN classification groups

	TWINSPAN group	BMWP[1]	Number of taxa[2]	Total abundance[3]
Summer	1	86.7	14.6	887
	2	82.0	13.5	810
	3	53.4	9.7	142
	4	24.5	5.1	16

Notes: [1]Mean Biological Monitoring Working Party (BMWP) score
[2]Mean number of taxa
[3]Mean Total abundance

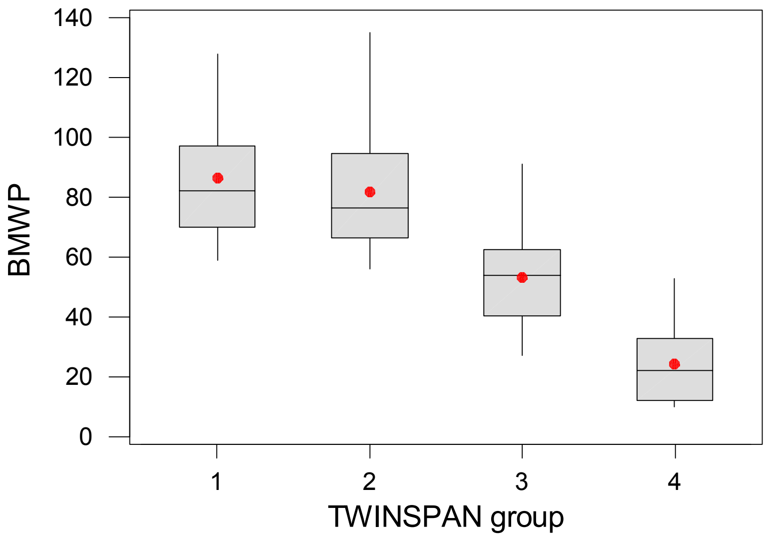


Figure 3.59 Variations in the BMWP score of the four summer TWINSPAN groups

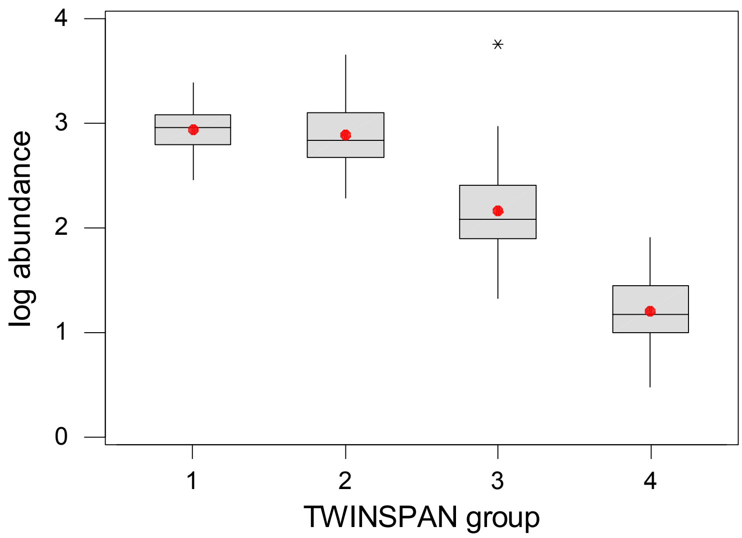


Figure 3.60 Variations in $\log_{10}$ total abundance between the four summer TWINSPAN groups

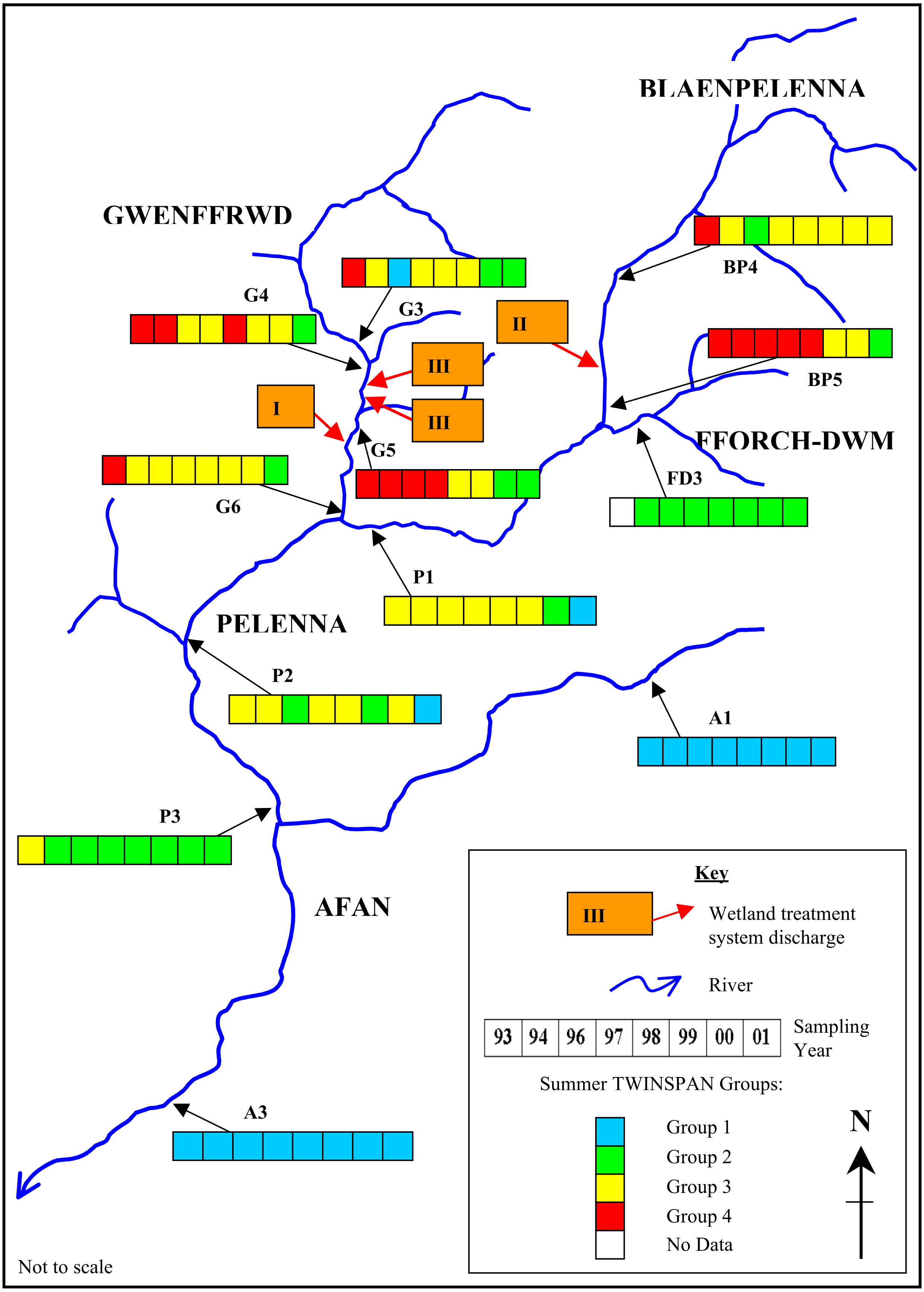


Figure 3.61 Summer TWINSPAN results

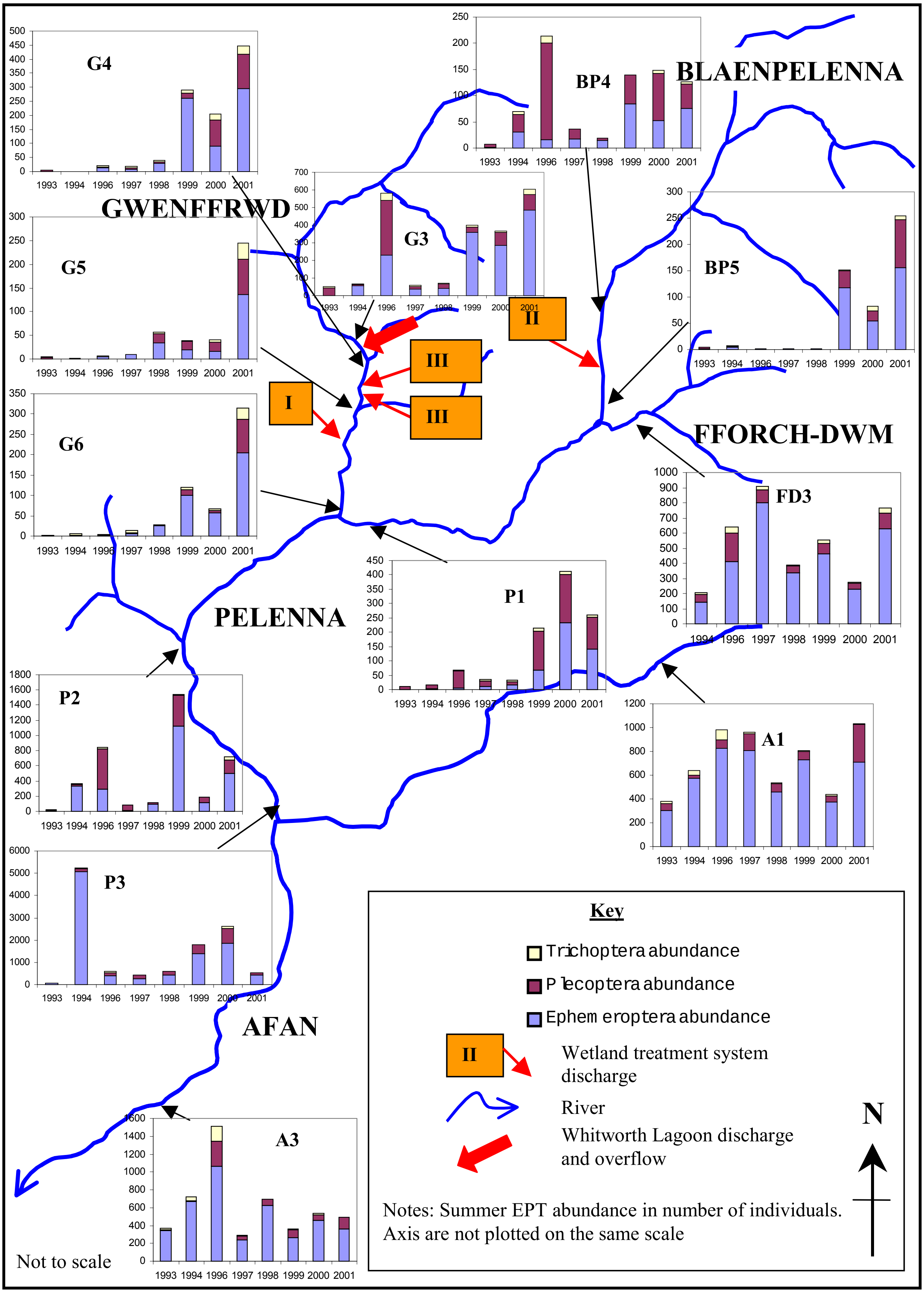


Figure 3.62 Summer Ephemeroptera, Plecoptera and Trichoptera (EPT) abundance

Nant Gwenffrwd summer monitoring

From Figure 3.58 it can be seen that all the sites on the Nant Gwenffrwd generally had very impoverished populations pre treatment. The year following minewater treatment the abundance increased markedly at site G4 below the original discharge point. At the upstream site (G3) a similar recovery was seen and the reason for this is not known. Further downstream on the Gwenffrwd the abundance increased a couple of years after treatment.

