An overview of diffuse pollution inputs and their impacts upon organisms in the River Almond, Scotland.

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March 2017

A thesis submitted in partial fulfilment of the requirements of Edinburgh Napier University, for the award of Master by Research

Contents

Acknowledgments

I would like to thank my project supervisors, Dr Rob Briers and Dr Linda Gilpin for their advice, expertise and guidance throughout this project.

I thank Tommy McDermott, Project Officer at the River Forth Fisheries Trust, for support, training and invaluable experience enabling the completion of this project.

Special thanks to the River Forth Fisheries Trust for the opportunity to collaborate and continue to contribute to the restoration and assessment of the River Almond.

I am thankful to Edinburgh Napier University, the Post-Graduate Research Student Group and Technical Staff for providing facilities and ongoing support throughout.

List of Abbreviations

— ASPT: Average Score Per Taxon.
— BACI: Before-After Control-Impact.
— BOD: Biological Oxygen Demand.
— BMWP: Biological Monitoring Working Party.
— CAR: Controlled Activities Regulations.
— CCI: Community Conservation Index.
— CORINE: Coordination of Information on the Environment.
— CSO: Controlled Sewage Outlet.
— DEFRA: Department for Environment, Food and Rural Affairs.
— DEM: Digital Elevation Model.
— DO: Dissolved Oxygen.
— DTM: Digital Terrain Model.
— EQR: Ecological Quality Ratio.
— EU: European Union.
— FS: Flow Score.
— FSSR: Fine Sediment Sensitivity Ratings.
— FWMC: Flow Weight Mean Concentration.
— GES: Good Ecological Status.
— GIS: Geographical Information System.
— ID: Identification.
— IMS: Industrial Methylated Spirits.
— LIFE: Lotic-invertebrate Index for Flow Evaluation.
— N: Nitrogen.
— NPK: Nitrogen, Phosphorus and Potassium.
— NGR: National Grid Reference.
— N-TAXA: Number of Taxa.
— OS: Ordnance Survey.
— P: Phosphorus.
— PSI: Proportion of Sediment Sensitive Invertebrates.
— RICT: River Invertebrate Classification Tool.
— RIVPACS: River Invertebrate Prediction and Classification System.
— RULSE: Revised Universal Soil Loss Equation.
— SEPA: Scottish Environment Protection Agency.
— SWMI: Significant Water Management Issues.
— TP: Total Phosphorus.
— USLE: Universal Soil Loss Equation.
— WFD: Water Framework Directive.
— WHPT: Whalley, Hawkes, Paisley and Trigg.

List of Tables

List of Figures

displayed above with error bars displaying the standard deviations of the sites. Blue bars represent riffle values and orange bars represent pool values.

- Figure 24: The Ecological Quality Ratio BMWP, ASPT, N-Taxa, WHPT ASPT and WHPT N-taxa scores for high phosphorus V's low phosphorus riffle sites are displayed above with error bars displaying the standard deviations of the sites. Blue bars represent high P input sites and orange bars represent low P input sites.
- Figure 25: LIFE scores for sites that are under high phosphorus inputs with differing flow types, riffle and pool, with standard deviations displayed by error bars. Riffle LIFE score is depicted by the blue bar and pool score by the green bar.
- Figure 26: LIFE Ecological Quality Ratio scores for riffle and pools at high P input sites with standard deviations at error bars. Blue bar represents riffle Ecological Quality Ratio values and green represents pool values.
- Figure 27: PSI percentage high phosphorus input riffle and pool sites, high and low phosphorus input sites, with standard deviations displayed by error bars. Riffle sites are represented by the blue bar, pool sites by the orange bar, high P sites represented by the green bar and low P input sites are represented by the yellow bar.
- Figure 28: Ecological Quality Ratio PSI scores for high phosphorus input riffle and pool sites and high and low phosphorus input sites. Error bars show the standard deviation. Riffle sites are represented by the blue bar, pool sites by the orange bar, high P sites represented by the green bar and low P input sites are represented by the yellow bar.
- Figure 29: Boxplots showing the median value, 25th quartile and 75th quartile of log nitrate values from high phosphorus (Green) and low phosphorus (Blue) input sample sites from the main stem of the river Almond. Error bars are the lowest and highest values recorded from nitrate analysis.

Abstract
The Water Framework Directive (WFD), published by the European Union in 2000, implemented by SEPA in Scotland, has been the main force behind changes to river environments within Scotland. The main focus is on improving water quality by not just considering the chemical characteristics of water but also the ecological characteristics of the ecosystems. Diffuse pollution, in the form of nitrogen, phosphorus and sediment inputs, has a major influence on river water quality. Catchment land use is seen as a key driver of diffuse pollution, with an estimated 75% of Scotland's diffuse pollution originating from agricultural inputs. The aim of this study was to assess the extent of diffuse pollution inputs and their effects on the River Almond, Scotland. This was achieved through a combination of catchment-scale assessment of diffuse inputs based on land use characteristics, a walkover survey quantifying key in-channel features, modification, pollution points and flow types and sampling and evaluation of macroinvertebrate communities at contrasting sites. The length of the river was further broken down into 12 sections of equal size and different riparian corridor levels, 25m, 50m, 100m, 200m and 500m to further analyse diffuse pollution impact at a local scale. Phosphorus data was used as the primary indicator of a 'high' or 'low' nutrient load as nitrogen coefficients lacked the specificity required to determine this accurately. To try and quantify the levels of diffuse pollution present within the Almond catchment, a walkover survey was conducted along the main stem inventorying key in-channel features, modification, pollution points and flow types. Variation in perceived pollution loads and habitat characteristics facilitated the selection of kick sampling sites for macroinvertebrate collection in order to assess the impact that diffuse pollution has upon habitat and water quality at a local level. Eighteen samples were collected in total in areas along the main stem consisting of 12 samples at high P input sites, 6 at riffle sites and 6 at pool sites, whilst 6 were sampled at low P input riffle sites. The aim of this was to assess how different flow types respond to diffuse pollution and whether differences between macroinvertebrate communities were evident between high and low P input sites. Analysis primarily consisted of the calculation of indices such as the Lotic-Invertebrate Index for Flow Evaluation (LIFE), Percentage of Sediment-sensitive Invertebrates (PSI), Biological Monitoring

and Working Party (BMWP), Average Score Per Taxon (ASPT), Number of Taxa (N-Taxa) and abundance weighted metrics for WHPT ASPT and N-taxa where once calculated were compared to predicted values provided by River Invertebrate Classification Tool (RICT) – this allowed for Ecological Quality Ratings (EQR) to be calculated to assess water quality at these sites.

Non-irrigated agricultural land was identified as the land use that contributes most significantly to overall predicted nutrient loading to the river. There were no clear relationships between nutrient loadings calculated based on land use characteristics and the quantity or extent of diffuse pollution sources recorded in the walkover survey. This would tend to indicate that catchment-scale analysis and walkover surveys provide complementary but different information regarding diffuse pollution inputs; a combination of both types of information is likely to be useful in assessing potential impacts.

Non-significant results were found when looking for relationships between the number of diffuse points per section, the levels of erosion and the levels of poaching in relation to the P loading for that section at any riparian corridor level.

Evaluation of the invertebrate communities between areas with high and low diffuse nutrient loading using Ecological Quality Ratios for several relevant biotic indices found no difference in quality. However, comparison of different habitat types (pools and riffles) within high nutrient loading areas showed significant differences in a number of metrics, with riffles having higher quality. The River Almond is subject to a range of other influences, notably point source pollution and hydromorphological modification which may obscure any response to variation in diffuse inputs. However, the observed differences between pools and riffles suggest that variation in local physical habitat conditions may modify the extent to which any impacts of diffuse pollution are evident.

1.0 Introduction

1.1 Diffuse Pollution

Diffuse pollution is classified as pollution that derives from land-use activities that are dispersed widely across a catchment which causes or contribute to water quality problems (Yuan et al. 2007). The main sources of diffuse pollution from agriculture are related to fertiliser application, organic matter from animal waste and soil erosion which can have a cumulative degrading effect upon water bodies, ultimately leading to issues such as eutrophication and sedimentation. Urban areas further contribute to diffuse pollution by a variety of sources such as garden fertiliser, animal and food waste, construction works and historic mining activities amongst other things. Run off from impermeable urban areas is considered to be a large facilitator in the transport urban diffuse inputs to water bodies causing degradation by increasing the number and intensity of flood events and reducing groundwater permeability, whilst often a direct result of these patterns, increasing stream bed erosion can occur (Hatt et al. 2004). The availability of nutrients, in particular phosphorus, is linked closely to sediment inputs through binding to soil particles, enabling sediment to act as a vector of movement for nutrients to enter freshwater environments (Ruiz-fernández et al. 2002). Urbanised areas contribute to the degradation of water bodies when contaminants are spilled, such as oil and other hydrocarbons, the treatment of nuisance plants via herbicide spraying and the bypassing of water treatment facilities by washing of water and contaminants into waterbodies (Scottish Environment Protection Agency. 2006).

Nutrient enrichment caused by point source pollution has proven easier to control, regulate and reduce due to the method of access to water bodies, diffuse inputs however are more difficult to regulate. An average of 60% of the total nitrogen (N) pollution present in European rivers is derived from diffuse sources (Romero et al. 2016). As a result of concerns over increased levels of nutrients entering water bodies in the latter part of the last century, the European Commission implemented the Nitrates Directive (91/676/EEC) to reduce diffuse pollution from agriculture along with the Waste Water

Treatment Directive with the specific target of reducing nutrients in urban areas.

1.2 Nutrients

Nitrogen and Phosphorus (P) inputs are the root causes of eutrophication in water bodies. Concentrations of N and P are strongly linked to the land use employed within the catchment (Viney et al. 2000). Intensively farmed areas lead to more severe problems such as eutrophication, caused by excess nutrients, which often occur within the lowland reaches of a river catchment where flatter areas allow for the growth and cultivation of crops and rearing of livestock (Wade et al. 2001). Nutrient release is affected by amount, type, method and timing of nutrient application, along with soil erosion and sediment transport (Arheimer & Lidén 2000). Nutrients are mostly contained within the uppermost few centimetres, closest to shallow rooted vegetation, meaning surface water can interact with this layer and dissolve stored nutrients making them more soluble and transportable.

1.2.1 Phosphorus

Like nitrogen, phosphorous helps to facilitate the eutrophication of waterways and is often applied in excess to agricultural land in the form of fertilisers so a greater amount is readily available for plant assimilation (Bowes et al. 2009) but this, of course, allows for leaching to occur into river environments. A positive correlation exists between the percentage of arable land in a catchment and the total level of phosphorus present in rivers (Kronvang et al. 2003). As erosion rates increase so do the levels of particulate phosphorus present; this was found for both unfertilised grasslands and for conventionally fertilised tilled wheat fields (Sharpley et al. 1994). The long-term use of animal manures is a major contributor to phosphorus leaching and to control this, animal waste management should be controlled and managed (Sims et al. 1998). Phosphorus from manures often binds to the most highly erodible soil components such as clays, organic matter and other components such as oxides of iron and aluminium, which further facilitates the movement of phosphorus into stream water via run off and leaching. Interestingly, Arheimer & Lidén (2000) make the point that previous studies suggest only near-stream land should factor in the decision-making process when establishing relationships between catchment characteristics and water quality. If land surrounding rivers was managed correctly then riparian wet low lying areas can contribute to denitrification by acting as an effective filter to reduce nitrogen and phosphorus run off to aquatic areas (Grant et al. 1996). However, if land use within the riparian corridor is designated as arable, with a high concentration of animal manure inputs, the leaching of phosphorus to water bodies results in an increased threat of eutrophication.

Human influence on waterbodies is well documented, with a plethora of mitigation techniques being discussed, developed and implemented. Wang (2015) approached the issues of water quality management and land use planning of a watershed in a more empirical approach which required water quality to be measured by: land uses, spatial statistics, biological and habitat indicators as well as water chemistry. Index values for biotic integrity and invertebrate community were significantly low in areas with high human impact as well as habitat quality ratings also being significantly low. Wang (2015) further found that biologically based indicators were negatively related to urban land use but positively related to agricultural land uses. Subsequently, from this Wang (2015) deduced that the biotic index was more sensitive to changes in land use composition and riparian habitat quality than the invertebrate community index. The impact of land uses on the water quality of the study river and watershed, at a catchment and local scale, suggest that land use and water management planning should be conducted with consideration for one another in an attempt to improve the overall water quality of the catchment (Wang 2015).

Differences in diffuse pollution level and water quality at local and catchment level scale levels have been investigated within the literature with varying results. Sliva & Williams (2001) used a similar GIS (Geographical Information System) based approach to that used by Wang (2015) and compared the influence of diffuse pollution on water quality at the 100m local scale and at a catchment scale. It was found that urban land use types had the greatest influence on water quality within local scales whilst forested and arable land had more of an influence on water quality at a catchment scale. These land use types were found to be negatively correlated to water quality meaning these land use types were predictors of diffuse pollution and degradation. Whilst Johnson et al. (1997) agreed with Williams (2001) that crop agriculture and urban land use were the largest contributing factors to diffuse pollution and a degraded water quality in streams, conversely, Johnson et al. (1997) found that riparian buffers of 100m were sufficient for predicting water quality issues rather than total catchment.