A similar pattern was evident in the abundance of the minewater intolerant EPT orders at site G3 (Figure 3.62). However the 2000 data at site G4 shows a drop in the number of these groupings. The Ephemeroptera abundance especially was reduced in 2000 (Figure 3.60). This reduction in the minewater intolerant species at G4 is in response to the overflow from Whitworth A that was occurring at this time. Figure 3.63 demonstrates the variations in total abundance with changes in the mean dissolved iron concentration over the previous six months. It can be seen that the increase in iron concentrations in 2000 did not obviously impact the overall abundance of invertebrates at this site. However Figure 3.64 contains a similar graph for the metal intolerant Ephemeroptera abundance. Here the increased mean iron levels did reduce the population at this point. It can be seen in Figure 3.62 that the overflows did not have such an obvious effect on the Plecoptera and Trichoptera abundances. Of the Ephemeroptera order, *Ephemerella ignita*, a species of mayfly, was the main species that characterized the recovery of this catchment. It exists in the control sites in fairly stable numbers but was not evident at any of the minewater impacted sites. Following treatment of the minewaters, *Ephemerella ignita* were one of the species to obviously recover. *Leuctra spp.* (stonefly) of the Plecoptera order were also identified as indicators of the recovery of the Pelenna catchment, but the improvements were not as marked as those of *Ephemerella ignita.*

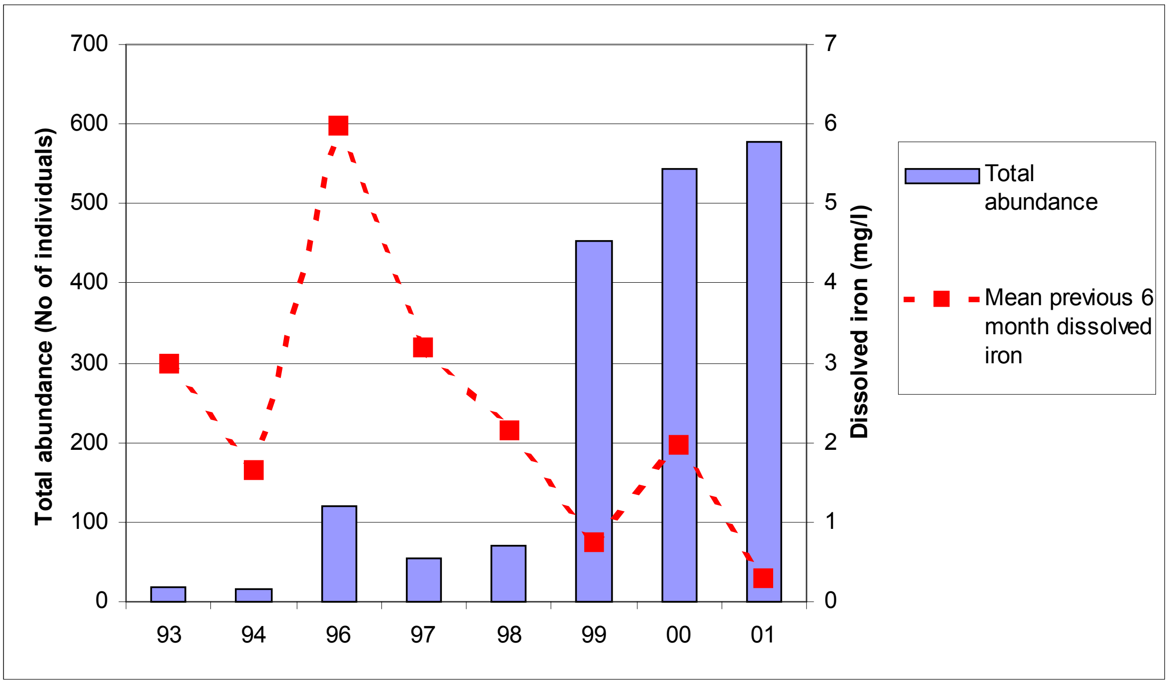


Figure 3.63 Variations in total macroinvertebrate abundance in summer and mean dissolved iron concentration in the previous 6 months at site G4 on the Nant Gwenffrwd

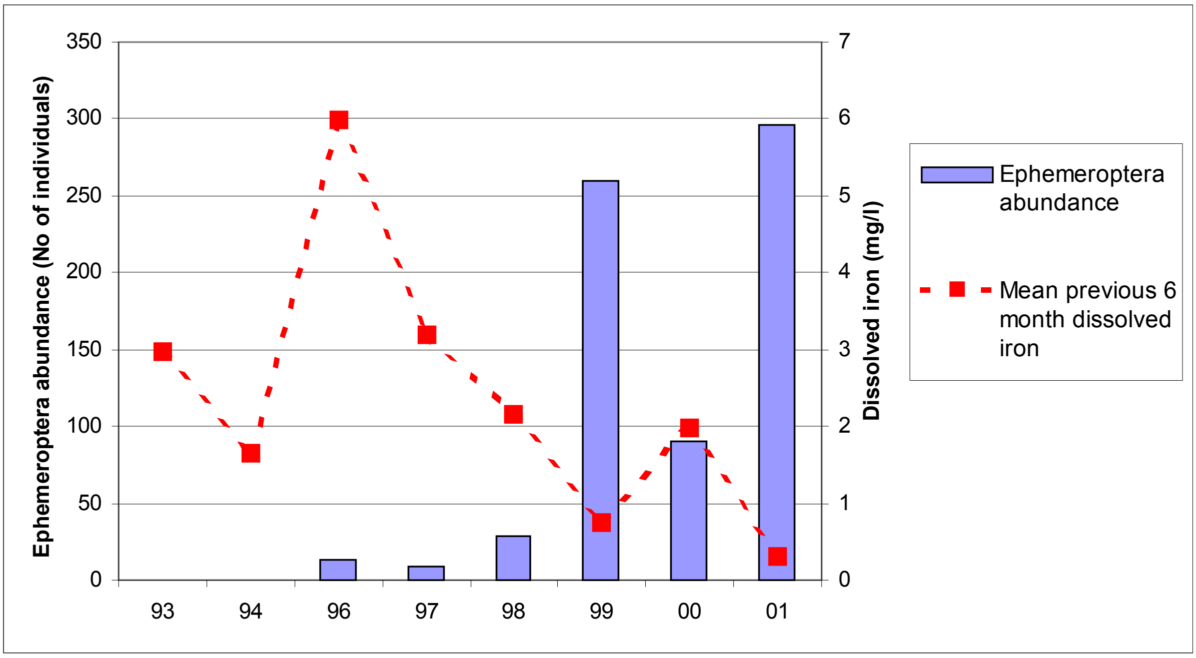


Figure 3.64 Variations in Ephemeroptera abundance in summer and mean dissolved iron concentration in the previous 6 months at site G4 on the Nant Gwenffrwd

Investigating further the changes in species diversity at the site over time can be done with the results of the TWINSPAN analysis in Figure 3.61. The upstream site in this catchment (G3) has improved from being generally group 3 to group 2 in 2000 and 2001. Site G4 downstream was generally group 4 before minewater treatment and then moved to group 3 in 1999 and up to group 2 in 2001. Site G5 again demonstrated this trend from group 4 to 3 to 2. It appears that the sites are increasing diversity to approach that seen at the upstream control site on the Fforch-Dwm. These changes are however taking a couple of years to develop. This will obviously have been slowed down by the problems encountered with the overflows in 2000.

Blaenpelenna summer monitoring

The total abundance at site BP4 upstream of the Garth Tonmawr minewater has varied over time (Figure 3.58). There does appear to be an increase in abundance since 1999 but the changes are not as great as those seen below the minewater. Site BP5 has increased markedly in terms of numbers of individuals from almost nothing before treatment. These improvements have been seen the year following minewater treatment. Further downstream on the Blaenpelenna at P1 there have also been post treatment improvements in population numbers from 1999.

Looking at the changes over time in terms of diversity on the Blaenpelenna. The upstream site BP4 has varied occasionally in terms of TWINSPAN group membership but has generally been in group 3 (Figure 3.61). This indicates an impoverished population compared to the Fforch-Dwm but is a likely result of the unbuffered episodic acidification effects this high in the catchment. Downstream of the minewater and treatment system at site BP5 it was in the most affected group 4 from 1993 to 1998. In 1999 the year after minewater treatment the site went up to a group 3 membership to match the population found upstream. This increase is most likely to have been caused by the recolonisation of the area by downstream drift from the upstream population. Two years later in 2001 the site went up to group 2 membership and therefore became similar to the nearby control stream the Fforch-Dwm.

Further downstream above the confluence with the Gwenffrwd at P1 the site had been at group 3 membership before treatment. Again the population numbers increased quickly following minewater treatment the diversity did not change straight away. Two years after minewater treatment the site achieved group 2 membership and then group 1 the next year. It is likely that some of the delay in species diversification would have been caused by the time taken for migration of the macroinvertebrates from the healthy populations downstream.

Again the changes in EPT abundance at these sites appear to be similar to the changes in total abundance (Figure 3.62). The populations of these metal pollution intolerant species appear to improve quickly at all sites on the Blaenpelenna immediately following minewater treatment. Figures 3.65 and 3.66 respectively demonstrate the variations in total and Ephemeroptera abundance with changes in mean dissolved iron concentration. It can be seen that the abundances increased markedly once the iron levels were reduced.

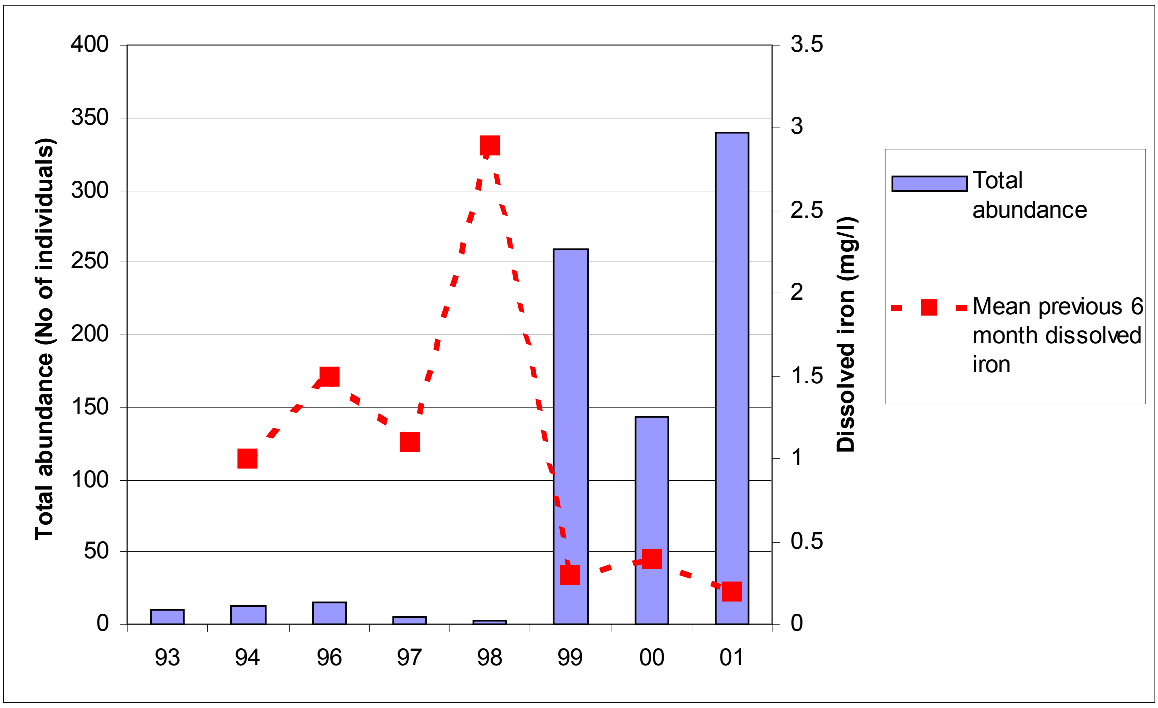


Figure 3.65 Variations in total macroinvertebrate abundance in summer and mean dissolved iron concentration in the previous 6 months at site BP5 on the Blaenpelenna

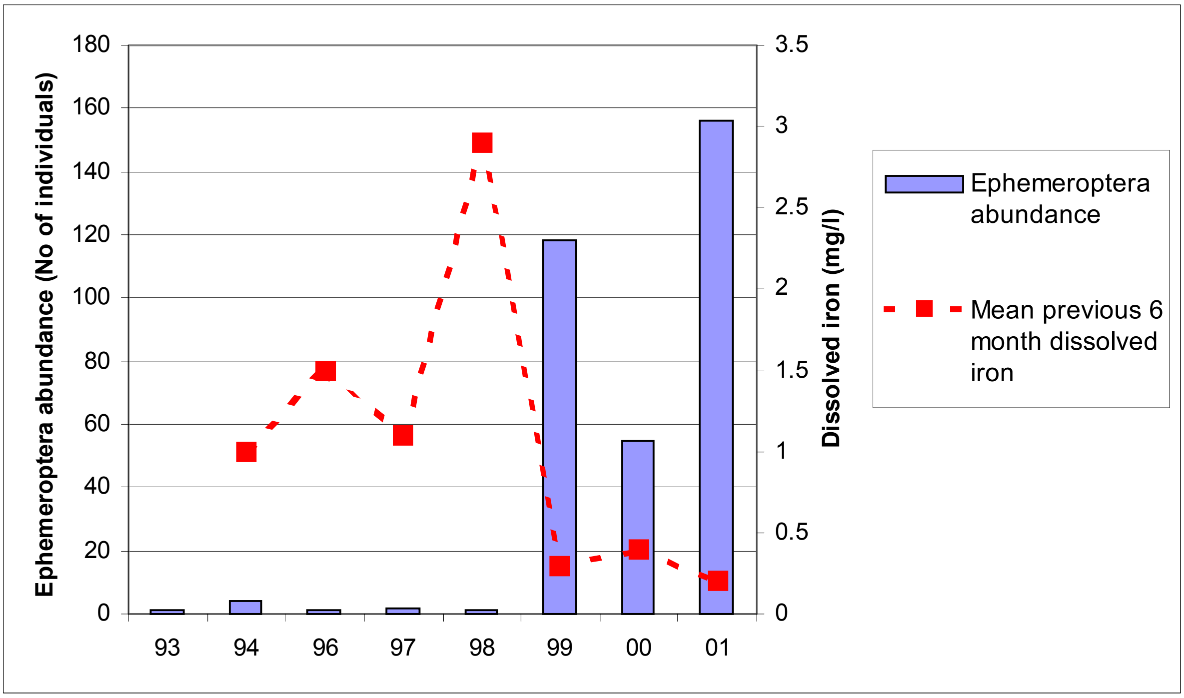


Figure 3.66 Variations in Ephemeroptera abundance in summer and mean dissolved iron concentration in the previous 6 months at site BP5 on the Blaenpelenna

Pelenna summer monitoring

At the two sites on the Pelenna the total and EPT abundances tend to be quite variable over time (Figures 3.58 and 3.62). There did appear to be some increases from 1999 onwards. At site P2 at Efail Fach the abundance's for 2000 are quite reduced which may be as a result of the overflows that were occurring that year. Looking at the TWINSPAN analysis at these points (Figure 3.61), the downstream site has maintained group 2 membership across most years and has not approached the populations found on the Afan. Site P2 at Efail Fach has varied over time between groups 3 and 2 and in 2001 achieved group 1 status. This indicates that its population had recovered to be similar to that found at the two sites on the Afan.

Population relationships to water quality variables.

Each yearly dataset from each site was categorized into a TWINSPAN group. These groupings indicate that the sites have similar taxonomic compositions. The range of mean water quality determinands for the previous six months at these points can be plotted. This will give an insight into the status of a population at a site with a certain water chemistry. Figures 3.67 to 3.72 respectively demonstrate the range of mean dissolved iron, mean pH and mean alkalinity for the four TWINSPAN groups in spring and summer. It can be seen that there is considerable overlap in the dissolved iron levels observed for the sites in groups 1 and 2. Then the more impoverished groups 3 and 4 have more distinct iron levels identified. It would therefore be logical to assume that the dissolved iron content is having more control on the sites in groups 3 and 4. Looking at the pH values observed the four groupings have fairly distinct pH ranges. Groups 3 and 4 have a lot more overlap in their pH ranges.

Membership of groups 1 and 2 that were essentially similar in terms of iron content therefore appear to be controlled more by the pH characteristics of the water. This is also reflected in the relationship between alkalinity and TWINSPAN groupings in Figures 3.71 and 3.72.

It would appear that dissolved iron content in the watercourse exerts the strongest control over the population of invertebrates at a site. High dissolved iron will be toxic to many species and reduce the population. Then as the iron levels are reduced the pH of the watercourse becomes a more important control over the diversity and abundance of species that will be found.

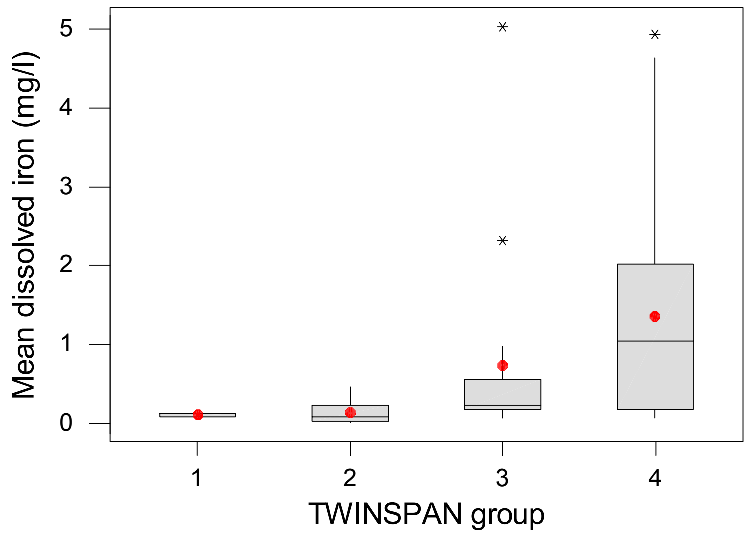


Figure 3.67 Range of mean dissolved iron for the TWINSPAN groupings in spring

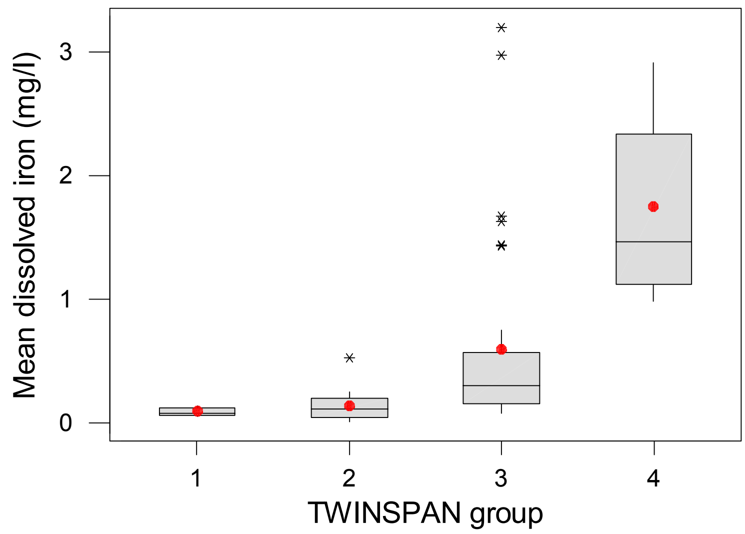


Figure 3.68 Range of mean dissolved iron for the TWINSPAN groupings in summer

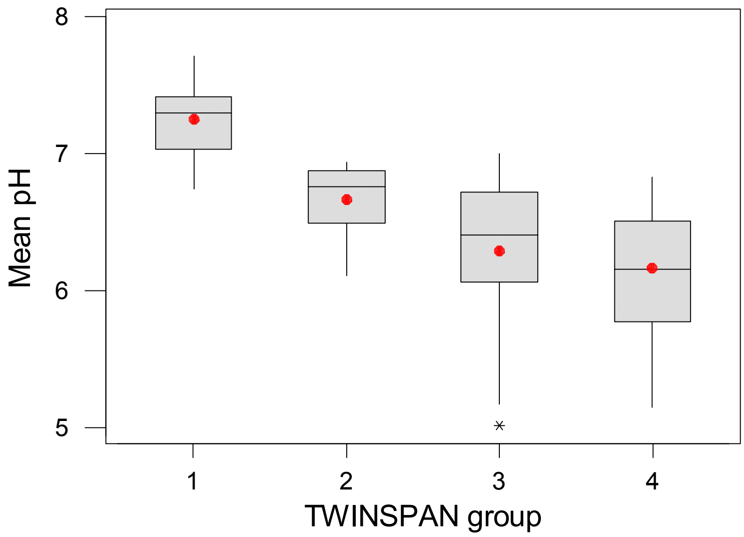


Figure 3.69 Range of mean pH for the TWINSPAN groupings in spring

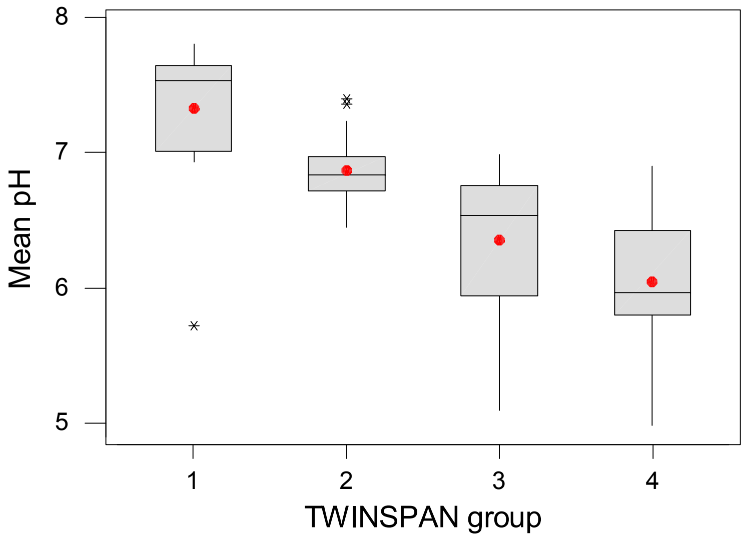


Figure 3.70 Range of mean pH for the TWINSPAN groups in summer

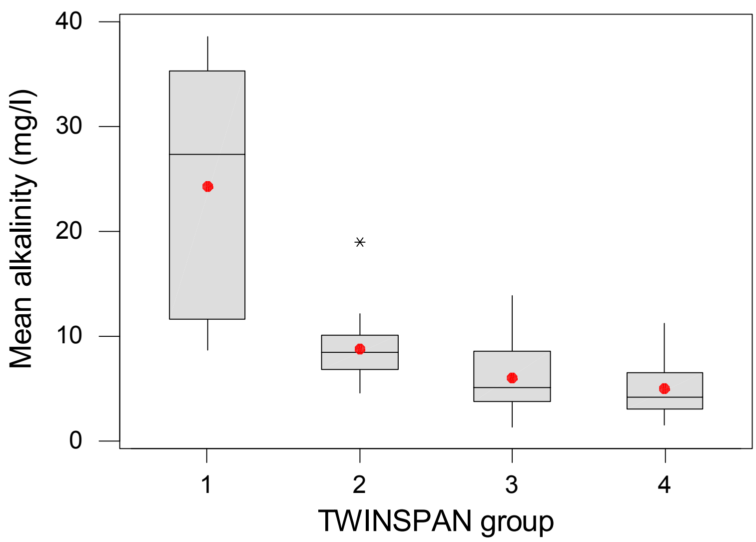


Figure 3.71 Range of mean alkalinity for the TWINSPAN groupings in spring

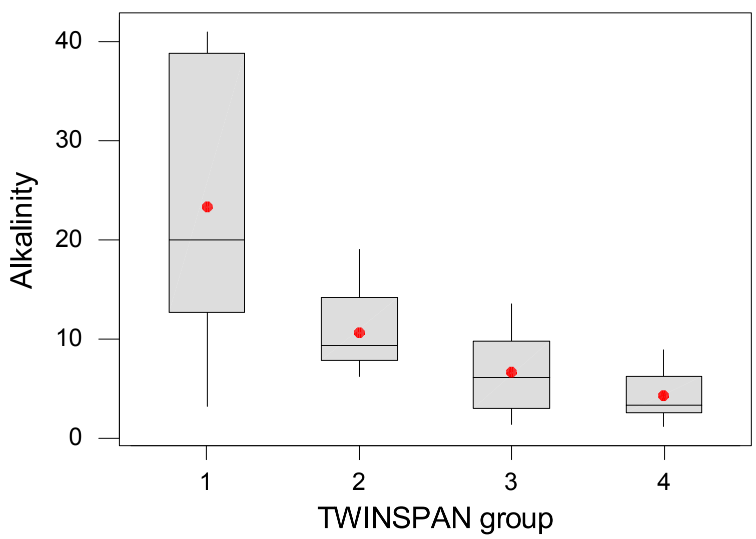


Figure 3.72 Range of mean alkalinity for the TWINSPAN groups in summer.

The mean previous six-month water quality data has been plotted against the log_{10} total abundance data for both spring and summer. The results are shown for dissolved iron, pH and alkalinity in figures 3.73 to 3.78. It can be seen that the high abundance results are clustered together at a low range of dissolved iron. As the iron concentration increases the abundance is lower but demonstrates a high degree of variability. Figures 3.75 and 3.76 demonstrate the changes in abundance with mean pH. Here it can be seen that as a general rule abundance increases with an increase in pH. However the datasets are quite varied and would not have a statistically significant relationship. Again a similar pattern is found with the alkalinity results. In general an increase in abundance is observed at higher alkalinity values but the datasets are again quite varied. Plots of the log Ephemeroptera abundance and dissolved iron content in both spring and summer are presented in Figures 3.79 and 3.80. A similar distribution to the total abundance is evident with a cluster of high abundance found in low dissolved iron content samples. Then as dissolved iron increases the Ephemeroptera abundance decreases.