Dissolved inorganic phosphorus has strong sediment interactions but little is known about how river stream-beds react when high levels of phosphorus are present within the overlying waterbody and whether interactions between the two act as short term or long term sinks (House & Denison 1998). Deeper and slower moving sections of a river, where much finer stream bed composition occurs, had higher concentrations of phosphorus than other larger sediments. There is a systematic increase in the levels of finer material between winter and summer months which indicates that low flow periods lead to smaller particle accumulation (House & Denison 1998). Diffuse sources are flow dependent and occur during periods of high flow caused by levels of high precipitation. It is generally assumed that phosphorus is regarded as being a point source input and the difference in mode of phosphorus delivery to water bodies leads to differences in phosphorus concentrations (Bowes et al. 2008). In rivers where point source pollution dominates, there is usually an intermittent supply of phosphorus inputs meaning that concentrations will be highest when these conditions persist. However, during high flow conditions, the concentration of phosphorus in water bodies will decrease due to the dilution effect from excess water. Conversely, in rivers that are influenced by phosphorus from diffuse sources, an increase in phosphorus load and concentration will occur during periods of high flow.

As suggested by several studies, it may take several years before the effects of management practices of phosphorus can translate into measurable improvements in water quality (Collins & McGonigle 2008). This time lag reflects the complexity of P accumulation, in high levels within soil and sediments, as well as the complex ways in which P is redistributed through catchments via storage and remobilisation between primary sources and catchment outlets. Diffuse pollution loads have multiple methods of entry to water bodies and as a result, are often underestimated but contribute a significant fraction of diffuse pollution load in a river (Arheimer et al. 2000). Agriculture is agreed upon, within the literature, as the highest contributor due to the usually high percentage of land cover and the extensive use of fertilisers (Collins et al. 2007) but is difficult to evaluate as it is strongly influenced by climate, soil type and geomorphology. As a result, integrated mathematical modelling is employed and used to identify and calculate pollution sources and impact of both diffuse and point source pollution (Candela et al. 2009). To quantify the total amount of pollution and effects upon water bodies, Candella et al. (2009) performed integrated modelling on point source and diffuse pollution sources in the Nocella basin, Italy. In total, pollution mass was 79.5 tonnes and of this, over 60% of nitrogen and phosphorus inputs were contributed by the natural catchment (rural inputs) whilst urban inputs, despite having a low total basin area (only 2%) provided more than 35% of total pollution load.

1.2.2 Nitrogen

Traditionally nitrogen is most closely associated with agricultural processes involving ammonia compounds and nitrate fertilisers that are applied to crops and arable land. Land based sources of nitrogen contribute a large proportion of the total N load and are most closely related to the application of NPK fertilisers and coupled with this, phosphorus inputs too (Arheimer & Lidén 2000). Inorganic nitrogen was the only nitrogenous species to directly be associated or correlate to arable land suggesting that either nitrogenous species are transformed into inorganic nitrogen or that arable land only inputs this form (Arheimer & Lidén 2000). Because of this, Arheimer & Lidén (2000) suggested nutrient transport is linked to sediment process rather than agricultural field-erosion processes.

Nitrogen can access soil from nitrogen fixation via legume pastures, plant residue and nitrogen fertiliser, as stated in the papers above, but nitrogen leaching frequently occurs from grazing animal urine patches as well as or instead of nitrogen fertilisers (Drewry et al. 2006). In forested catchments, the most dominant form of nitrogen is particulate and dissolved organic nitrogen and that generally, lower nutrient generation occurred from forests when compared to agriculture. In forested areas with high nitrogen deposition rates, the dominant species is nitrate and in some instances, can cause increased nitrate leaching from forested areas (Drewry et al. 2006; Kortelainen et al. 1997). Most leaching events occur in spring-time, which coincided with high water flows, similar to autumn, winter and spring leaching periods of agricultural practices. Spring leaching concentrations from forested catchments in total released 50% of the nitrogen present within the forested areas; similar figures were also found in agricultural areas at a similar time-period within Finland (Kortelainen et al. 1997). Leaching from forested areas was lower per unit surface area than agricultural land but total leaching from Finnish forestry land is comparable or exceeds rates experienced in agricultural areas. Conversely, Johnes & Hodgkinson (1998) proposed that, while nitrate is not considered to be easily retained within soil so therefore has a high leaching potential, phosphorus, on the other hand, is assimilated strongly within soil layers and is subjected to negligible amounts of leaching. This has not been fully explored in catchments with very intensively farmed areas but would suggest that phosphorus poses less risk to water bodies and having a reduced impact on eutrophication unless soil erosion is significant within the catchment.

Goodale et al. (2009) characterised seasonal and spatial patterns of stream nitrogen loss from small forested areas within catchments in the Upper Susquehanna River Basin. Nitrogen losses were expected to follow trends already previously described, where nitrogen would enter riverine systems during winter periods where high flow occurs. However, the results showed that summer months, typically associated with low flows and low nitrogenous inputs, had in fact higher concentrations of nitrogen species (Goodale et al. 2009). They expected lower concentrations in summer months as this is when growing events usually occur so the demand for nitrogen is high and the reduction in flow usually drives this. Mechanisms such as high levels of evapotranspiration and summer snowmelt may allow the release of nutrient rich groundwater, weathering of geologic nitrogen and summer increases in soil nitrification within soils

Atmospheric decomposition is thought to contribute up to a quarter of nitrogen input to a river catchment, where forests uptake a large proportion of this. Creed & Band (1998) found that the atmospheric deposition of nitrogen to forested areas increased export of nitrogen to surface waters. It was previously assumed that nitrogen cycled within the forested areas with limited or no export to surface waters, which has implications on a land use approach to management as forested areas and the planting of trees is thought to reduce or slow down the amount of nutrients released into a river environment. It is worth noting that forests can act as nitrogen sinks but once the forest matures and the demand for nitrogen inputs decrease, an increase in nitrogen export can occur.

1.3 Sediment

Sediment that persists within rivers often originates from the top soil of arable lands, grazing grassland for cattle or mining activity and urban areas contribute sediment from practices such as early construction stages and solids from waste water treatment plants and road contaminants (Merritt et al. 2003). Nutrients often persist in rivers as a particulate form with P having a great affinity with soil and sediment particles (Mainstone & Parr 2002) - this attachment to sediment and particles often occurs during fertiliser application to arable ground. Erosion of sediment such as top soils is prevalent within river catchments and results in sediment entering streams and rivers and this is a major conduit for nutrients to enter river systems. Erosion of sediment occurs in three main stages, detachment, transport and deposition, and it was originally thought that erosion was driven by one main force, shear stress-causing raindrop impact, mentioned in Merritt et al. (2003) but detailed by Hudson et al. (1975). A second mechanism, overland flow, is also responsible for the erosion of sediment into rivers (Merritt et al. 2003). Sediment is transported into rivers in two main forms: solid material and within a solution and can be further sub-divided into organic and non-organic forms. Solid forms of sediment are important for biogeochemical cycling as they are chemically active and are thus able to transport nutrients and other organic and non-organic material. However, the sediment and bound nutrients are transported in small groups, flocs or aggregated particles (Merritt et al. 2003).

Research has indicated that sediment fluxes and sediment inputs to river systems are increasing in general, and are caused by human activity such as deforestation, agriculture, construction and mining activities (Foster & Lees 1999; Merritt et al. 2003; Owens et al. 2005). Although human activity has resulted in an increase in sediment being deposited into river systems, it is common place for the average sediment levels within the water column to remain unchanged in lowland reaches of the system mainly due to changes in storage within the river and the construction of impoundments such as dams holding sediment in place.

An increase in fine sediment inputs to rivers can cause problems with turbidity and sedimentation whilst also changing river morphology, behaviour and navigation - associated effects could be detrimental to salmon spawning gravels and in stream habitats (Owens et al. 2005). As a result, fine sediment is generally renowned as having a negative impact on the overall habitat of a river, however a broader range of sediment, in terms of size, can improve already existing habitat whilst also creating new habitat (Kondolf 1997). Sediment quality is an issue that resides primarily around the fine grain sediment fraction, <63µm fraction in particular, as this fraction has an affinity to nutrients and often acts as a vector for the movement into water bodies thus causing increases in not only turbidity but also eutrophication (Owens et al. 2005). Sediment, in general, is required by water bodies and is utilised primarily to create and stabilise new and existing habitat but the negative effects are caused primarily by the fine sediment fraction. The effects of fine sediment on food webs are known to be detrimental to macroinvertebrates and cause severe disruption to communities (Yule et al. 2010). There is a direct link to suspended sediments causing most disruption with activities such as mining, logging and construction work causing bank destabilisation and facilitating the movement of sediment. Sediments of urban streams often act as sinks that store sediment, heavy metals and other pollutants. Riparian, as well as instream habitats and water quality usually deteriorate when urbanisation of a catchment occurs, therefore it becomes difficult for agencies responsible for water quality to decide upon which mitigation to employ. Benthic dwelling macroinvertebrates, as stated, are utilised as bio indicators due to the close affiliation with sediments allowing for information to be gathered regarding interactions between sediment and sensitive species (Pettigrove & Hoffmann 2005). A reduction in macroinvertebrate diversity is noted following increased sediment exposure indicating that sediment quality, type and volume has a strong influence on community structure, diversity and general health of populations (Pettigrove & Hoffman 2005).

The movement of sediment throughout a river catchment is often complicated by sediment being stored between the high and low reaches which has implications with regards to nutrients that are often sediment bound. Flow causes remobilisation of these sediments and associated constituents which ultimately can lead to a reduction in water quality, therefore causing a severe impact on any organisms present. Due to the relationship between sediment and nutrients, it is thought that if sediment is deposited and stored further upstream it is more beneficial overall as there is less potential for large-scale impacts occurring from sediment bound nutrients (Walling et al. 1999). On the other hand, the continued storage of sediment bound nutrients has the potential to cause widespread ecological damage, as a large-scale release to river environments would be detrimental. Floodplains play an important role in storing phosphorus that is associated with sediment due to the mobility of phosphorus when bound to sediment. Particulate-P accounts for a large proportion of phosphorus within river systems and mechanisms such as overbank sedimentation on floodplains act as vital sinks. As phosphorus is held within sediment on flood plains, changes in phosphorus content can occur as it is metabolised by plants or mobilised by water, but as phosphorus is usually readily attached to sediment, the most likely result is that deposited sediment will be retained within the flood plain. Agricultural areas have more sediment retained phosphorus than pastures, but an increase in phosphorus levels deposited within flood plans and sinks is increasing (Walling 1999). Sediment storage within a drainage basin or river catchment means that understanding the links between land use, erosion and sediment yield, overall sediment budget and sources of sediment should be considered when looking at a catchment rather than solely sediment outputs.

1.4 Land use

Land use and diffuse pollution are closely inter-related and it is often regarded that diffuse pollution originates from the type of land use present as well as, in some cases, land use facilitating the movement of pollution to water bodies. Diffuse pollution is difficult to measure due to the nature and mechanism of inputs but utilising existing land use classifications in conjunction with estimated export coefficients i.e. the nutrient load released from a land use type per unit area, have been used to predict nutrient loadings from catchments (Smith et al. 2005). Unlike the methodology in Smith et al. (2005), Foy & Girvan (2004) utilised the flow weighted mean concentration (FWMC), where in order to eliminate bias during sampling of nitrogen inputs they weight each concentration collected by the flow at the time of sampling. CORINE land use data was also used grouped into six major categories rather than individual classes. These approaches can be integrated to estimate the total levels of P and N a waterbody receives. As stated soil loss is becoming a pressing issue with regards to eutrophication and the flow of water is regarded as being the major driver of erosion of river banks, especially when compared to other factors such as wind erosion (Panagos et al. 2015). The RUSLE 2015 model (Revised Universal Soil Loss Equation) is a revised version of the USLE (Universal Soil Loss Equation) model currently employed by a vast majority of EU member states to monitor and predict soil and sediment erosion. It estimates long term average annual soil by sheet and rill erosion mechanisms and uses a combination of field sampling points (across Europe), land use cover, rainfall data from 1541 monitoring stations, vegetation density data, land management practices and Digital Elevation Modeling (DEM) (Panagos et al. 2015).

Human activity in the surrounding areas or rivers alter water and sediment supply which can lead to a stabilization or destabilization of rivers which ultimately cause changes to in stream habitat, whether causing degradation or improvements (Allan 2004). A coefficient modelling approach was applied to the river Kennet for the 1931-1991 period and found that increased levels of nitrogen would occur based upon the model with the expected changes to land use, fertiliser use and rainfall levels. Land use changes including an

increase in human population, and in land devoted to agricultural areas and a decrease in the number of forested areas is thought to be the primary driver (Whitehead et al. 2002). Lowland areas are usually the more densely populated and with that bring larger levels of industrial and residential areas. Areas surrounding the upper reaches of a river are typically more sparsely populated and agriculture usually dominates. Wade et al. (2001) found that land use changes along the river Dee would have a predominantly greater effect in the level of nutrients present within a river in the upper reaches of a river than compared to at a lowland level due to an already high level of nutrients present in the densely populated lowland areas. Land use change in the upper reaches would have a more prominent and noticeable effect as they are usually nutrient deficient as nutrients flow downstream to lower reaches.
Large river catchments have varying land use types and have complex hydrological and chemically reactive drainage systems that often respond rapidly to land use changes. It was determined that the total load of nitrogen and phosphorus exported to the Windrush catchment was defined by the land use type and management scheme involved upstream of sampling points instead of traditional beliefs that proximity and connectivity of the source to the drainage network affects readings obtained (Johnes 1996). Nutrient export is often determined by the type of bedrock, distance of nutrient sources and the connection of source to water body whilst monitoring of the total nutrient export of P and N is better monitored by looking at individual species of each nutrient type as more variation persists (Johnes 1996).