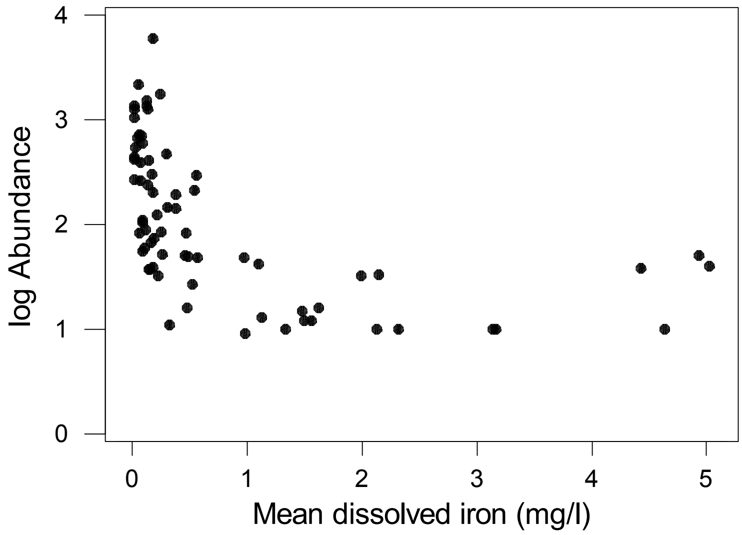


Figure 3.73 Variations in $\log_{10}$ abundance in spring with changes in mean dissolved iron at all sites and years on the Pelenna

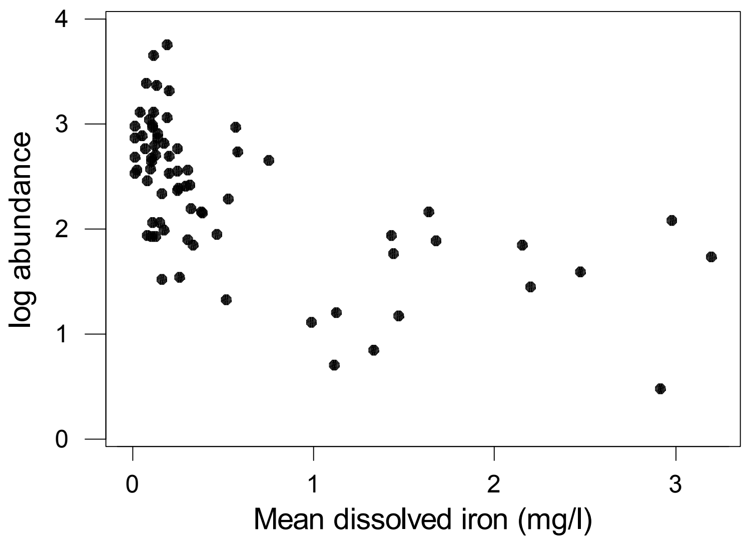


Figure 3.74 Variations in $\log_{10}$ abundance in summer with changes in mean dissolved iron at all sites and years on the Pelenna

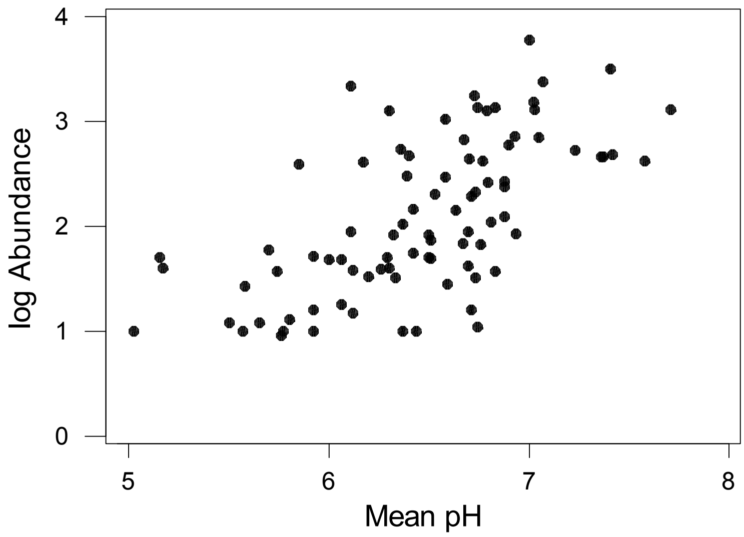


Figure 3.75 Variations in $\log_{10}$ abundance in spring with changes in mean pH at all sites and years on the Pelenna

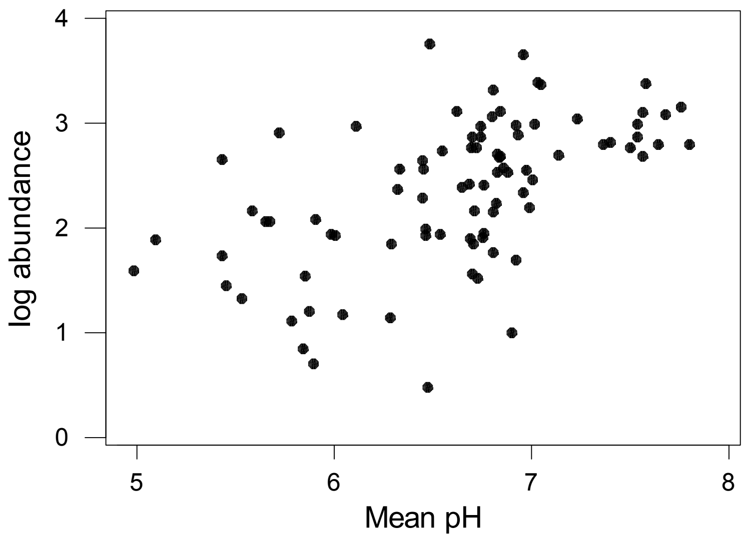


Figure 3.76 Variations in $\log_{10}$ abundance in summer with changes in mean pH at all sites and years on the Pelenna

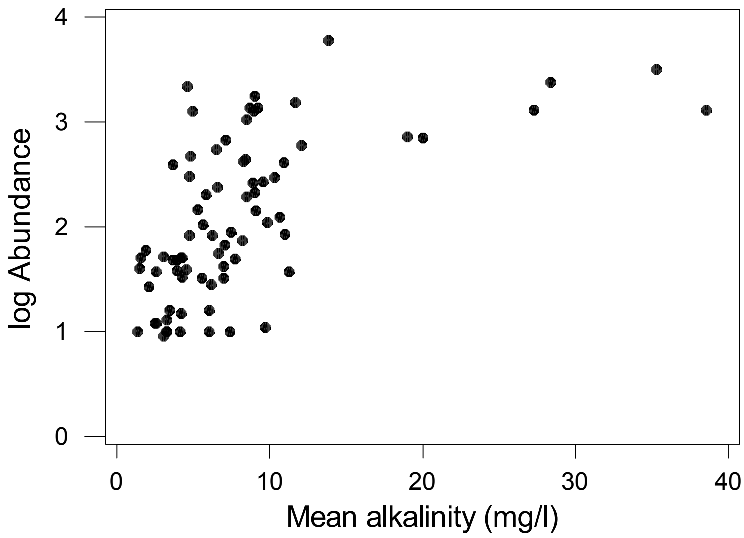


Figure 3.77 Variations in $\log_{10}$ abundance in spring with changes in mean alkalinity at all sites and years on the Pelenna

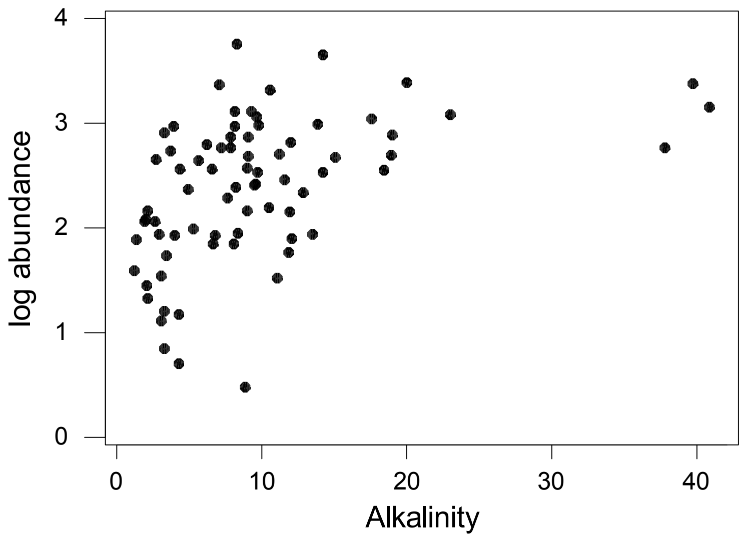


Figure 3.78 Variations in $\log_{10}$ abundance in summer with changes in mean alkalinity at all sites and years on the Pelenna

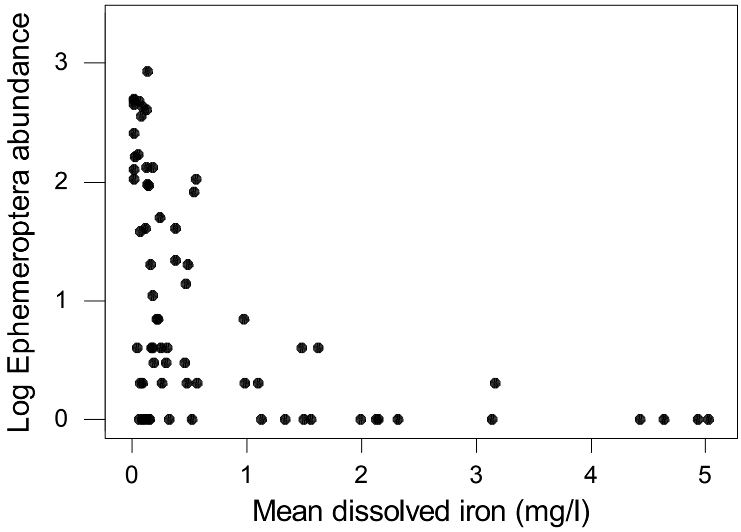


Figure 3.79 Variations in $\log_{10}$ Ephemeroptera abundance in spring with changes in mean dissolved iron at all sites and years on the Pelenna

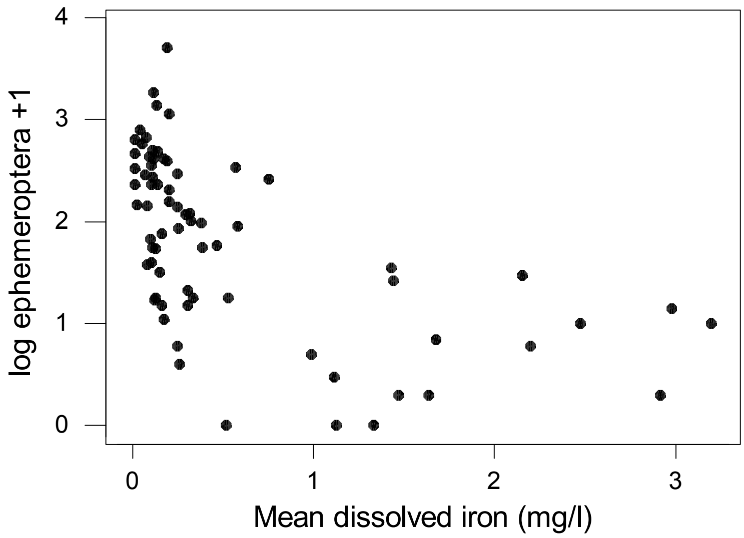


Figure 3.80 Variations in $\log_{10}$ Ephemeroptera abundance in summer with changes in mean dissolved iron at all sites and years on the Pelenna

3.3.3 Fisheries populations

Trout were the only salmonid species caught on the Pelenna and its tributaries. Occasional eels were found but the river has a main interest as a trout fishery. Population densities were calculated for both fry and parr from the observed fish catches for each of the sites using the method of Carle and Strub (1978). Upper and lower 95% confidence intervals were utilized to assess if the population densities were significantly different before and after treatment (R&D Project Record P2-181/PR Appendix 5). The changes in population density each year are shown for fry and parr in Figures 3.74 and 3.75 respectively. These graphs include the predicted habitat quality score (HQS) that indicates the expected fish population density based on the habitat available assuming pristine water quality. National Fisheries

Classification Scheme (NFC) scores were calculated for each site for each year and a predicted score based on the HQS was identified. The results from this are shown in Table 3.13.

Table 3.13 Observed and predicted National Fisheries Classification Scores.

Site	1993	1994	1996	1999	2000	2001	2002	Predicted
G3[1]	F	F	F	F	F	F	F	**C**
G4	F	F	F	C	C	B	C	**B**
G5	F	E	E	C	C	A	B	**C**
G6	F	F	F	C	B	B	B	**C**
BP4	E	E	E	D	D	D	D	**C**
BP5	E	E	D	B	C	B	C	**B**
FD3	-[2]	C	A	A	D	A	A	**A**
P1	E	D	C	C	A	B	C	**C**
P2	D	C	B	B	B	A	B	**D**
P3	D	B	C	C	B	C	B	**D**

Note: [1]Site G3 is inaccessible to fish due to a migratory barrier
[2]Fforch-Dwm not fished in 1993.

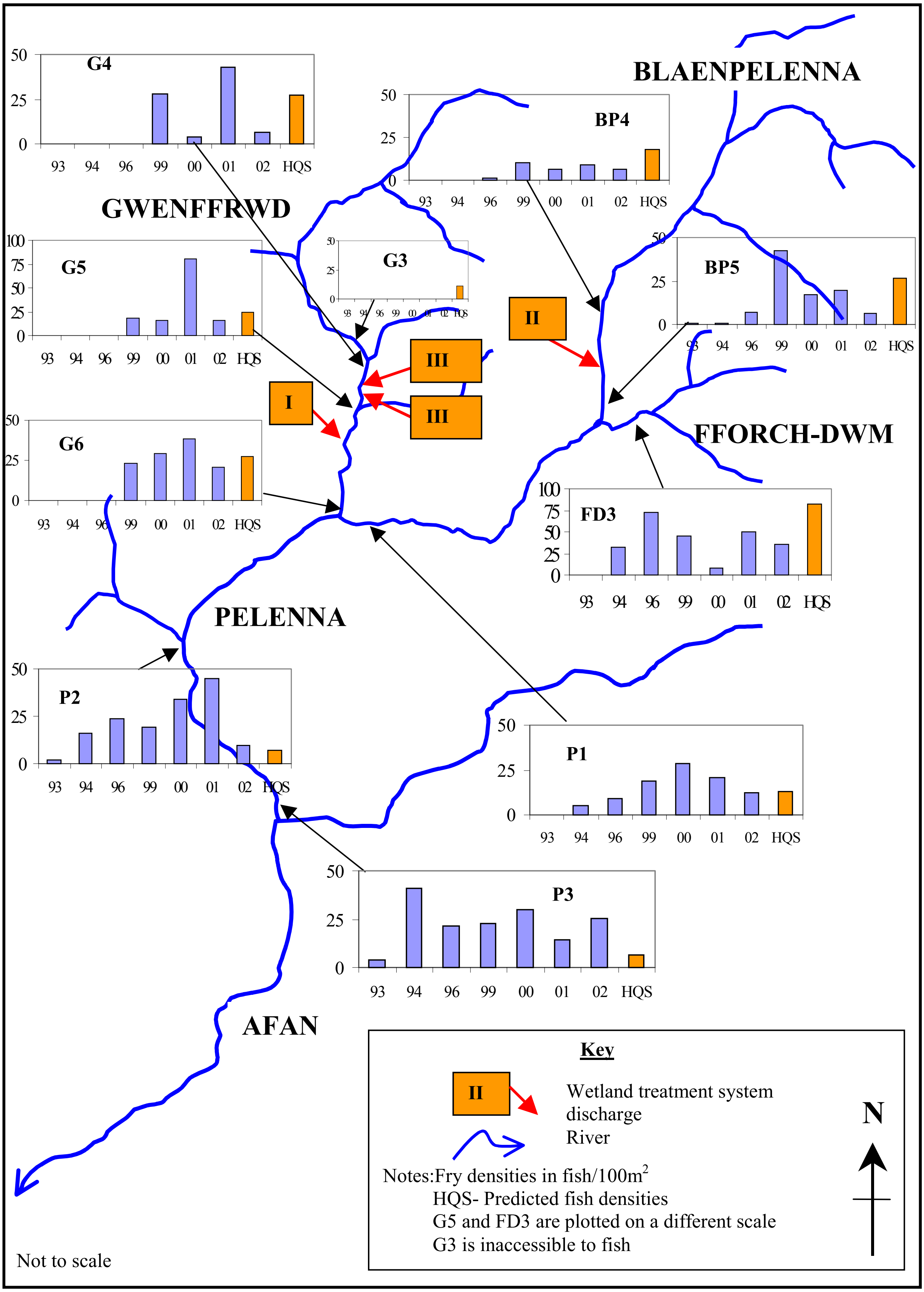


Figure 3.81 Observed and expected trout fry densities.

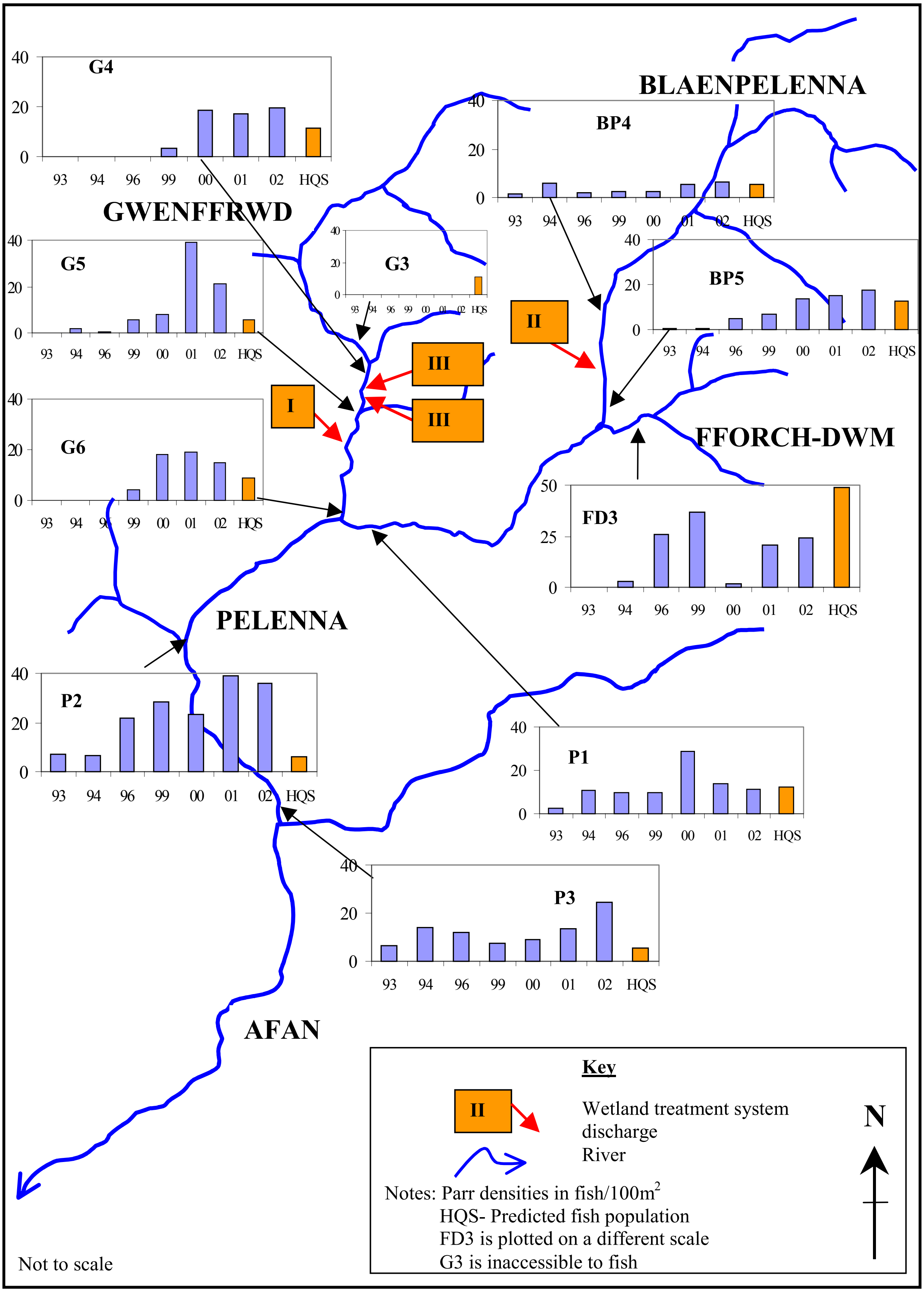


Figure 3.82 Observed and expected trout parr densities.

Figures 3.74 and 3.75 demonstrate that the Fforch-Dwm, the control stream, has had populations of both fry and parr during all the survey years. The population density has been variable and generally below the level predicted. However it mostly falls into the NFC class A category (Table 3.13) which represents a good fish population for its type of fishery. The year 2000 data was very low for both fry and parr and this was thought to be due to habitat loss caused by erosion of a spoil tip at the survey site during that year.

Nant Gwenffrwd

The upstream site on the Gwenffrwd (G3) is inaccessible for fish due to a barrier to migration immediately downstream of the Whitworth Lagoon. This takes the form of a 2-3m freefall drop for the water out of a culverted section. Therefore no fish have been observed at this sample point on any surveys and explains the continuous class F results observed in Table 3.13.

The sites downstream from this culvert on the Gwenffrwd, G4, G5 and G6 have all demonstrated significant improvements in both fry and parr populations following treatment (Figures 3.74 and 3.75). None of the sites had any fry before treatment. At site G4 immediately downstream of the original Whitworth A discharge, the population recovered quickly the year following minewater treatment. It recovered to be almost at the predicted NFC population level (Table 3.13). It has been observed that trout have been migrating up the Pelenna to spawn each year (M. Brett, Fisheries Enforcement Officer, Personal Communication). However the water quality and ochre smothering have meant that eggs and fry have not survived. Once the minewaters were fully treated the eggs survived and a healthy fry population returned.

In the spring and summer of 2000 there were problems with the inlet structure to the Whitworth A wetlands blocking. This caused the untreated minewater to overflow and raise the dissolved iron level in the river as discussed in section 3.3.1. The impact this had on the trout fry is very significant, the population is severely reduced compared to the previous year. Figure 3.83 graphically demonstrates the changes in fry population at this sample site. The mean dissolved iron content for the previous six months is included on the graph. It shows quite well that when the minewaters overflow and the dissolved iron level exceeds the EQS of 1mg/l the fry population is severely reduced. In 2001 the population again demonstrates a very good fry density as these overflow problems had been sorted. However due to a lack of routine maintenance the system partially blocked again in the spring and summer of 2002 and the fry population was again impacted and reduced to the 2000 levels. This indicates the immediate effect that the overflows have on the susceptible early stages of the fish. It also demonstrates how quickly they can recover in a minewater impacted stream if the water quality issues are effectively dealt with and there are spawning fish available.

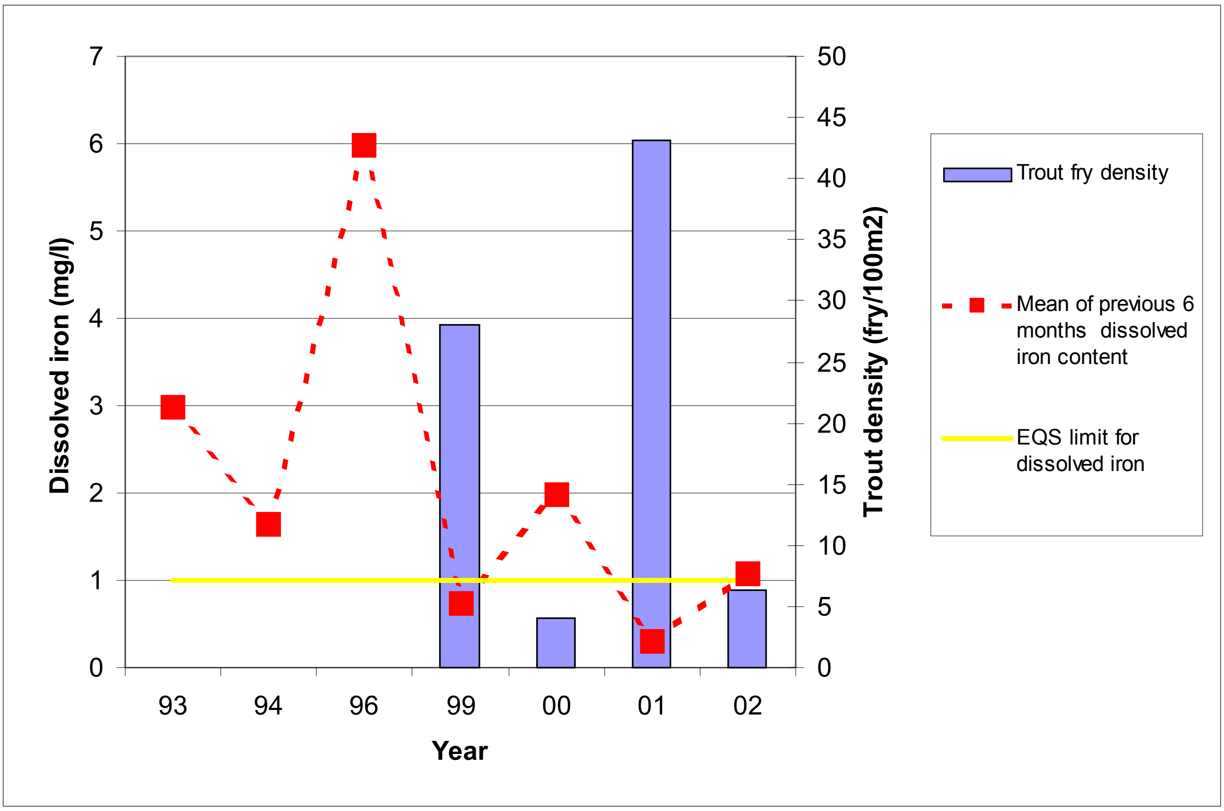


Figure 3.83 Variations in trout fry density at site G4 with variations in previous six monthly mean dissolved iron content.