The riparian corridor surrounding riverine habitats plays a vital role in regulating the levels of diffuse pollution that can enter the aquatic environment, for example the predominantly wetland surrounding the river Havel, Germany, allows for the presence of heathland and pasture which are generally renowned to provide large levels of diffuse nitrogen to water bodies (Krause, et al. 2008). These areas are closest to waterbodies, and therefore the land use type within these areas and have the greatest potential to contribute nutrients or influence the levels of nutrients entering the water body. Riparian areas can therefor potentially contribute to or ameliorate diffuse pollution depending on their prevailing land use. Whilst many studies have employed different approaches to predict diffuse inputs from catchment land use, the relative importance of riparian characteristics compared to broader scale land use patterns in determining diffuse inputs remains not well established. To protect rivers and catchments, regulations are in available to reduce the impact of diffuse pollution during construction activities, agriculture and livestock farming. The Controlled Activities Regulations (CAR) published by the Scottish Environment Protection Agency (SEPA) aim to regulate land-use practices to reduce such effects. At the core of the regulations are standards of good practice, the four point plan (covering guidelines on nutrient use, dirty water minimisation, risk assessment guidelines and managing water margins) and Water Guidelines that provide a baseline of good and accepted practices when working near river environments to provide protection and facilitate improvements to water quality (Scottish Environment Protection Agency 2014b).

1.4.1 Hydromorphology in relation to pollution impacts

Hydromorphology incorporates the movement of water and the physical structure of the river including any alterations, either artificial or natural, and can have significant impacts upon macroinvertebrate communities by altering flow types and magnifying pollution impacts. Hydromorphology of river systems is a complex and well-researched area, but there are still important limitations in the extent to which the links and interactions between Hydromorphology and ecology are understood (Vaughan & Ormerod 2010). In many cases, systems are treated as uniform in terms of variation in solute type and concentration in relation to depth and width (Terradas 2010), leading to potentially unrealistic views of interactions between, Hydromorphological characteristics, nutrients and sediment. Transient storage of nutrients is the temporary storage of stream water separately from the main flow within the stream which, as a result, increase the contact time of the main channel water with nitrogen and phosphorus containing sediments (Ensign & Doyle 2005). Higher levels of transient storage are often found in areas of the river with slow moving or low flowing water, such as pools, and increased transient storage is linked to increase nutrient retention in water bodies. It is physical stream features such as vegetation, rocks and coarse woody debris, and how these alter flow regimes that influence the levels of transient storage occurring – otherwise known as hydromorphology. This, in turn, slows down the flow of water catering for nutrient storage to occur (Ensign & Doyle 2005) where streams with large changes in gradient accentuate this effect. Essentially even when river flow seems uniform there are complex interactions taking place within the water column between the groundwater, the closest layer of water to the streambed, and the main body of water. The vertical mixing of the two is stipulated as one of the main controls of fine sediment deposition, storage and flushing within river substrates and as a result can cause substratum smothering. This has implications for the associated impacts that diffuse pollution entities, mainly N and P, could have upon the waterbody – a reduction in vertical mixing and a reduced level of nutrient transport (Mathers & Wood 2016).

1.5 Water Quality Assessment and Monitoring

Within Europe, one of the main drivers of change to riverine environments is the Water Framework Directive (WFD) (European Community 2000). The WFD has been transposed into Scottish Law through the Water Environment and Water Service Act (Scottish Government 2003) and, the Scottish Environment Protection Agency is the body responsible for monitoring and assessment of progress towards targets. Diffuse agricultural pollution is the biggest contributor of diffuse pollution to water ways in Scotland with 75% of land being utilised for agricultural purposes (Scottish Environment Protection Agency 2007). Diffuse pollution is a persistent problem world-wide as well as the Almond catchment, with large sections of its reaches running through agricultural and urban areas leading to fluxes in the levels of nitrogen and phosphorus present within the water body, which ultimately can lead to eutrophication. Eutrophication severely reduces water quality of rivers, bathing waters and marine environments and is the process by which oligotrophic or mesotrophic waterbodies become ‘flooded’ with excess nutrients causing a large spike in growth of aquatic plants and algae (Smith et al. 1999). This in turn leads to the degradation of riparian habitat, smothering of salmonid spawning areas and the deoxygenation of water bodies (D'Arcy & Frost 2001).

1.6 The Use of Macroinvertebrates in Monitoring Water Quality

The sampling of macroinvertebrates in freshwater forms a key component in the classification of water bodies within the United Kingdom. Environmental agencies such as SEPA (Scottish Environment Protection Agency 2016), Environment Agency (Environment Agency 2016) and Environmental Protection Agency (Ireland) (Environmental Protection Agency 2016) are responsible for monitoring, protecting and restoring waterbodies utilise macroinvertebrate surveys as part of regular monitoring schemes. Macroinvertebrates are a diverse group of organisms that can inhabit a wide variety of habitats within the stream or river environment under varying conditions. As a result they are used extensively as a biological indicator for assessment regarding the overall ecological health of a water body (Kartikasari et al. 2013). Benthic dwelling invertebrates can be sensitive to changes within the environment and are utilised as a primary indicator of water quality caused mainly by organic pollution. However, they are also used to assess other pollutions like heavy metals, sediment and in some cases climate change (Durance and Ormerod 2007) which perhaps emphasises their usefulness as an assessment tool. Macroinvertebrates occupy an important role in the food webs of organisms such as fish species and with a drastically reduced presence of these, fish communities could reduce in size. Therefore, macroinvertebrates are effective at indicating a variety of ways the surrounding habitat could be degraded. Due to the ease of collection, relatively low costs and the above ecological factors macroinvertebrates are now sampled and integrated into water quality and ecological assessment protocol, which has facilitated the growth and development of indices as detailed in the section below.

1.6.1 Macroinvertebrate Indices

Macroinvertebrates are utilised when assessing a variety of forms of pollution that could alter water quality or ecological status of a river, thus allowing for the assessment and quantification of impacts upon water quality and ecology within the river. Pollution in terms of water quality and possible effects upon macroinvertebrate assemblages is a well-researched topic with some wide-range methods of assessment having been developed across different habitats. In relation to sediment effects, Extence et al. (2011b) utilised invertebrates to create an index known as the Proportion of Sediment Sensitive Invertebrates (PSI) to determine the feasibility of using macroinvertebrates as an indicator for the quality of fine sediment accumulation and/or erosion over time. The method predominantly relies upon work and models previously developed by Extence, such as Lotic-invertebrate Index for Flow Evaluation (LIFE) and Community Conservation Index (CCI) where macroinvertebrate species were assigned one of four possible scores from the Fine Sediment Sensitivity Ratings (FSSR) scale.

The development of metrics and indices facilitated the use of prediction based metrics and posed serious questions regarding water quality and allowed the determination and implementation of reference conditions to become standard practice when assessing water body characteristics. The River Invertebrate Classification Tool Using (RICT) is software that is used to predict and classify the state of a water bodies freshwater invertebrate abundance, diversity and community structure based upon physical and chemical parameters taken at sample sites. This uses the RIVPACS (River Invertebrate Prediction and Classification System) model to provide site-specific reference values whilst complying with the WFD classification. Two metrics are used during this classification and are assessed separately and then a combined approach to provide invertebrate classification. Average Score Per Taxon (ASPT) and Number of Taxa (N-Taxa) are the metrics used during this process and are commonly calculated metrics based upon invertebrate data. Using a combination of RICT and RIVPACS, outputs include an Ecological Quality Rating (EQR) that provides a face value classification and an estimate of the probability of the result belonging to any of the WFD classes. The WFD assessment dictates that the ASPT element of classification is abundance weighted and the metrics provided are based upon site specific predicted reference values which are derived from inputted physical and chemical parameters (Clarke & Davy-Bowker 2014).

1.6.2 Diffuse Pollution in Scottish Freshwaters

Diffuse pollution has been a recognised problem within Scotland since as early as 1996 and is detailed within the SEPA state of environment report (Scottish Environment Protection Agency. 1996). The Significant Water Management Issues report (SWMI), published by SEPA, highlights more recent issues and the importance of diffuse pollution within river systems. The report was last updated in 2014 and identifies diffuse pollution, particularly from agriculture, as a key factor for water bodies that are at risk of failing target water quality standards. It further states that within rivers, diffuse pollution is the single and most important pollution pressure (Scottish Environment Protection Agency. 2012). Fish species, in particular salmonids, are susceptible to pressures caused by diffuse pollution as increased siltation causes damage to fish spawning sites and these effects are seen most prominently in the riffles where spawning occurs (D'Arcy & Frost 2001).

1.7 River Almond

The river Almond is a river system running through the Central Belt of Scotland, rising in the Shotts hills and discharging into the Firth of Forth at Cramond with the main stem totalling 46 km in length. In the most recent assessment by SEPA in 2009, the reaches the designated surface water bodies along river Almond generally attained a poor ecological grade meaning they had not met the quality standards set by SEPA. This led to the longer term objective that not only the water quality of the reaches is increased to a good standard, but the ecological status also by 2020 as stipulated within the WFD (Scottish Environment Protection Agency 2011). Historically mining activity occurred within close proximity to the river and since 1963 the areas surrounding the river have become increasingly more urbanised with housing, major road networks and a variety of industrial areas being constructed. Prior to industrialisation, the area surrounding Livingston was open farmland but the increasing population within Glasgow meant more housing was required and Livingston was chosen as a part of the New Towns Act of 1946 to ease overcrowding within the city (Her Majesty's Stationary Office 1946). The original settlement of Livingston is referred to now as Livingston Village with the new areas being built up around it. The Census in 2001 revealed that Livingston had a population of 50,826 and the most recent census in 2011 showed the population had grown to 56,269 showing that Livingston is growing in size. The more rural areas within the catchment leave the river susceptible to agricultural run-off and forested areas, such as Almondell Country Park, pose an increased sedimentation risk. The pollution sources outlined above are detailed in D’Arcy& Frost (2001) and they state that diffuse pollution has been a recognised problem within Scotland since as early as 1996 and is detailed within the SEPA state of environment report (Scottish Environment Protection Agency 2006). The increased levels of urbanisation coupled with a more scientific and professional approach to farming to maximise crop yields has resulted in large levels of diffuse inputs to the river Almond, which in the first instance are driven by land use changes. One of the main factors in determining the ecological status of a river, along with macroinvertebrate sampling, is the presence of migratory fish species, which in the case of the Almond catchment suggests that diffuse pollution, coupled with the effects of barriers to migration, is having a prominent role in the reducing ecological status.

1.8 Aims and Objectives

Motivation for and questions addressed by the current study

The preceding review has established the characteristics and significance of diffuse pollution as an impact on environmental quality in freshwaters across the world and more specifically in Scotland. Catchment land use has been identified as a key driver of diffuse inputs in both urban and rural contexts and a range of studies have begun to examine the relationships between land use and diffuse pollution impacts.

Despite the considerable body of literature developing in this area, there are still significant gaps in knowledge of how catchment land use-based approaches relate to observed patterns of diffuse pollution and the impact of these inputs on the environmental quality of freshwaters of different habitat characteristics. This study aims to contribute to the understanding in these areas, specifically addressing:

- How do predictive catchment land use-based approaches to assessment relate to the extent and type of diffuse pollution inputs actually observed along a river?
- Are any relationships between land use based assessment and observed inputs sensitive to the spatial scale of land use considered?

- How do predicted or observed diffuse inputs relate to observed environmental quality, as measured by invertebrate-based biotic indices or water chemistry?

The project was split into three main phases with specific details of each phase and associated procedures used to achieve the objectives detailed below. The focus of this study was on the assessment of diffuse pollution sources and their impact within the River Almond catchment, Scotland primarily upon macroinvertebrate communities. In order to ascertain the impact of diffuse pollution with relation to land use along the main stem of the river Almond, the following specific objectives were formulated:

1. Assess the extent of diffuse and point source pollution sources along the main stem of the river Almond.
2. Determine the relationships present between land use and diffuse pollution at a catchment and local scale.
3. Quantify the impact of diffuse pollution inputs and associated impacts on environmental quality of the river.

The first objective was achieved by analysing land use within the catchment as a whole, sub-catchments and the specific areas along the river corridor or riparian areas. The second objective was achieved through a walkover survey of the main stem of the river Almond quantifying the number and extent of diffuse and point source inputs present as well as mapping flow and habitat types. In turn, this facilitates identification of areas containing high diffuse loading with contrasting habitats. Finally, the third objective was achieved through the targeted sampling of high and low diffuse impact areas along with calculation of invertebrate-based metrics and habitat characterisation. This aimed to highlight any differences between invertebrate community structure with regards to contrasting habitat whilst providing an insight into any variation between populations on a broader scale. The River Invertebrate Classification Tool (RICT) was utilised to highlight and identify key differences between macroinvertebrate community composition against a predicted or reference composition, allowing the expected composition to be shown when no pressure or impact, chemical or physical, is placed upon that invertebrate community. Utilising RICT in conjunction with invertebrate indices available aided in the detection of impacts upon communities.

2.0 Methods

Catchment Determination

The boundaries of the river Almond catchment was derived utilising the hydrology toolset found within ArcGIS Version 10.4 (ESRI 2016), based on the data supplied by the Ordnance Survey Terrain 5 raster DTM at 5 meter intervals (Figure 1).