The parr population at this site again was non existent before treatment (Figure 3.82). In 1999, the year after minewater treatment there was a small population well below the HQS. This had most likely migrated in from downstream on the Pelenna. In 2000 the population was greater than the HQS, indicating the maturation of the fry population from the previous year. This population has been maintained at this level in the following years. The minewater overflows do not appear to affect the parr population as much as the fry. This is presumably due to the fact that the egg and fry life stages are more vulnerable than the parr. The parr population is also likely to be more mobile and can represent fish that have moved into the area after maturing from healthier fry populations downstream.

The next two sites down the Gwenffrwd, G5 and G6 again demonstrate a good recovery of parr over time (Figure 3.82). The fry (Figure 3.81) demonstrated the same quick recovery seen at G4. They appear to have some reductions in populations in 2000 and 2002 as a result of the overflows from the treatment systems. These are not however as noticeable as those seen upstream. This indicates that the overflows that occurred did not have an impact for a great distance downstream despite being obvious in the water quality data as far as Efail Fach (site P2).

Blaenpelenna

On the Blaenpelenna, which was impacted by the Garth Tonmawr minewater, the site upstream of the minewater (BP4) has had a low population density of parr that has been consistent over time (Figure 3.82). It rose slightly in the last two survey years to be greater than the predicted population. The low HQS value and observed populations at this site are mostly as a result of the poor habitat. The fry at this site have recovered from almost no fish in 1993, 1994 and 1996 to approximately half of the predicted population from 1999 onwards (Figure 3.81). At this site the combined NFC is still in the category D below what had been predicted (Table 3.13).

Figure 3.81 demonstrates that downstream of the wetland treatment scheme discharge, at BP5, the numbers of fry have risen greatly following treatment. Their populations have been fairly variable following this date. The population appeared to be reduced in 2002. This was originally thought to be as a result of the overflow of untreated minewater from the natural wetland. This was causing the deposition of ochre on the river bed at this point. However this appears less likely after investigating the water quality data from site BP5. There were no obvious changes in any determinands that would impact the fish at this point. However short term impacts or pulses of contaminants may not be identified in the monthly sampling dataset. The parr population at this point again recovered from a very impoverished status pre treatment and has increased each year from 1999. This site is now more productive for parr than it was predicted that it would be. In 2001 the combined trout population was at the level predicted by HABSCORE (Table 3.13) but due to the reduction in fry in 2002 it fell back under this level.

Further downstream on the Blaenpelenna, at site P1, the impact of the minewaters on fish was not quite as severe. However recovery of fry can be seen with a marked difference between the 1996 population and the 1999, 2000 and 2001 populations. The results for fry in 2002 were again reduced as was seen at the upstream sites. The parr population does not appear to have significantly changed post treatment except for in year 2000.

Pelenna

Further downstream at the two sites on the Pelenna some changes in fish populations have been observed. At Efail Fach (P2) both the fry and parr populations (Figures 3.74 and 3.75 respectively) have increased markedly in most years following minewater treatment. There has been a decrease in the fry population in 2002, which is similar to that seen on the Blaenpelenna. The cause for this is unknown. At the bottom of the Pelenna (P3) the populations of both fry and parr have varied over time and are healthy when compared to the predicted population levels (Table 3.13). This site is sufficiently far downstream from the minewater sources that it did not appear to be significantly effected by the minewater discharges in the past. Following the construction of the treatment schemes there has been no noticeable change in fish populations at this point. This was to be expected after investigating the water quality of the river before and after treatment in section 3.3.1. None of the samples taken at this point have had dissolved iron levels exceeding the EQS of 1mg/l.

Economic benefits of improved fishery

The estimated economic benefit of the observed increases in trout parr densities in the Pelenna catchment, based on the estimated subsequent increase in the market value of the River Afan sea trout fishery, is approximately £181,500. This figure is based on a number of assumptions, many of which are based on studies on other rivers in South Wales; therefore the figure is only expressed as a guide and should not be quoted as an absolute value. The figures used to derive the estimated economic benefit are shown in Tables 3.14 and 3.15.

Table 3.14 Estimated increases in trout parr numbers following minewater treatment

Stretches (representative sites)	Total length (km)	Mean #parr/50m 93-96	Mean #parr/50m 99-02	Mean total # parr 93-96	Mean total # parr 99-02	Increase in #parr
G4, G5 & G6	1	0.24	24.63	5	493	488
P1	2.5	22.44	46.54	1122	2327	1205
P2	2	37.39	99.64	1495	3986	2490
P3	1	43.98	56.54	880	1131	251
BP5	0.5	4.69	30.92	47	309	262
Total	**7**	**108.74**	**258.27**	**3549**	**8245**	**4696**

Table 3.15 Estimated increase in capital value of Afan sea trout fishery resulting from increased numbers of trout parr in the Pelenna catchment

Stretches (representative sites)	Increase #smolts[1]	Increase in adults returning[2]	#including previous spawners[3]	#returning within season[4]	Increase in rod catches[5]	Estimated value (£)[6]
G4, G5 & G6	244	37	44	31	6	18851.43
P1	602	90	108	76	15	46560.35
P2	1245	187	224	157	30	96218.42
P3	126	19	23	16	3	9704.73
BP5	131	20	24	17	3	10133.38
Total	**2348**	**352**	**423**	**296**	**57**	**181468.31**

Notes: Figures used in calculations and reference:

[1]Parr-smolt survival	50%	Kennedy, 1988; assumed equivalent to salmon.
[2]Smolt-adult survival & return to freshwater	15%	Scranney, 1998
[3]Allowance for second spawning	120%	Scranney, 1998
[4]Proportion returning within season	70%	Scranney, 1998
[5]Exploitation rate	16.10%	Solomon, 1995 (River Tawe)
[6]Capital value of rod-caught sea trout	£3,200	Radford et al, 1991; Evans, 1996

\# number of

3.3.4 Riverine bird populations

The species observed during the survey were grey wagtails, pied wagtails, dippers, mallards, grey herons and a kingfisher. Of these, dippers are considered the best indicator of river quality as they feed purely in the river on aquatic invertebrates and occasional fish fry. Species such as the wagtails feed both within the rivers and in the surrounding banks and fields and as such are less limited by the water quality. However increased numbers may still be evident as the aquatic food supplies (invertebrate populations) have been shown to increase at all sample points in section 3.3.2.

Table 3.16 contains the dates of the survey visits to each stretch for the three survey years. It should be noted that there was no access to the polluted control stream (Corrwg Fechan) during the 2001 survey due to access restrictions imposed during the foot and mouth crisis.

This affected the rest of the survey that year which necessitated a much later start date than in 1996 or 2001. Figure 6.1 in R&D Project Record P2-181/PR Appendix 6 outlines the area studied and contains the key for the following result maps. Figures 6.2 to 6.14 in Appendix 6 show the bird sightings and observations for each species during all the visits in 1996. Territories have been identified around these sightings which indicate the birds regularly using the area and possibly breeding. They do not necessarily indicate successful breeding has occurred. Figures 6.15 to 6.30 and 6.31 to 6.51 in Appendix 6 show similar information based on the data from the 2001 and 2002 surveys respectively. Tables 3.17 to 3.19 contain the total number of sightings recorded in 1996, 2001 and 2002 respectively.

Based on the numbers of birds observed over the six visits in 2001 and 2002, compared to 1996, the Pelenna catchment is supporting a much healthier bird population. There were approximately four times as many bird sightings in 2001 and five times as many in 2002. The results are discussed in detail for each individual stretch and species in the following section.

Table 3.16 Bird survey dates

Stretch	Visit	Date 1996	Date 2001	Date 2002
Pelenna	A	01/03/96	03/05/01	13/03/02
(SS79300 94150 to	B	16/03/96	18/05/01	08/04/02
SS79800 96200)	C	10/04/96	05/06/01	03/05/02
	D	25/04/96	12/06/01	14/05/02
	E	16/05/96	21/06/01	05/06/02
	F	17/06/96	27/06/01	16/07/02
Gwenffrwd	A	01/03/96	03/05/01	20/03/02
(SS79800 96200 to	B	16/03/96	18/05/01	12/04/02
SS79650 97550)	C	10/04/96	05/06/01	02/05/02
	D	25/04/96	12/06/01	14/05/02
	E	16/05/96	21/06/01	21/05/02
	F	17/06/96	27/06/01	16/07/02
Blaenpelenna	A	02/03/96	10/05/01	13/03/02
(SS79800 96200 to	B	17/03/96	24/05/01	12/04/02
SS82500 99000)	C	11/04/96	29/05/01	03/05/02
	D	26/04/96	08/06/01	15/05/02
	E	18/05/96	19/06/01	21/05/02
	F	16/06/96	03/07/01	11/07/02
Corrwg Fechan	A	03/03/96	No access	20/03/02
(SS87550 99250 to	B	18/03/96	No access	08/04/02
SS88100 01000)	C	17/04/96	No access	02/05/02
	D	28/04/96	No access	15/05/02
	E	19/05/96	No access	05/06/02
	F	09/06/96	No access	16/07/02

Table 3.17 Total observed bird sightings in 1996

Stretch	Species	VISIT A	B	C	D	E	F	Total sightings
Pelenna	Dipper	-	1	2	-	-	1	4
	Grey Wagtail	1	3	4	-	2	5	15
	Mallard	-	-	-	-	-	3	3
	Grey Heron	-	-	-	-	1	-	1
	Kingfisher	-	1	-	-	-	-	1
Gwenffrwd	Grey Wagtail	1	-	-	-	-	-	1
	Pied Wagtail	-	1	-	-	1	-	2
	Mallard	-	-	-	2	-	-	2
Blaenpelenna	Grey Wagtail	-	3	4	1	1	4	13
	Pied Wagtail	1	-	2	1	1	-	5
	Mallard	-	-	-	-	1	1	2
Total sightings per visit		3	9	12	4	7	14	49
Corrwg Fechan	Grey Wagtail	-	1	1	-	1	-	3
	Pied Wagtail	-	1	1	1	-	1	4
	Grey Heron	-	-	-	1	-		1

Table 3.18 Total observed bird sightings in 2001

Stretch	Species	VISIT A	B	C	D	E	F	Total sightings
Pelenna	Dipper	3	3	7	7	5	2	27
	Grey Wagtail	2	8	8	15	7	6	46
	Pied Wagtail	1	2	-	-	-	-	3
	Mallard	2	-	1	-	-	-	3
	Grey Heron	1	1	1	-	1	-	4
Gwenffrwd	Dipper	-	1	-	1	3	3	8
	Grey Wagtail	-	3	2	2	10	6	23
	Pied Wagtail	-	-	1	-	-	-	1
	Mallard	-	1	2	2	-	1	6
Blaenpelenna	Dipper	2	1	-	1	1	3	8
	Grey Wagtail	7	20	13	5	11	10	66
	Pied Wagtail	-	1	4	3	1	-	9
	Mallard	-	-	2	9	2	-	13
Total sightings per visit		18	41	41	45	41	31	217

Table 3.19 Total observed bird sightings in 2002

Stretch	Species	VISIT A	B	C	D	E	F	Total sightings
Pelenna	Dipper	4	7	4	7	8	3	33
	Grey Wagtail	8	5	8	17	14	1	53
	Pied Wagtail	-	1	2	1	-	-	4
	Mallard	2	6	1	9	1	-	19
	Grey Heron	1	1	-	-	-	1	3
Gwenffrwd	Dipper	1	2	-	1	-	1	5
	Grey Wagtail	-	1	4	2	1	3	11
	Pied Wagtail	-	1	1	3	1	1	7
	Mallard	3	2	10	-	1	-	16
	Grey Heron	1	-	-	-	-	-	1
Blaenpelenna	Dipper	-	1	-	1	-	3	5
	Grey Wagtail	-	3	6	17	13	5	44
	Pied Wagtail	-	2	4	-	1	1	8
	Mallard	4	6	14	6	5	2	37
	Grey Heron	1	-	-	1	-	-	2
Total sightings per visit		25	38	54	65	45	21	248
Corrwg Fechan	Grey Wagtail	6	1	3	3	5	8	26
	Pied Wagtail	-	4	-	-	1	-	5

Nant Gwenffrwd

The Gwenffrwd had a large increase in numbers of grey wagtails. There was one sighting in 1996 compared to 23 individual sightings in 2001 and a decrease to 11 in 2002. From this two territories were identified in 2001 with young wagtails observed in one of them. In 2002 only one territory was identified. It is suggested that this decline in 2002 is possibly due to the overflows from Whitworth A treatment system that occurred during the spring and summer and impacted the invertebrates. This is backed up by the fact that it is the territory downstream of this overflow that has disappeared in 2002 and no grey wagtails were observed in this area that year.

No dippers were observed on the Gwenffrwd in 1996. In the 2001 survey there were 8 individual sightings. The majority of these were around the Phase III wetlands and Whitworth Lagoon. This has been interpreted as being a dipper territory, although if breeding was attempted no young were seen. In 2002 a similar pattern emerged with 5 sightings but none of young.