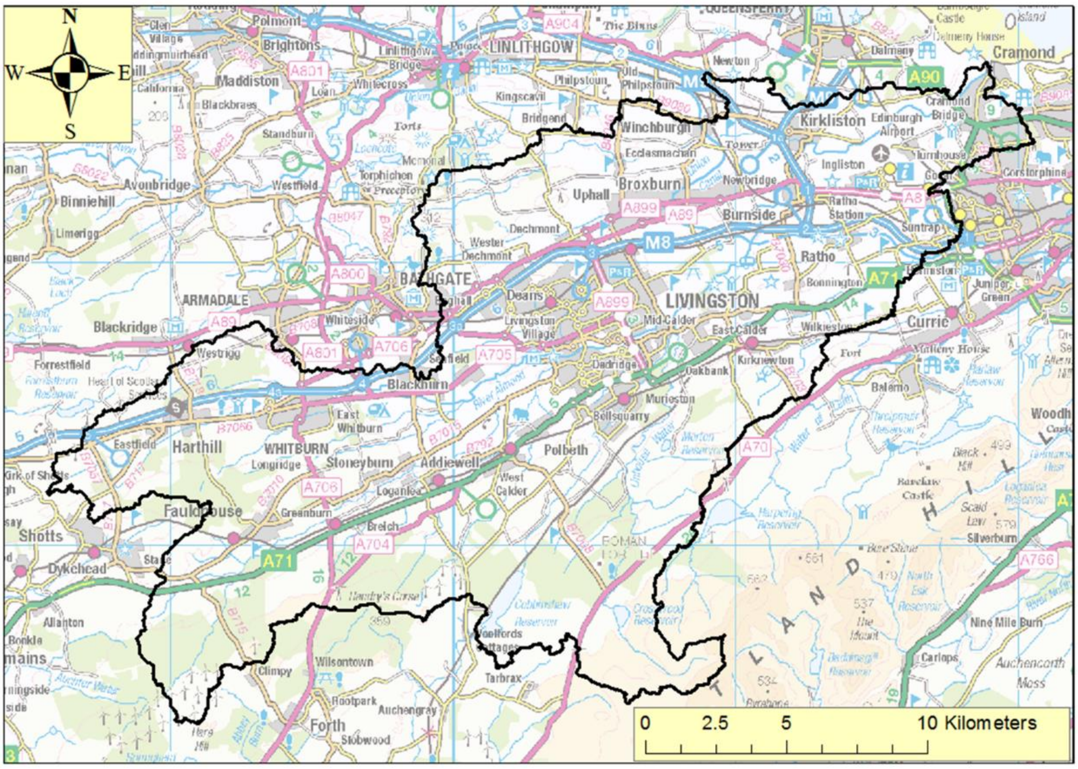


Figure 1: Almond catchment as derived by the watershed tool in ArcGIS. © Crown Copyright and Database Right (2016). Ordnance Survey (Digimap Licence).

SEPA have further sub-divided the catchment into a series of waterbodies for the purposes of WFD reporting (Scottish Environment Protection Agency 2014a) Based on pour points placed at the lowest point of each waterbody, sub-catchments have also been determined, as shown in Figure 2.

Utilising the OS OpenRivers Dataset (29/03/2016) the extent of the main stem of the river Almond was determined, along with burns and rivers joining the main stem, to allow for surveys to be completed. This is also shown in Figure 2 along with waterbody and sub catchment information.

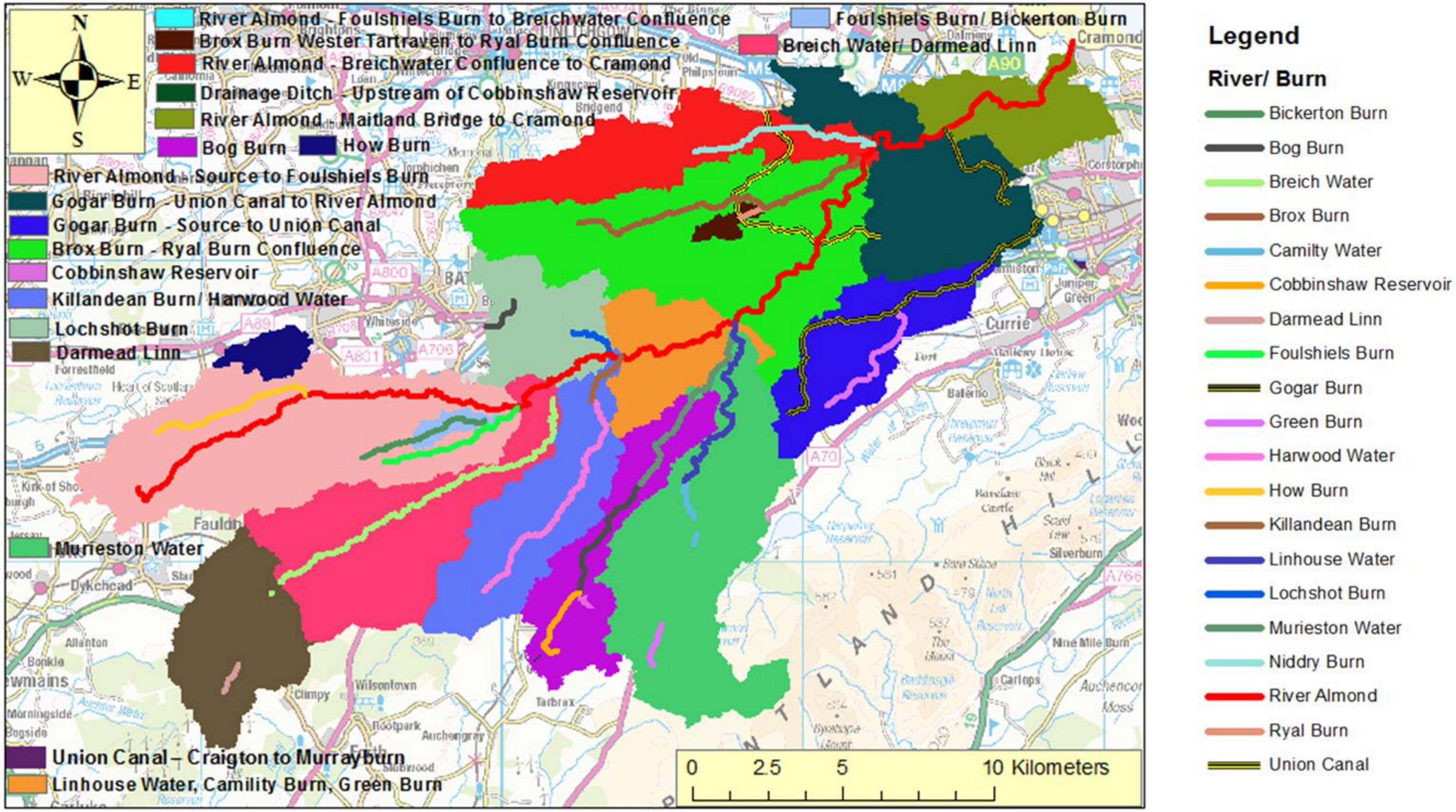


Figure 2: SEPA waterbody classification with OS OpenRivers information and SEPA sub catchment information (SEPA, 2014). © Crown Copyright and Database Right (2016). Ordnance Survey (Digimap Licence).

2.1 Phase I

The first phase of the work aimed to provide a catchment level overview of diffuse pollution present utilising existing models of nitrogen and phosphorus inputs based on land use types. CORINE land use data (Environmental Protection Agency 2015), for the years 2000, 2006 and 2012 was used in conjunction with land-use specific nutrient loading coefficients to predict nutrient loads in different areas of the catchment. An overview of the land-use types and distribution in each year is shown in Figures 3a-c.

Legend

CORINE Land Use Categories

Land Use

- Airports
- Broad-leaved forest
- Complex Cultivation Patterns
- Coniferous forest
- Construction sites
- Discontinuous urban fabric
- Dump sites
- Green urban areas
- Industrial or commercial units
- Land principally occupied by agriculture, with significant areas of natural vegetation
- Mineral extraction sites
- Mixed forest
- Moors and heathland
- Natural grasslands
- Non-irrigated arable land
- Pastures
- Peat bogs
- Road and rail networks and associated land
- Sparsely vegetated areas
- Sport and leisure facilities
- Transitional woodland-shrub

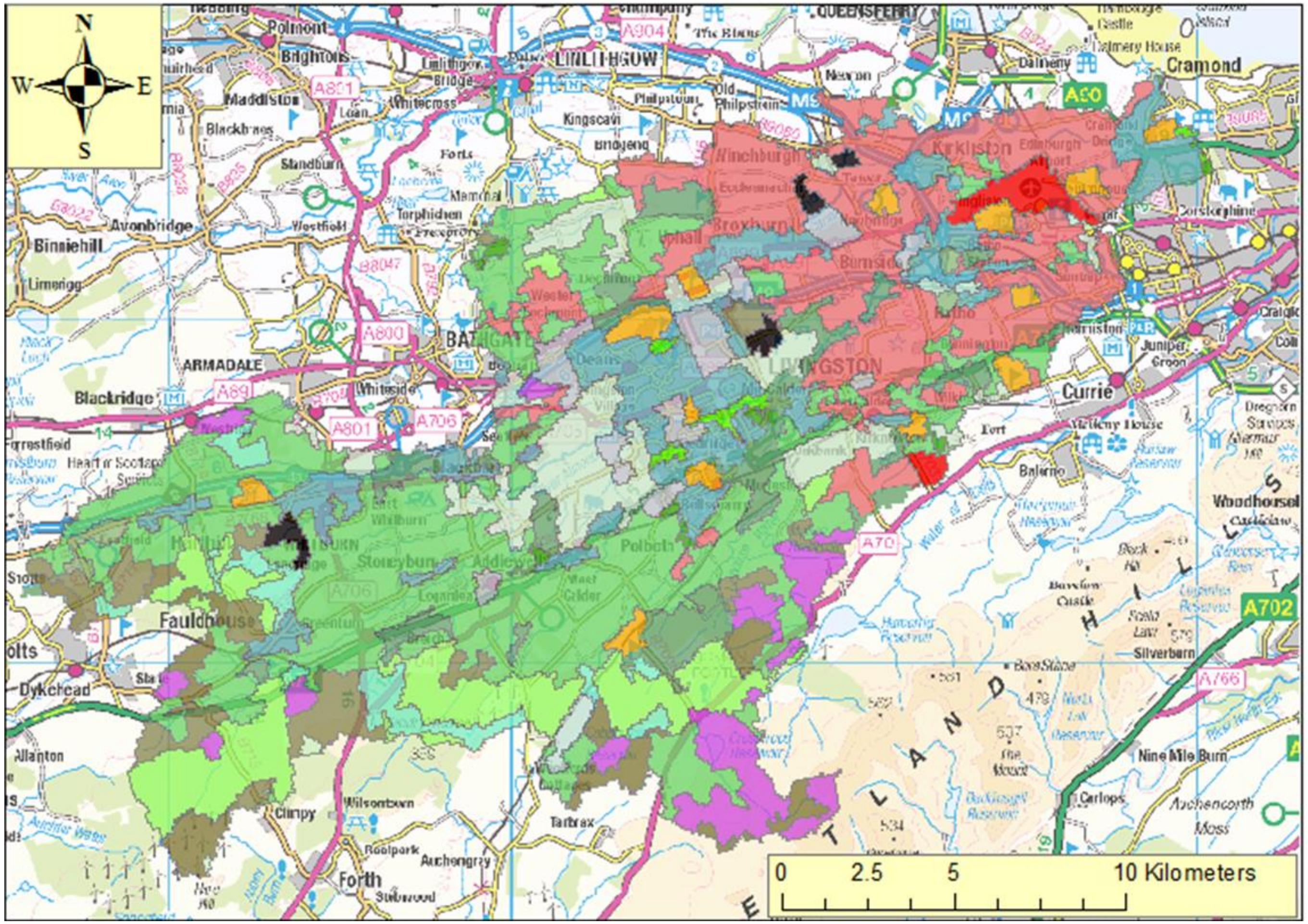


Figure 3a: 2000 CORINE land use dataset raster © Crown Copyright and Database Right (2016). Ordnance Survey (Digimap Licence). CORINE land use data supplied by European Environment Agency (2016).

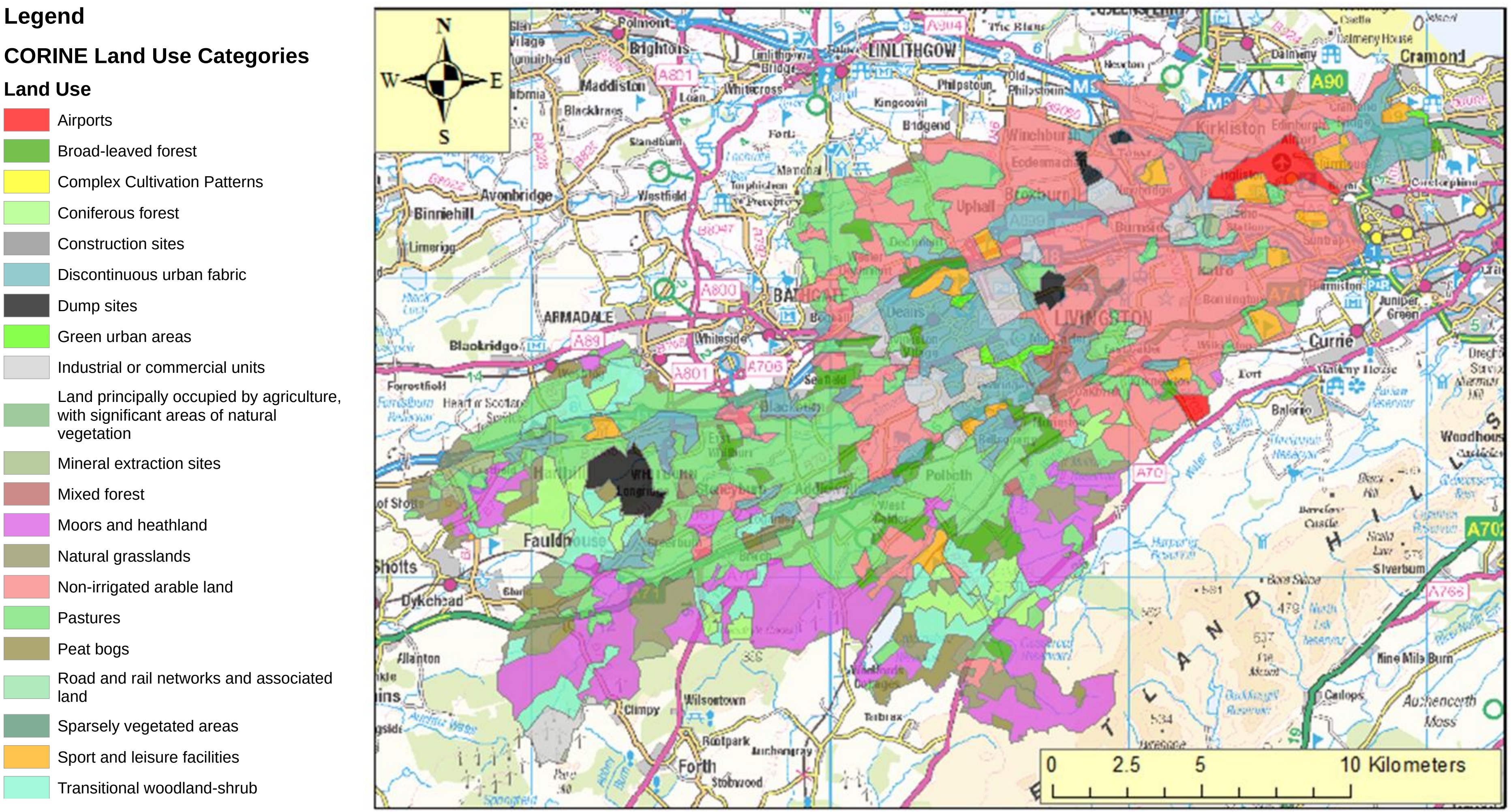


Figure 3b: 2006 CORINE land use dataset raster © Crown Copyright and Database Right (2016). Ordnance Survey (Digimap Licence). CORINE land use data supplied by European Environment Agency (2016).

Legend

CORINE Land Use Categories

Land Use

- Airports
- Broad-leaved forest
- Complex Cultivation Patterns
- Coniferous forest
- Construction sites
- Discontinuous urban fabric
- Dump sites
- Green urban areas
- Industrial or commercial units
- Land principally occupied by agriculture, with significant areas of natural vegetation
- Mineral extraction sites
- Mixed forest
- Moors and heathland
- Natural grasslands
- Non-irrigated arable land
- Pastures
- Peat bogs
- Road and rail networks and associated land
- Sparsely vegetated areas
- Sport and leisure facilities
- Transitional woodland-shrub

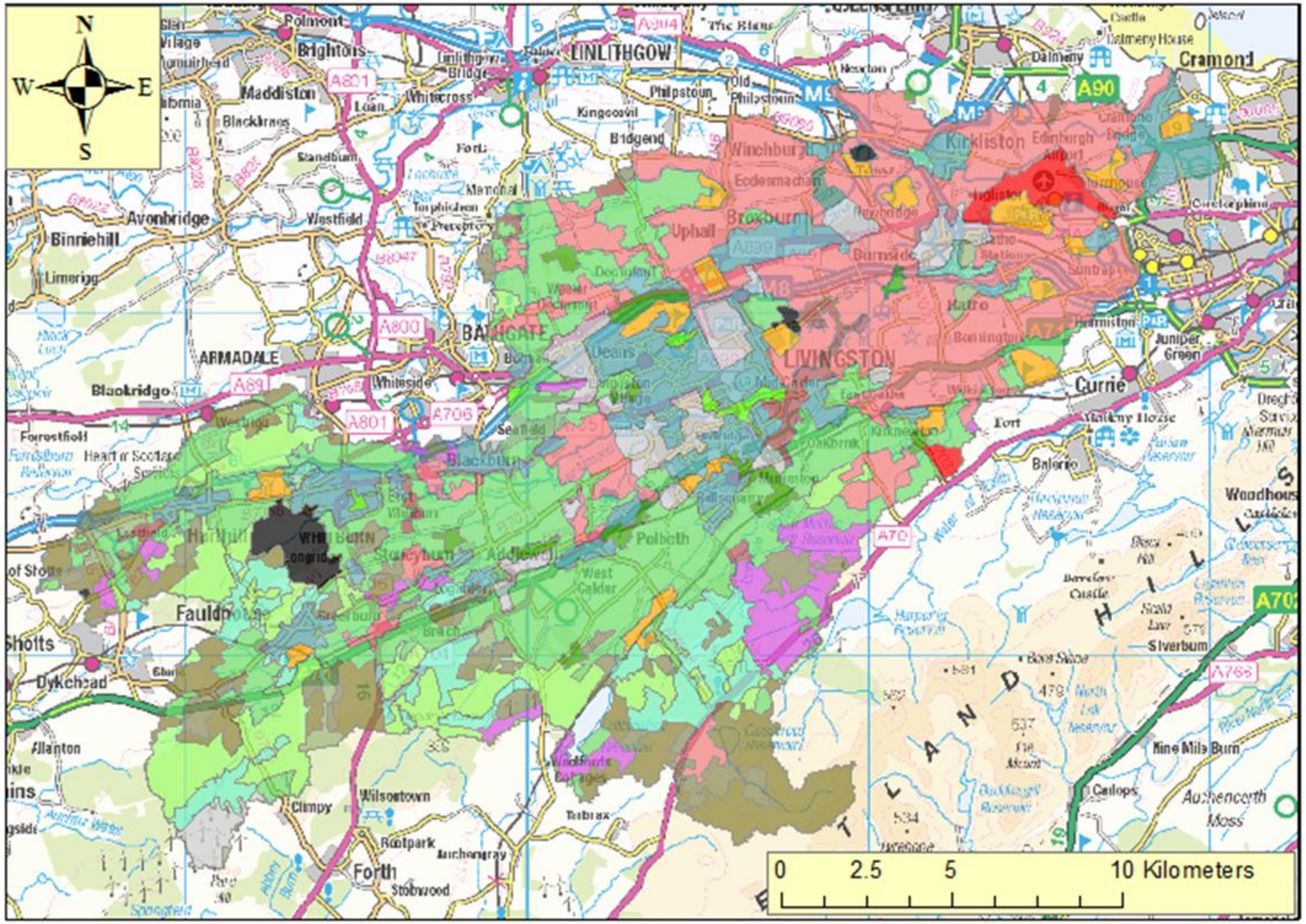


Figure 3c: 2012 CORINE land use dataset raster © Crown Copyright and Database Right (2016). Ordnance Survey (Digimap Licence). CORINE land use data supplied by European Environment Agency (2016).

Each colour shown in the Figures above represents a different land use type and shows the area within the catchment that is designated to that land use type. The land use types present within the CORINE data can be found in Table 1 as well as accompanying legends.

Table 1: Land use types designated by the CORINE land use data.

Surface Type	Sector	Land Use
Artificial surfaces	Urban fabric	Continuous urban fabric
		Discontinuous urban fabric
	Industrial, commercial and transport units	Industrial or commercial units
		Road and rail networks and associated land
		Port areas
		Airports
	Mine, dump and construction sites	Mineral extraction sites
		Dump sites
		Construction sites
	Artificial, non-agricultural vegetated areas	Green urban areas
		Sport and leisure facilities
Agricultural areas	Arable land	Non-irrigated arable land
		Permanently irrigated land
		Rice fields
	Permanent crops	Vineyards
		Fruit trees and berry plantations
		Olive groves
	Pastures	Pastures
	Heterogeneous agricultural areas	Annual crops associated with permanent crops
		Complex cultivation patterns
		Land principally occupied by agriculture, with significant areas of natural vegetation
		Agro-forestry areas
Forest and semi natural areas	Forests	Broad-leaved forest
		Coniferous forest
		Mixed forest
	Scrub and/or herbaceous vegetation associations	Natural grasslands
		Moors and heathland
		Sclerophyllous vegetation
		Transitional woodland-shrub
	Open spaces with little or no vegetation	Beaches, dunes, sands
		Bare rocks
		Sparsely vegetated areas
		Burnt areas
		Glaciers and perpetual snow
Wetlands	Inland wetlands	Inland marshes
		Peat bogs
	Maritime wetlands	Salt marshes
		Salines
		Intertidal flats
Water bodies	Inland waters	Water courses
		Water bodies
	Marine waters	Coastal lagoons
		Estuaries
		Sea and ocean

2.1.1 Nutrient Loading

In order to provide an estimate of the nutrient loading to the water bodies from the different land use types, the CORINE land use data were combined with nutrient coefficients, principally for P but also N. Phosphorus (as Total Phosphorus, TP) coefficient data was taken from Smith et al. (2005), (Table 2). The coefficients calculated by Smith et al. (2005) were derived from data collected from rivers in Northern Ireland but due to the similarity of land use and geographical context of the overall study area, these were deemed appropriate for use in the current study.

Table 2: Land use data with phosphorus coefficient data. Values given are mean ± standard deviations. (Smith et al. 2005).

Surface Type	Sector	Land Use	Export Coefficient (kg P ha^{-1} $year^{-1}$)
Artificial surfaces	Urban fabric	Continuous urban fabric	1.2±0.90
		Discontinuous urban fabric	1.2±0.90
	Industrial, commercial and transport units	Industrial or commercial units	
		Road and rail networks and associated land	1.2±0.90
		Port areas	2.5±1.6
		Airports	2.5±1.6
	Mine, dump and construction sites	Mineral extraction sites	2.5±1.6
		Dump sites	2.5±1.6
		Construction sites	2.5±1.6
	Artificial, non-agricultural vegetated areas	Green urban areas	0.83±0.17
		Sport and leisure facilities	1.2±0.90
Agricultural areas	Arable land	Non-irrigated arable land	4.88±1.12
		Permanently irrigated land	-
		Rice fields	-
	Permanent crops	Vineyards	0.83±0.17
		Fruit trees and berry plantations	0.83±0.17
		Olive groves	0.83±0.17
	Pastures	Pastures	0.78±0.12
	Heterogenous agricultural areas	Annual crops associated with permanent crops	0.83±0.17
		Complex cultivation patterns	2.33±0.27
		Land principally occupied by agriculture, with significant areas of natural vegetation	0.49±0.11
		Agro-forestry areas	-
Forest and semi natural areas	Forests	Broad-leaved forest	0.26±0.14
		Coniferous forest	0.36±0.04
		Mixed forest	0.26±0.15
	Scrub and/or herbacous vegetation associations	Natural grasslands	0.65±0.25
		Moors and heathland	0.13±0.07
		Sclerophyllous vegetation	-
		Transitional woodland-scrub	0.26±0.14
	Open spaces with little or no vegetation	Beaches, dunes, sands	-
		Bare rocks	-
		Sparsely vegetated areas	-
		Burnt areas	-
		Glaciers and perpetual snow	-
Wetlands	Inland wetlands	Inland marshes	0.23±0.17
		Peat bogs	0.23±0.17
	Maritime wetlands	Salt marshes	-
		Salines	-
		Intertidal flats	-

Water bodies	Inland waters	Water courses	-
		Water bodies	-
	Marine waters	Coastal lagoons	-
		Estuaries	-
		Sea and ocean	-

A study by Foy & Girvan (2004) used a similar approach to develop nitrate-N coefficients for rivers in Northern Ireland (see Table 3). The number of coefficients available for N data was limited in comparison to P data.

Table 3: Land use data with nitrogen coefficient data (Foy & Girvan 2004).

Surface Type	Sector	Land Use	Nitrate – N Tonnes km^2 yr^{-1}
Agricultural areas	Arable land	Non-irrigated arable land	4.367
	Heterogenous agricultural areas	Annual crops associated with permanent crops	2.036
		Complex cultivation patterns	2.759
		Land principally occupied by agriculture, with significant areas of natural vegetation	0.829
	Pastures	Good	2.036
		Poor	0.747
		Mixed	1.511
Forest and semi natural areas	Forests	Broad-leaved forest	0.000
		Coniferous forest	0.086
		Mixed forest	0.000
	Scrub and/or herbacous vegetation associations	Transitional woodland-scrub	-

Nutrient loading to water bodies was achieved by calculating the area for each land use present and combining this with the nutrient coefficients in the following formula:

$$\text{Total Nutrient Export} = \text{Nutrient Coefficient X Total Land Use Area}$$

Calculations were undertaken per area of each land-use and then combined to give an overall estimate for each land-use type using standard GIS tools. Calculations were undertaken for each year that CORINE data were available for (2000, 2006 and 2012) to determine the extent of changes over time.