Two pied wagtail territories were identified above the Whitworth A overflow in 2002. Only one pied wagtail sighting was made on the stretch in 2001, and two had been seen in 1996. Mallard were observed in greater numbers during the 2001 and 2002 surveys and were concentrated around the Phase III and Whitworth Lagoon wetlands. Young were observed on the wetlands during both years. The use of the Phase II and III wetlands by many birds was noticed, especially in 2002. They were most frequented by mallards but also supported groups of grey and pied wagtails.

Numbers of birds have increased markedly on the Gwenffrwd, especially grey wagtails and dippers. Because dippers only feed within the watercourse the establishment of a dipper territory is a definite sign of improved water quality and an increased population of invertebrates. The decrease in grey wagtails in 2002 is potentially linked to the minewater overflows that occurred during the breeding season and therefore lack of food supply in this stretch.

Blaenpelenna

The Blaenpelenna had the largest numbers of grey wagtail sightings of all the stretches in 2001. Approximately five times as many sightings were made in 2001 compared to 1996. Ten territories were identified on the stretch with young birds being observed in five of these. This compares to three territories in the 1996 survey with two observations of young birds in nests. In 2002 the numbers had reduced but were still considerably higher than in 1996. Only six territories were identified in 2002 compared to the ten the previous year. The main decrease in territories was in the section downstream of the minewater and wetlands down to the confluence. As noted previously in section 3.3.3 water quality data does not indicate why this may have occurred.

No dippers were observed on the Blaenpelenna in 1996, but in 2001 eight sightings were made with a territory being identified on the lower part of the stretch. In 2002, 5 sightings were made in the same stretch and were again identified as a territory.

Pied wagtails were observed in slightly higher numbers in 2001 and 2002 compared to 1996. One territory was identified in 2001 and 2002 at the site of the Garth Tonmawr wetlands. More evidence of this territory was found in a later visit in 2001 when four young pied wagtails were observed in and around the wetlands.

Mallard were observed on the stretch in high numbers in 2001 and especially 2002. They were mainly using the Garth Tonmawr wetlands and have been sighted on here during most site visits over the final year of this project.

On the Blaenpelenna the numbers of sightings for all species was much greater in 2001 and 2002 compared to in 1996. Grey wagtails had expanded their numbers of territories and dippers have established a territory in the stretch. Again the presence of dippers is a very good indication of the improvements in water quality and invertebrate populations.

Pelenna

Five grey wagtail territories were identified on this stretch in both 2001 and 2002 with evidence of successful breeding in four of them in 2001 and three in 2002. This is a big improvement over 1996 when only two territories were identified, one of which had young observed in it. Numbers of sightings were greatly increased with 46 in 2001 and 53 in 2002, compared to 15 in 1996.

Three dipper territories were identified in 2001 with young dippers observed in all of them, indicating successful breeding. In 2002 four territories were identified with young birds observed in 2 of them. However one of these territories was at the confluence with the Afan and judging by the size of it is most likely a part of a territory encompassing both rivers. In total 27 individual sightings were made in 2001 and 33 in 2002 compared to only 4 in 1996.

The 1996 sightings were all in separate locations and not thought to indicate evidence of birds holding territory on the Pelenna.

In 2002 a pied wagtail territory was identified at Efail Fach on the Pelenna, no territories were found in 2001 or 1996. Two mallard territories were identified in 2002 and again none were identified in 2001 or 1996. Numbers of Mallard were much higher in 2002 than the previous two survey years. For the other species observed there were none that appeared to be holding territory on the Pelenna. Three grey heron sightings were made in 2002, four in 2001 and one grey heron in 1996. A kingfisher was sighted on this stretch in 1996, however none were seen here in 2001 or 2002.

The major increases observed in terms of numbers of sightings and recognized territories are most likely to be due to the improved water quality. The exception to this is the mallard, which appear to be primarily utilizing the wetland schemes themselves. The establishment of at least three dipper territories in this stretch is the best evidence for this as the dippers food supply is purely aquatic invertebrates and the occasional fish fry. These invertebrate and fry populations have been shown to be growing in numbers following water quality improvement in sections 3.3.2 and 3.3.3.

Corrwg Fechan

The Corrwg Fechan was chosen as a minewater polluted control stream that was of similar size to much of the Pelenna tributaries and was unlikely to change landuse or have the minewaters treated in the foreseeable future. In the 1996 survey a heron was seen once, three grey wagtails were seen but no territories were identified and two pied wagtail territories were observed. During the 2001 survey no access was possible to this river due to the foot and mouth crisis, as the predominant land use was sheep farming. In 2002 no herons were observed and the numbers of pied wagtail seen were very similar with one territory identified. However the numbers of grey wagtails had increased significantly from 3 sightings and no territories in 1996 to 26 sightings and two territories in 2002. One of these territories was at the confluence with the Corrwg that is also minewater impacted and an unpolluted side stream and will therefore allow a greater access to food. The other was mid way up the Corrwg Fechan where the river is culverted for approximately 200m. It would appear from this that grey wagtail numbers might have increased naturally in the region. However as they are not completely reliant on an aquatic foodsource they are not as important an indicator of recovery as the dippers.

4 CONCLUSIONS

4.1 Wetland Treatment Performance

The monitoring of the three wetland treatment systems over the period of this project has demonstrated how effectively they are removing contaminants. The main conclusions concerning the contamination removal rates and discharge loadings are summarized below:

4.1.1 Whitworth No 1

The Whitworth No 1 system has, between 1995 and 2002, been removing 83% of the incoming loading at a rate of removal of 8.1 g/day/m^2. The iron loading of its discharge is 1.5 kg/day. This is the highest removal rate per m^2 but percentage wise the system is removing one of the lower amounts. It is believed that the system performs less well percentage wise due to its design characteristics. Having four cells in parallel rather than sequentially means that the water passing through the system can have a reduced residence time. The system is consistently producing a discharge just below pH 7. Importantly no trends in removal performance have been identified which would indicate that the system should continue to operate efficiently.

Operationally the main problems with this system are the frequent blockages (every couple of months) that occur at the inlets and outlets. The design of both the inlet and outlet structures is over complicated for the task. It slows flows promoting ochre build up which blocks the systems. Once blockages occur the placement of these structures within closed chambers that are confined spaces is an extra problem. Only specifically trained individuals can clean the systems which makes maintenance more difficult to arrange quickly and more costly. A simpler more open inlet distribution and outlet system would be advisable at all treatment sites where possible.

It must be noted that while inherent design features frequently cause sections of the system to block. It is a lack of routine maintenance that is exacerbating these problems. If a routine inspection and maintenance programme were in place then these problems could be cleared quickly. This would ensure the performance of the system could be optimized.

4.1.2 Garth Tonmawr

In this minewater both the iron content and acidity were issues. The system has been effectively removing 95% of the incoming iron loading from 1999 to 2002. It has been coping well with a highly fluctuating minewater loading. The majority of the iron removal has been in the first two cells of the system. Overall the system has been removing iron at a rate of 7.9 g/day/m^2. This is the second highest of all the systems. The mean discharge iron loading has been 2.6 kg/day. This system has no identifiable trends in any of the removal performances measured. This would suggest that the system would continue to operate as it has been to date.

The pH drops within the system as iron oxidation occurs. It drops to a mean of 3.5 prior to flow through the first RAPS cell. These RAPS cells add sufficient alkalinity to buffer the acidity. This occurs despite the fact that a significant proportion of the flow has been bypassing them. By the discharge point the pH is has a mean of 6.8 and 40 mg/l of alkalinity.

The system has been removing acidity at a rate of 15.4 g/day/m^2. This is over one and a half times greater than the next best system Whitworth A.

Operationally this system has had problems with downward flow through the RAPS cells for most of its life to date. However the amount of flow bypassing the system has remained steady and does not appear to be affecting the iron or acidity removal performances. Physically the system has suffered from the collapse of two blocks of brick facing. This has only had cosmetic implications to the system though.

In 2002 the natural wetland prior to the system inlet has become overly full with vegetation and sediment. This is currently causing a minor discharge to the Blaenpelenna of untreated minewater. It should be an easy process to rectify this.

4.1.3 Whitworth A

This minewater contains both high iron and acidity levels. Overall it has been removing 96% of the incoming iron loading from 1998 to 2002. This was the highest percentage removal of all the systems. The mean loading of the discharge to the Gwenffrwd is the lowest of all the treatment systems at 1.1 kg/day. The removal rate has been 5.8 g/day/m^2. This is lower than that seen at some of the other systems. It is believed that this is most likely due to the area of aerobic wetland within the bunds that has no water flowing through it. It must be noted that most (91%) of the iron removal has been occurring on the RAPS system. The implications of this are assessed in section 4.2.2. The RAPS cell has been effectively producing alkalinity to buffer the minewater acidity and the mean discharge has been a pH 7.3. As has been noted on the other systems, the TAPIR trend analysis software has found no significant trends in any of the removal performances. Based on these assessments there are no indications of impending problems with changes in treatment performance.

From the data above it can be seen that this system has been working perhaps the best of all the systems. However this has only been the case when all the minewater has been flowing through it. The design of the inlet structure and piped transfer system to the wetlands has caused frequent problems. Ochre deposits in much of the inlet structure and this then has been leading to blockages of the long transfer pipe. As this is buried it is impossible to inspect the extent of blockages and difficult to clear. There have been two long periods when the system has been partially blocked and overflowing into the Nant Gwenffrwd. The impacts these overflows had were significant and are discussed in more detail elsewhere in this report. It is suggested that open channel structures for conveyance of minewater would be much easier to inspect and maintain and would be less prone to blockages.

It must be noted that again frequent maintenance or importantly the lack of it is an issue at this system. If the inlets were inspected frequently and a more preventative maintenance programme were initiated (i.e. clearing the pipe maybe every 6 months) then this should avoid these long destructive periods of overflows.

4.1.4 Whitworth B and Gwenffrwd

This system has had no contaminated minewater flowing through it for the majority of this study. This was due to a natural underground diversion of the minewaters. Since March 2001 the system has been receiving an input of minewater. The iron loading has been low given the area of treatment. This is reflected by the mean removal rate of the system, which is by far the lowest observed at 0.9 g/day/m^2. The mean percent removal of the incoming iron loading is also the lowest observed at 82%. The mean outlet iron loading has been 2.1 kg/day.

Operationally the system has had no problems. Due to the low iron loading there have not been the rates of ochre buildup seen in some of the other systems. Therefore the problems associated with this buildup have not been observed.

4.2 Internal Processes and Sustainability

The internal processes occurring within the treatment systems have been investigated quite intensely during this project. The majority of work has concentrated upon the Whitworth No 1 and Whitworth A systems. These were investigated primarily because the Whitworth No 1 system is the oldest and the Whitworth A was considered to be of most concern. A summary of the major processes identified, projected lifespan and operational implications of each system is mentioned below.

4.2.1 Whitworth No 1

This system has been removing iron primarily by oxidation processes and precipitation of oxides and hydroxides. The design characteristics of each cell are not promoting the treatment processes they were intended to. Instead substrate type is determining how aerobic the cells are and therefore the metal removal processes operating.

No likely changes in metal removal or pH modifying processes have been identified. It is believed that this system will continue to operate until all the physical space is utilized. This is assuming that the structural integrity of the system does not deteriorate. A tentative suggestion based on observed rates of ochre accumulation is for the system to completely fill by 2022. This makes no allowance for organic matter buildup within the substrates.

To achieve this volume of ochre deposition it would require the amount of surface water to be minimized. If surface water levels are sufficiently higher than the sediment surface the water will flood back into the inlet structures and create blockages. By keeping pipework clear and therefore maintaining a steady flow through the system this should minimize the depth of surface water.

Once ochre and sediment levels have reached the point where surface water frequently impacts the inlet weirs the system will need emptying. Assuming that at this point in time the walls and other physical structures are still in good condition. This process would just necessitate substrate removal and disposal, followed by the addition of fresh substrate.

4.2.2 Whitworth A

The main area of concern in this system was identified as the unplanned volume of iron that is depositing on the surface of the RAPS cell. It is removing 91% of the iron loading and creating a significant depth of ochre (mean 21 cm in 2001). The build up of iron oxide and hydroxide precipitates on the surface of the RAPS cell has the potential to negatively affect its performance and especially lifespan in two main ways:

- The ochre will be compacted as its depth increases, with increased compaction the permeability and hydraulic conductivity should decrease. If the permeability falls to the stage where downward flow through the RAPS is impeded then the cell will overtop through the overflow structures. This will obviously then reduce residence and treatment time in the system as a whole and minimize the alkalinity production from the RAPS.

- If the permeability and hydraulic conductivity do not increase to a point where they impede downward flow. Then the system can continue to accrete ochre. This will continue until the depth is great enough for the water pooled on top of the ochre to overflow the system. This again would reduce the residence and treatment time of the wetlands and reduce alkalinity generation.