2.1.2 River Corridor/ Riparian Areas

The land use closest to the main stem of the river has the potential to export the greatest amount of diffuse pollution into the river. Total export of P nutrient pollution entering the river from land-use within a distance of 25m, 50m, 100m, 200m and 500m from the channel was determined using the Buffer tool within ArcGIS in order to clip the land use data for respective years to the differing distances.

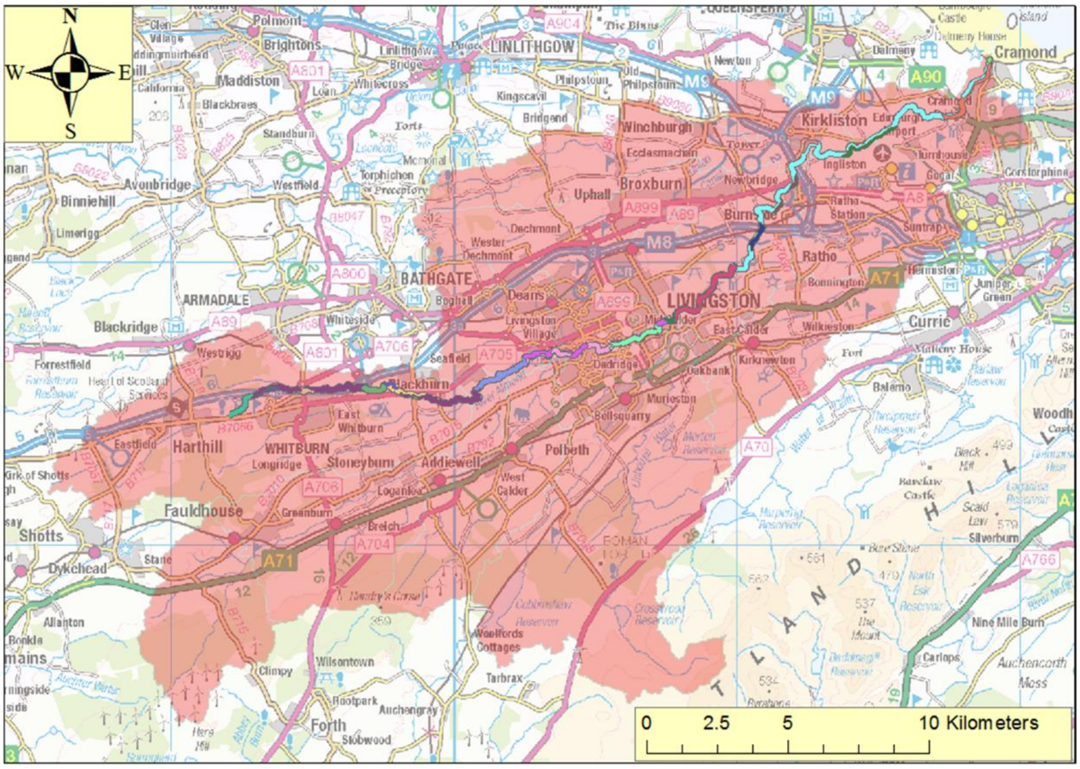


Figure 4a: CORINE land use data clipped to form riparian corridor 100m from river Almond. © Crown Copyright and Database Right (01/01/2016). Ordnance Survey (Digimap Licence). CORINE land use data supplied by European Environment Agency (2016).

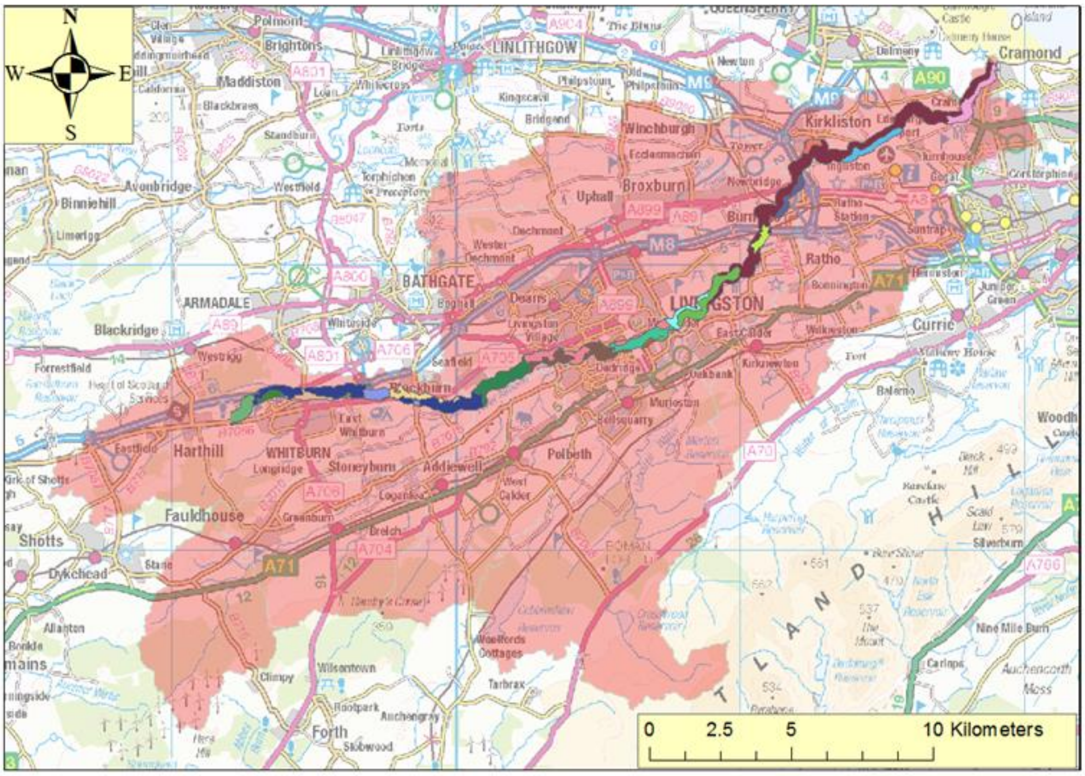


Figure 4b: CORINE land use data clipped to form riparian corridor 200m from river Almond. © Crown Copyright and Database Right (2016). Ordnance Survey (Digimap Licence). (1:90000 scale). CORINE land use data supplied by European Environment Agency (2016).

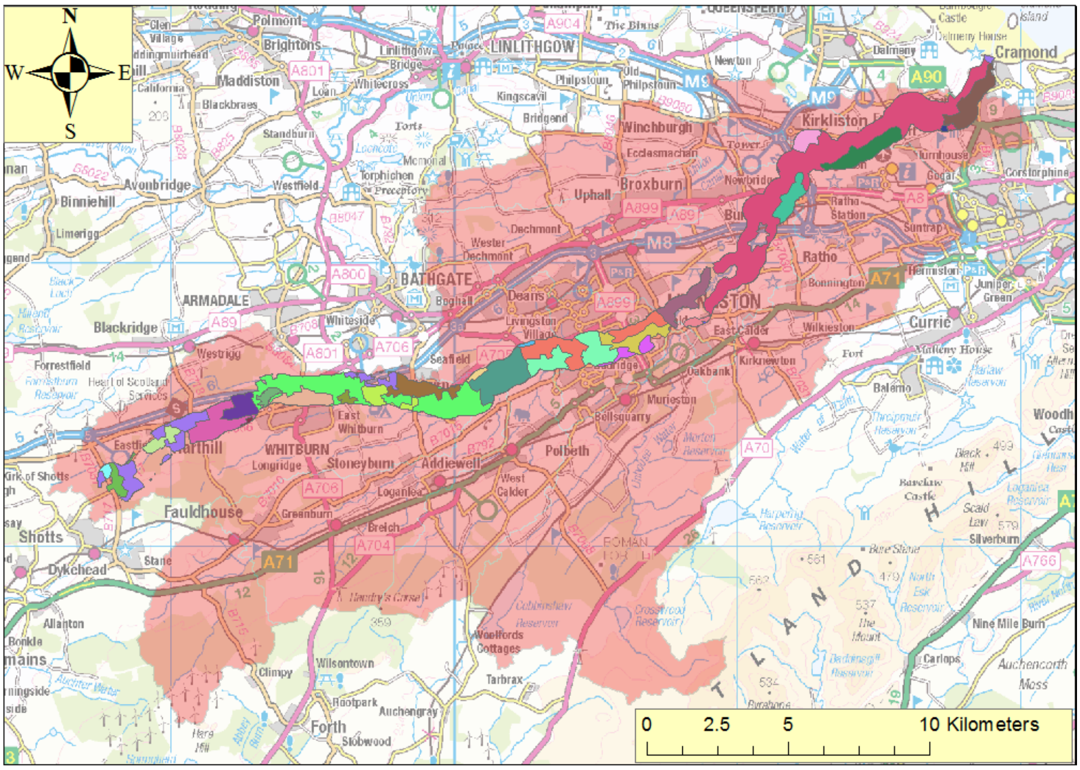


Figure 4c: CORINE land use data clipped to form riparian corridor 500m from river Almond. © Crown Copyright and Database Right (2016). Ordnance Survey (Digimap Licence). CORINE land use data supplied by European Environment Agency (2016).

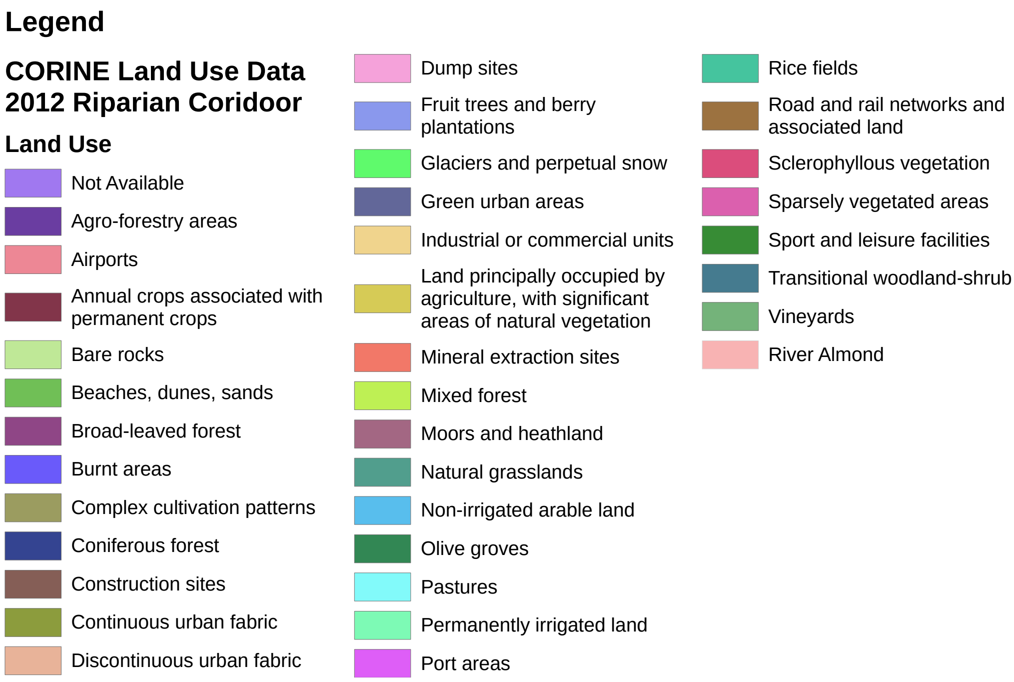


Figure 4d: Land use legend for riparian corridor.

2.2 Phase II

This involved undertaking a walkover survey of the main stem of the river, noting down key habitat types, flow types and diffuse and point source pollution entering the river. These were then digitised using ArcGIS in order to highlight areas under high and low diffuse pollution pressure. This was undertaken during winter months where peak levels of diffuse pollution regularly occur and was responsive to prevailing weather conditions to maximise sources being identified. Weather conditions over the winter months, November and December, were particularly harsh and as a result walkover surveys commenced later than expected.

2.2.1 Walkover Data Capture

The walkover data capture was carried out with training and guidance from the River Forth Fisheries Trust, specifically a training day taking place on the River Esk at Roslin Glen in November, on procedures as well as assistance in identifying habitat types, flow types, any bank and in channel modification present and any diffuse and point source inputs. The total length of the main stem is around 46km in length and was split into 12 sections for the purpose of the walkover - each section was approximately 4km in length with one section at 2 km with the lowest numbered section at the mouth of the river. Whilst a large part of the main stem was covered in the walkover and successfully digitised, some small sections where access was restricted or unsafe to continue have not been completed, this is illustrated in Figure 5.

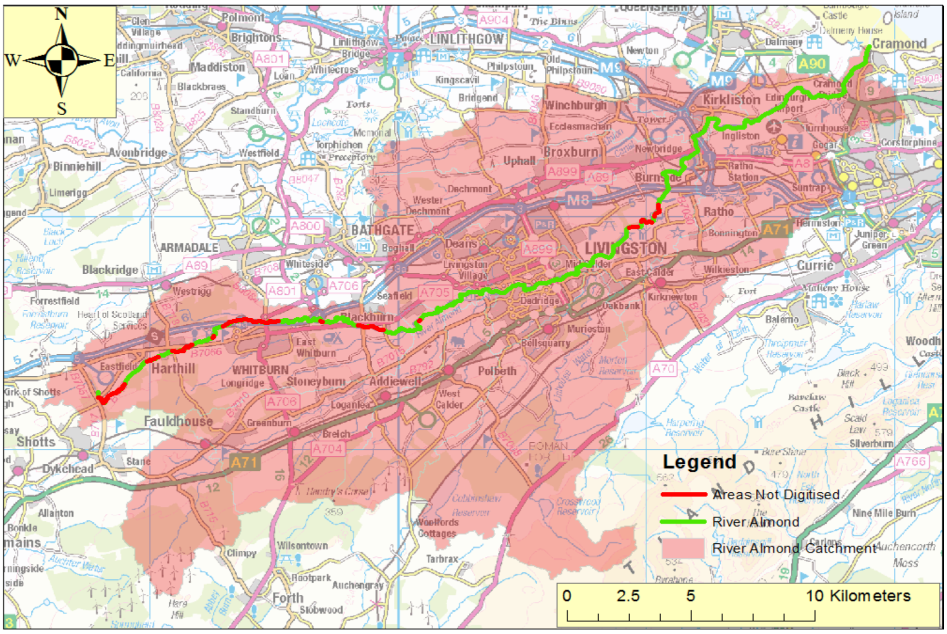


Figure 5: Progress line showing the areas completed during the walkover and digitised (Green) and the uncomplete sections (Red). © Crown Copyright and Database Right (2016). Ordnance Survey (Digimap Licence).

A list of the features recorded during the walkover are shown in the Table below.

Table 4: Recorded Features from walkover of river Almond.

Feature Class	Feature	Type
Flow Type	Broken Water	Area
	Cascade	Area
	Chaotic	Area
	Chute	Area
	Exposed Bedrock	Area
	Free Fall	Area
	Glide	Area
	Impoundment	Area
	No Perceptible Flow	Area
	Riffle	Area
	Run	Area
In-Channel Features	Boulder	Area
	Debris Dam	Point
	Erosion	Polyline
	Ford	Point
	Island	Point
	Large Woody Debris	Point
	Mid Channel Bar	Area
	Point Bar	Area
	Reinforcement	Polyline
	Side Bar	Area
Modification	Leves	Polyline
	Overhanging Vegetation	Polyline
	Poaching	Polyline
	Sand Bank	Polyline
	Weir	Point
Pollution Inputs	Diffuse Pollution Source	Point
	Point Source	Point
	Mine Leach	Point

The implemented methodology supplied by the River Forth Fisheries trust takes elements from several already established methodologies, notably the Environment Agency manual for salmonids habitat restoration (Hendry & Cragg-Hine 1997), habitat distribution recorded using the River Habitat Survey biotype approach (Raven et al. 1998) and the Environment Agency's 'Catchment Walkover for River Basin Management (Environment Agency 2014). The features were recorded using pencil onto Aqualase A3 waterproof paper with an Ordnance Survey Vector Map Local print out of the section of river being recorded. All maps used in the field were to 1:500 scale to allow sufficient detail to be recorded (Figure 6).

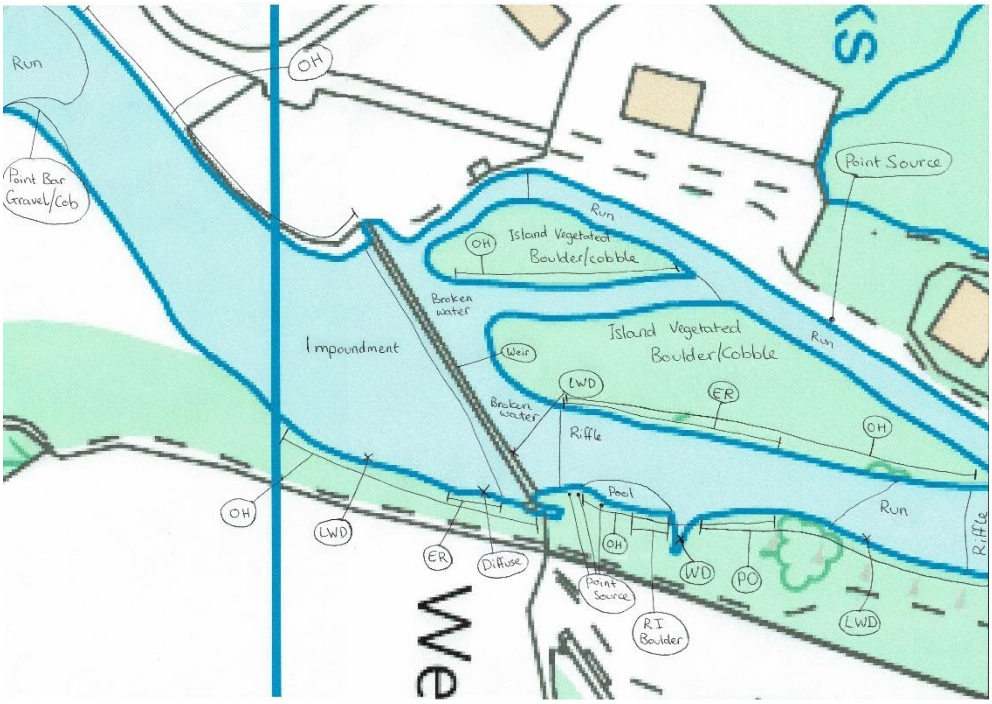


Figure 6: An example of an annotated A3 section of river showing habitat types, flow types, modification and pollution sources. © Crown Copyright and Database Right (01/01/2016). Ordnance Survey (Digimap Licence).

Some sections of the river were not mapped as access to the river was not available due to permission issues, excess vegetation or construction work on or close to the river bank. These sections have not been digitised but detailed field notes were taken at the end of each walkover day providing information on any areas of interest or accessibility issues that may have occurred. Photograph examples of different forms of diffuse pollution, modifications and flow types were taken during the walkover to convey the conditions and features present within the riparian corridor and can be found in Appendix section 1.

2.2.2 Digitising Walkover Data

Digitising of the walkover data using ArcGIS commenced on the 10/03/2016 to 29/05/2016 and began with a training day taking place 10/03/2016 at the River Forth Fisheries Trust office. Initial digitisation commenced with the creation of four blank shapefiles as shown in the Table 5.

Table 5: Details and topology of shapefiles created to enable digitising of walkover catchment to commence with type of data being recorded within the shapefile.

Shape File Type	Record Type
Bank Features	Point
Channel Features	Area
Channel Features	Point
Channel Features	Polyline

An example of a digitised section from the hand drawn maps is shown in Figure 7 below.

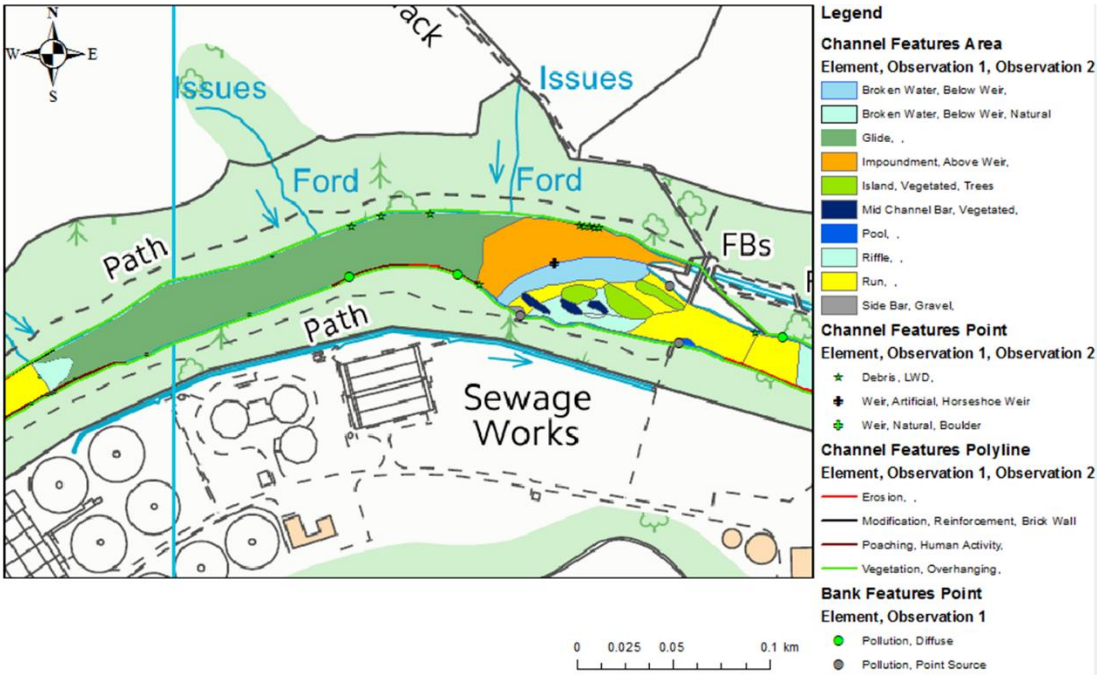


Figure 7: Digitised section of main stem of river Almond at Mid Calder waste water treatment plant. **© Crown Copyright and Database Right (2016). Ordnance Survey (Digimap Licence).**

2.3 Phase III

Phase III involved sampling of macroinvertebrate communities at specific sites on the river. There were two main aims of this work. Firstly, to establish whether predicted differences in nutrient loading and observed differences in extent of diffuse inputs (from the walkover survey) corresponded with differences in the invertebrate community present (and associated indices). Secondly to examine the extent to which local hydromorphological context affected the observed response of macroinvertebrate communities to diffuse pollution inputs.

2.3.1 Sampling Sites

Locations for sampling sites were determined using a combination of the catchment walkover and the CORINE land use data. Sections of the river with the riffle flow type were selected firstly for comparison between areas with high or low diffuse inputs. For the second aim, paired selecting riffles and pools in areas with high diffuse input land use types were identified, as outlined below. An additional criterion of being 250m from a point source, unless located upstream, was used to try and minimise the effect that point source pollution had on the results. Known CSO locations provided by the SEPA GIS team as well as the point source locations highlighted from the walkover survey were used in the site selection process. Sample sites are shown in Figure 8.

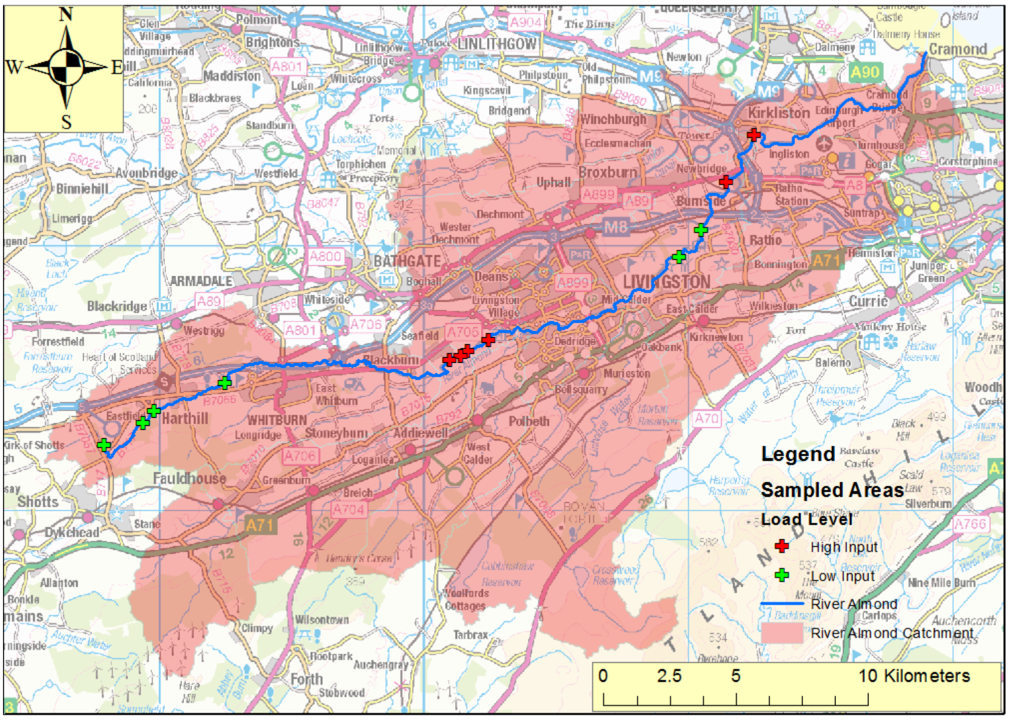


Figure 8: High and Low diffuse input sample sites along the mains stem of the river Almond. © Crown Copyright and Database Right (2016). Ordnance Survey (Digimap Licence).

According to the nutrient loading data, the highest diffuse phosphorus and nitrogen contributor is non-irrigated arable land in terms of area and coefficient value. In order to locate a sufficient number of areas where the riffle flow type and low diffuse phosphorus contributors coincided and to allow for enough spatial variation between sites, low diffuse phosphorus contributors included coniferous forests, broad leafed forests, natural grasslands, road, rail and associated land, dump and construction sites. Although more than one low phosphorus contributing land use was selected, the levels of phosphorus delivered to the river as derived by the coefficient method detailed previously are similarly low. These sites were selected utilising the buffer tool in ArcGIS, were the corresponding land use intersects with the riffle habitats, as shown below in Figure 9.