Both of these processes were investigated and the time taken for either to occur varied by approximately 4 to 5 years. However they indicate that the physical space on the RAPS cell will fill and water will start bypassing the RAPS somewhere between 2010 to 2016. At or before this point it would be necessary to remove this ochreous material. Suggested methods could include sludge pumping or mechanical digging. Both would create differing operational issues as mentioned in section 3.2.2.

It is not thought that the substrate would need removing at this time. No calculations have been made for organic carbon use but the lifespan of the limestone has been predicted. The date suggested for complete limestone usage is 2034. The confidence intervals however, were high, and give a date range of 2025 to 2048. This is the date for complete limestone use and alkalinity generation would decrease significantly prior to this date. It is acknowledged that the large range of these dates is not beneficial for decision making concerning when and how to maintain the systems. However they are an indication that the limestone should last its predicted 30-year lifespan.

4.2.3 Garth Tonmawr

Little work has been specifically targeted at the processes occurring in this system. The depth of water creates a number of operational problems to substrate sampling. No trends in removal performance have been observed and the system appears to be operating well despite the partial RAPS bypass. It is therefore difficult to suggest a potential lifespan at this stage. There appears to be no reason why it should not operate for its design lifespan provided regular maintenance in undertaken.

4.2.4 Whitworth B and Gwenffrwd

This set of wetlands has never been studied in any detail. For the majority of its operational life to date it has been receiving uncontaminated flow. Following engineering works in March 2001 the Whitworth B and Gwenffrwd minewaters have both been flowing through this system. The loading of iron has been very low and there is not an obvious ochre layer on the RAPS. It is difficult to suggest when any problems may occur with this system but it is likely that it will continue to operate efficiently longer than Whitworth A, the other treatment leg of phase III, due to the relatively low iron loading.

4.3 Environmental Benefits

4.3.1 Water quality improvements in the rivers

Following minewater treatment all sample points downstream of the minewaters on the rivers have seen dissolved iron levels drop below the EQS of 1 mg/l. The pH has also risen at all these points. This has therefore proved that one of the major aims of the treatment project is achievable. However the overflows from the Whitworth A system has increased the levels of dissolved iron to above the EQS for significant periods. The impacts of these overflows have been noticeable in water quality data at Efail Fach 2.8 km downstream. The main periods of overflow have been through the winter of 1999 to spring 2000 and spring to summer 2002. The pH of the river has been seen to drop during these prolonged overflow periods.

An unforeseen benefit of the RAPS cells has been to buffer the pH of the Nant Gwenffrwd. Downstream of the discharges from the phase III treatment schemes the pH has been noticeably higher than that found immediately above the discharges.

The visual impact assessment key has been used to assess changes in visual quality over the years. This has provided a good indication of how the visual appearance of the rivers has recovered. It also demonstrates how the overflows to the Nant Gwenffrwd have significantly affected the intensity of staining. The VIS key does provide a good indication of how the visual impact of a minewater impacted site changes over time. This is of use in a case such as the Pelenna where minewaters have been treated. It can also be used on untreated minewaters to provide an assessment of how the impact in the receiving watercourse may have changed over time.

There appears to be some unexplained variations in the scores attributed to some of the Pelenna sites. Modifying the assessment sheet questions should hopefully reduce the variations in the method observed so far. It is thought that a proportion of the variation in results may be attributed to variation in experience and training of the samplers concerned. Personnel who were less used to seeing minewater impacted streams had a tendency to score sites at a higher impact level. The key has therefore been modified slightly to reflect this and has photographs of comparison sites. These can then be laminated and taken out on site visits, which will be of use to those with less experience of the severity of some minewater visual impacts. The modified key is presented in R&D Project Record P2-181/PR Appendix 7.

4.3.2 Macroinvertebrate populations

The macroinvertebrate populations have demonstrated population recovery at all the sites downstream of the minewaters following treatment. The recovery and subsequent declines observed are related to the changes in dissolved iron concentrations brought about by the treatment systems. The effects observed on each stretch are concluded below.

Afan and Fforch-Dwm

These are the control sites for the catchment. Looking at total and EPT abundance and the results of TWINPSAN analysis there were no obvious trends at these sites over time. The Fforch-Dwm is the control for the upper catchment whose population the smaller streams may recover too. The Afan sites represent a bigger deeper river and would therefore indicate a population structure which sites lower on the Pelenna may reach.

Nant Gwenffrwd

The sites downstream of the minewaters increased both total and EPT abundance the year after minewater treatment. The results of the TWINSPAN analysis demonstrated that the diversification of the populations was taking at least an extra year. This recovery was much more obvious in the summer samples as they were less impacted by the natural acidification of the catchment. It is thought that as water quality improved the existing population expanded to make use of the improved conditions. Then new species moved in by downstream drift from the non-minewater impacted site upstream. This population was however impoverished due to episodic acidification impacts. Further population diversity will therefore rely on the slower process of species moving up the catchment.

The impacts of the overflows in 2000 and 2002 were very noticeable. This was especially the case with the effect of the 2000 overflow on the spring populations. Both the total and EPT abundance's decrease with a reduction in Ephemeroptera populations being especially obvious. The sample sites below the overflow on the Nant Gwenffrwd all dropped back to the lowest quality TWINSPAN grouping. The effects were not so obvious on the summer population in 2000 as the blockages had been cleared in late spring. However its negative effect was still noticeable especially within the Ephemeroptera population.

Blaenpelenna

On the Blaenpelenna the abundance and EPT abundance increased the year following minewater treatment. The TWINSPAN analysis has demonstrated that it is taking a couple of years to move up through the TWINSPAN groupings. This demonstrates that the diversity of the population is taking longer to develop. It is likely that this is occurring for a similar reason to that found on the Nant Gwenffrwd. The upstream macroinvertebrate population is impoverished and will reduce the degree of diversification available through downstream drift. Diversification by recolonisation from downstream will therefore be taking longer.

Pelenna

The two sites on the Pelenna appear to be fairly variable in terms of population and EPT abundances in both spring and summer over time. However there are improvements in the years following minewater treatment. These are most noticeable in the changes in TWINSPAN classification. P2 (Efail Fach) has recovered to group 1 membership in both spring and summer. The impacts of the overflows on the Nant Gwenffrwd are not obviously apparent at this site. At P3 above the confluence with the Afan the Pelenna has improved diversity and population numbers in spring with little change observed in the summer populations.

Macroinvertebrate response to water quality

The data gathered from the Nant Gwenffrwd has demonstrated how increases in dissolved iron can impact the populations. This is especially evident in the minewater intolerant orders such as Ephemeroptera. The TWINSPAN groupings can be related to water quality variables. These indicate that it is the dissolved iron that governs population structure in the two lower quality groupings identified. As the iron level decreases the pH of the water at the sample site has the largest control on the population.

4.3.3 Fisheries populations

In general the populations of trout fry and parr have increased following minewater treatment at all sites. On the Blaenpelenna below Garth Tonmawr there is an obvious increase in 1999 especially in the fry population. Further down on the Blaenpelenna and Pelenna the populations have increased but were not so impacted prior to minewater treatment.

On the Nant Gwenffrwd fry were absent from all the sites prior to treatment. Parr were only occasionally found. The year following minewater treatment the fry immediately recovered at all points followed by a recovery of parr the next year. Fish had been previously observed attempting to spawn on the Nant Gwenffrwd but the eggs and fry had not survived. With the improved water quality the next years spawning was successful. The increase in parr the following year demonstrated that the fry matured through to a parr population.

The impacts the minewater overflows on the Nant Gwenffrwd in 2000 and 2002 had on the fish is very obvious. Fry numbers at the two sites immediately downstream from the overflows were severely reduced both years. This indicates the effects of the elevated dissolved iron concentrations and ochre smothering of the gravels. Parr numbers were not significantly affected by the overflows. When the overflows were stopped and the dissolved iron levels returned below the EQS of 1 mg/l the fry populations immediately recovered again the following year.

As a fisheries resource the economic benefits to the fishery as a result of minewater treatment have been calculated to be in the order of £181 000. This figure is as previously noted not an absolute figure but more a guide of the order of magnitude of the benefits. Obviously if the rivers were all to be allowed to return to pre treatment contamination levels then all the financial benefits observed will be removed.

4.3.4 Riverine bird populations

Based on the numbers of birds observed over the visits in 2001 and 2002, compared to 1996, the Pelenna catchment is supporting a much healthier bird population. There were approximately four times as many bird sightings in 2001 and five times as many in 2002. There has especially been a considerable increase in numbers of Grey Wagtails. This increase has been demonstrated on all survey sections.

The establishment of five dipper territories and evidence of breeding and young birds is a major indicator of the improvements to the water quality and aquatic ecology following treatment of the minewater discharges. Prior to treatment the catchment was not supporting any dipper territories. Roberts (1996) suggested that in his opinion the catchment could support five dipper territories if the water quality and food supply was improved and it now appears to be doing so.

The 2002 survey on the Gwenffrwd indicates that the overflow from Whitworth A was potentially having an impact on the bird populations. A grey wagtail territory that existed downstream of the treatment systems in 2001 was not found in 2002. However while it is suggested that water quality changes and reduced food supply could limit the birds use of an area it cannot be proven that this was an impact of the Whitworth A overflow.

4.4 General Conclusion

This research has shown that the wetland treatment systems were effectively removing the majority of their contaminant loadings. The river water quality has recovered to be below the target EQS qualities. Following these improvements in water quality the macroinvertebrate abundance has increased the next year. Diversification of the macroinvertebrate populations was developing a couple of years later as there was limited downstream drift from the impoverished population upstream. Fishery populations have recovered in the year following minewater treatment. Riverine bird species populations have demonstrated recovery over time.

These recoveries have been demonstrated below all the former minewater discharges. However problems with overflows have had substantial impacts on the recovery of all these populations. The reasons for these problems are all essentially linked to a lack of routine preventative maintenance on the systems. With an effective maintenance programme it would be expected that the improvements seen in the catchment could be continued.

No potential problems with the systems failing in the short term have been identified. The Whitworth A RAPS cell is the one that is anticipated to require attention first. However this may just involve removing the ochre layer from the surface.

This study has determined that passive constructed wetland treatment schemes can be a low cost and effective method for treating minewater discharges. They do lead to significant and sustainable improvements in the aquatic ecology. It has also demonstrated how important routine maintenance is to keep schemes operating at an optimum level.

5 RECOMMENDATIONS

5.1 The Pelenna Wetlands

The main recommendation for the Pelenna systems is for a thorough routine inspection and maintenance programme to be instigated. This should concentrate on keeping pipework and distribution systems free flowing. This will ensure that the environmental benefits and longevity of the systems are maximized. It is suggested that inspection is undertaken monthly.

Monitoring the ochre buildup on the Whitworth A RAPS should be included in these inspections. This is the first area of the systems where problems involving substantial site works may be expected.

Monitoring of the inlet and outlet loading of the system should be continued. This will provide useful data to alert the relevant authorities if the system performances start to drop. In order to provide the optimal conditions for the continued aquatic recovery in the rivers it would be necessary to continue the average treatment performances seen to date.

5.2 Future R&D Work on the Pelenna

The data gathered and presented in this report have provided a good insight into the performance of the Pelenna wetlands. The environmental recovery potential has been demonstrated. The work has also shown that overflows from the systems can have big impacts. At this stage it is not considered necessary to continue the full environmental monitoring every year. It would be suggested that a repeat of at least the invertebrate and fisheries summer surveys is considered in 3 to 5 years time to assess the continued effect over the lifespan of the wetlands.

The monthly water quality survey is much less time consuming. In the period following the end of the data collection for this project, the monitoring has been substantially reduced. The data gathered now is the minimum possible to produce an accurate picture of continued performance and river water quality. It is suggested that this be carried on each year in its current reduced form. From this it will then be possible to infer broadly the condition of the aquatic ecology of the rivers.

If any further evidence of the damaging effects of the minewater overflows is needed the summer 2002 samples should be sorted and identified. These were taken and preserved but not assessed due to time constraints within the project. The resources would have to be made available to do this.

5.3 Minewater Treatment in General

The main recommendation that is relevant to future minewater schemes is for the designs to keep distribution structures as simple as possible. These should be open channels where possible to allow easy inspection and maintenance.

When schemes are designed the aim should be to get dissolved iron in the river to be continuously below 1 mg/l and therefore meet the EQS targets. It has been shown in this study how well the aquatic ecology can recover when dissolved iron is reduced below this level. Obviously if other contaminants are evident then these will also require reductions to achieve the appropriate standards.

If aquatic ecosystem recovery is a specific aim of a treatment project then the supply of recolonisation species should be investigated. Having a healthy invertebrate population upstream will speed up recovery of the invertebrates. The existence of healthy fish populations either nearby or attempting unsuccessfully to spawn in the area will maximize the speed at which the fish recover. Lack of these populations will either slow recovery or necessitate the introduction of species.

An important characteristic of constructed wetlands is that they are low maintenance, not maintenance free. When projects are initiated then long term maintenance funding should be identified.

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