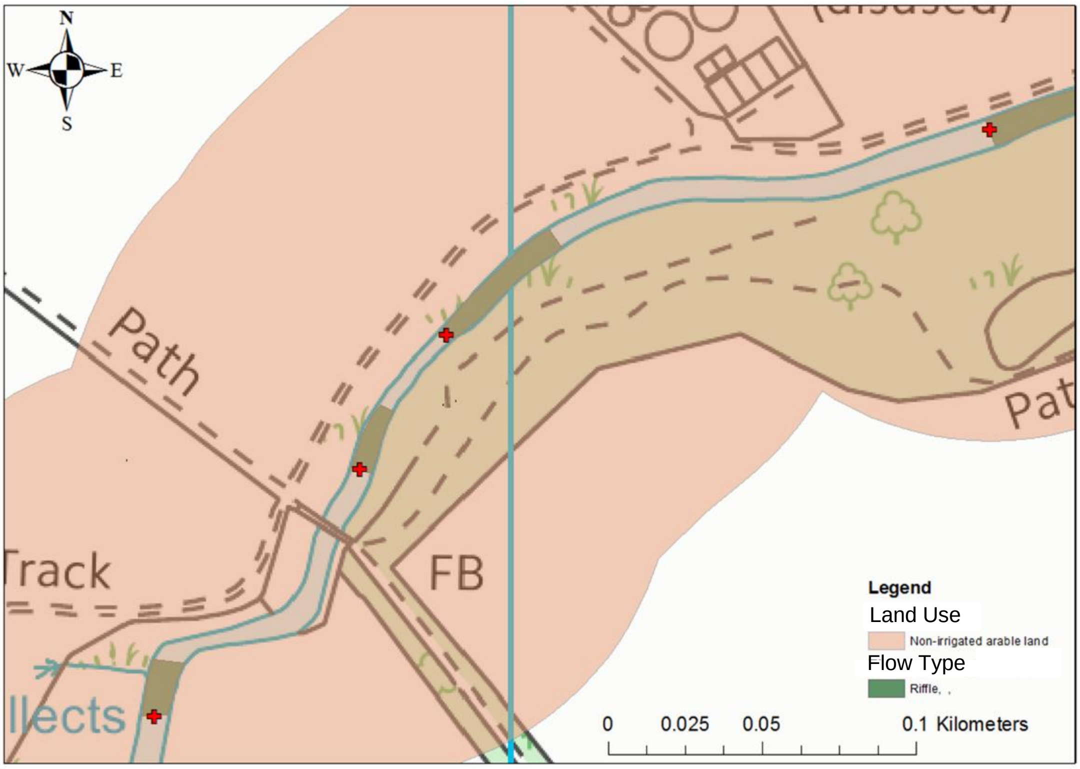


Figure 9: Sample area where riffle flow type, taken from catchment walkover, intersects non-irrigate arable land diffuse inputs from CORINE land use data, highlighted by the red cross. © Crown Copyright and Database Right (2016). Ordnance Survey (Digimap Licence).

National grid references for all sample sites are shown in the Table 6 below, with one grid reference available for the high phosphorus loading sites taken at the riffle sampling area. Sampled pools are the nearest downstream pool from sampled riffle.

Table 6: National Grid References (NGR) for all sample sites along the river Almond. with designated phosphorus loading (high or low) as determined by ArcGIS and CORINE data.

Sample Site	Phosphorus Loading	National Grid Reference
Seafield #1	High	NT 01630 66007
Seafield #2	High	NT 01376 65854
Seafield #3	High	NT 00945 65717
Seafield #4	High	NT 22385 66438
Newbridge	High	NT 11423 72410
Kirkliston	High	NT 12469 74181
Source	Low	NS 87898 62473
Harthill #1	Low	NS 89469 63408
Harthill #2	Low	NS 89763 63759
Polkemet	Low	NS 92483 64834
Almondell	Low	NT 09648 69566
Lin's Mill	Low	NT 10457 70599

High Phosphorus Sample sites

Seafield

Figure 10a: Riffle sampled at Seafield sample site #1.

Figure 10b: Riffle sampled at Seafield sample site #2.

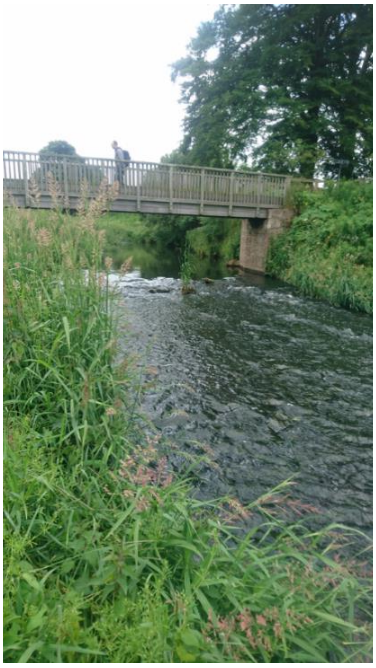


Figure 10c: Riffle sampled at Seafield sample site #3.

Figure 10d: Riffle sampled at Seafield sample site #4

Four samples were taken around the Seafield area (493m minimum distance) as not only is it the third highest contributing sub catchment of diffuse phosphorus pollution, as derived from coefficient and CORINE land use data, there is also large amounts of non-irrigated arable land. At all sampled riffles significant growth of filamentous algae and moss was evident. Figures 10a, 10b, 10c and 10d, detail the sampled riffles.

Newbridge

Figure 11: Sampled riffle at Newbridge.

Figure 11 above details the large riffle present at Newbridge with shallow depth at point bar becoming deeper further towards outside edge of the bend. The pooled area sampled below this riffle was very deep so to gain a sample, sampling took place close to near side point bar.

Kirkliston

Figure 12: Sampled riffle at Kirkliston.

The sample site located at Kirkliston (Figure 12) contained large sections of brick wall that provides an artificial habitat for invertebrates to inhabit. During the walkover survey of the river, the riffle appeared larger in length due to the increased winter flow conditions. During summer conditions, the appearance of a riffle below a slower moving section of water similar to that caused by a small weir. The depth of the riffle is shallow with deeper sections persisting towards the far bank. Dispersed between the artificial brick wall was smaller substrate such as cobble and gravel. Fish were seen jumping above the riffle, potential indicator of good spawning habitat.

Low Phosphorus Sample Sites

Source

Figure 13: Sampled riffle at the source of the river Almond.

In the upper section of the catchment the river is small (around 1 meter in width) with a high percentage of gravel present with low water depth (Figure 13).

Harthill

Figure 14a: Harthill sample site riffle #1.

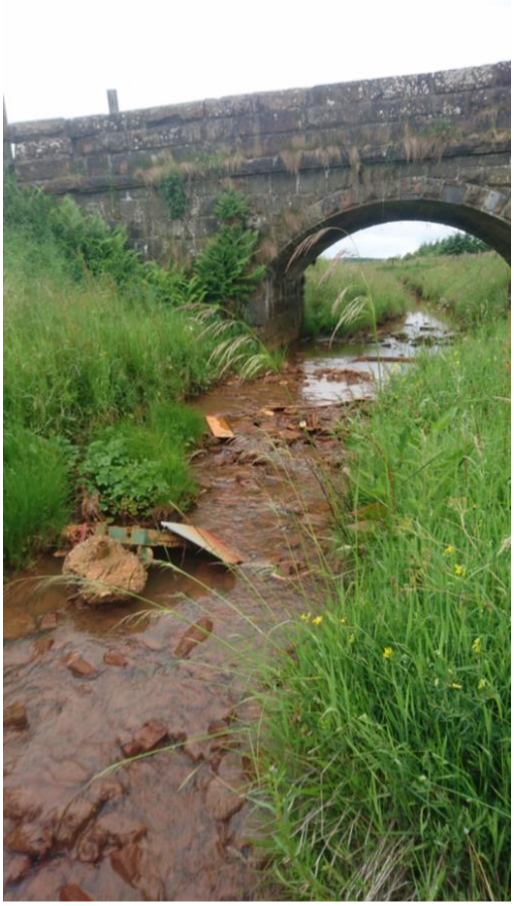


Figure 14b: Harthill sample site riffle #2.

Figures 14a and b show the riffles sampled in the Harthill area under low diffuse phosphorus inputs as determined by CORINE. During the walkover carried out in phase II of the project, it was noted that some areas of the river had potential for mine leach into the river as some discolouration was present. It was however not to the extent evident in the figures above and instead evidence of this was seen further downstream at Whitwell.

Polkemmet Country Park

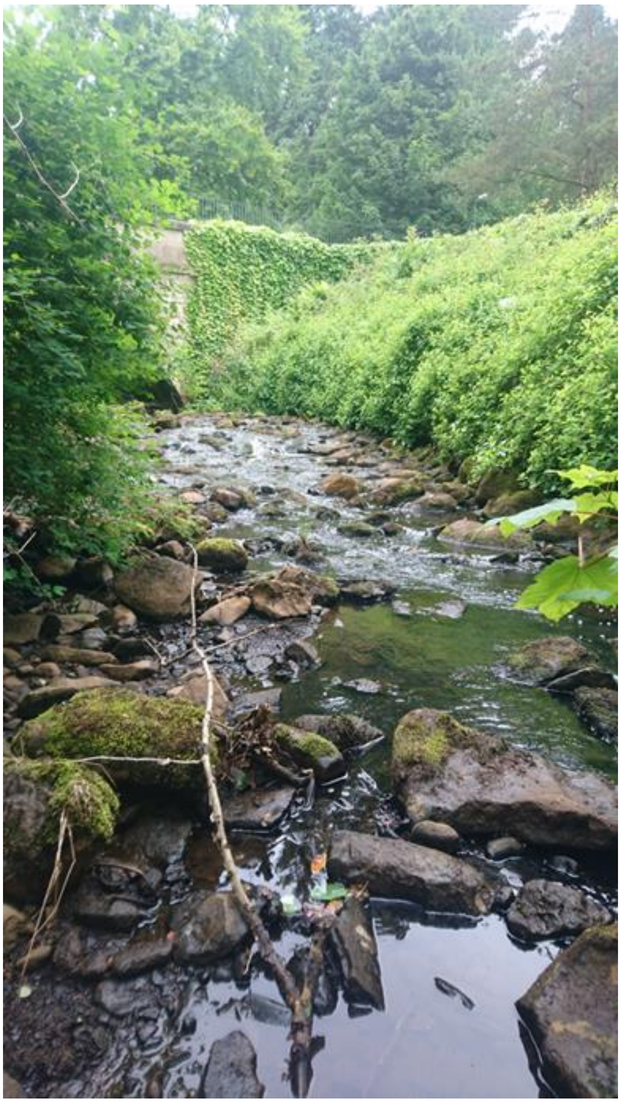


Figure 15: Sampled riffle at Polkemmet Country Park.

The sampled site at Polkemmet Country Park shown above (Figure 15) contains large amounts of cobble and boulder with the banks completely overgrown by vegetation. Water was relatively clear with little turbidity and although there are a number of poached footpaths adjacent to the river, this section of river had some bank modifications to inhibit the proximity visitors could get to the river banks. This riffle is located to a low diffuse phosphorus input land use as indicated by CORINE land use data.

Almondell Plantation

Figure 16: Riffle sampled at Almondell.

The sample riffle at Almondell consisted of large levels of boulder and cobble with some gravel and pebbles dispersed inbetween. This sample site (Figure 16) is located in an area that was limited for walkover data capture due to access and safety issues. Because of this, use of the CSO database supplied by the SEPA GIS team, it was determined that this sample site was under no immediate influence from point source pollution, like all other sample sites in this study.

Lin's Mill

Figure 17: Riffle sampled at Lin's Mill.

The sample site at Lin's Mill (Figure 17) contained large proportions of bedrock with some gravel present with little growth of algae. The flow of water slowed below the riffle as a large drop off or natural weir is present causing some impoundment. The Union Canal flow crosses the river at this point via an aqueduct and some evidence of water falling into the river from the canal was seen. Relatively little or no poaching was present due to the remoteness of the site, a small road is present leading to a house on the banks of the Almond.

2.3.2 Field Sampling

Sampling took place from the 21/06/2016 – 03/08/2016 on four separate days mainly due to weather conditions and availability of equipment and vehicles. Eighteen samples were taken in total, one sample at each riffle for high and low diffuse areas. Six more samples were taken from pooled areas below each of the sampled riffles at high input sites in order to ascertain whether differing flow types alter the impacts that diffuse pollution has on invertebrate communities. Three-minute kick samples were taken for each sample along with a one-minute hand search of the area including vegetation and substrate with methodologies complying with those outlined by SEPA (Scottish Environment Protection Agency 2001). Samples were collected using a 1mm fine mesh net and placed within a sealed plastic container for transport to Edinburgh Napier University where preservation in 70% IMS and sorting was undertaken within three days of samples being collected.

In order to allow prediction of expected communities in the absence of pollution impacts, chemical and physical parameters required for RICT predications were also collected. Details of these are shown in Table 7.

Table 7: Physical and chemical parameters collected at each sample site with the equipment or instrument used for collection.

Physical/ Chemical Parameter	Equipment/ Instrument
Conductivity (μS)	Hanna instruments conductivity measurer
Dissolved oxygen (%)	YSI 95 dissolved oxygen meter
Depth (m)	Meter ruler
Flow rate (cm/s)	MJP Geopacks flow rate measurer
National Grid Reference	GPS or Ordnance Survey
Percentage substrate	Visual Assessment
Slope (%)	GIS or Ordnance Survey
Width (m)	Nikon Coolpix range finder

2.3.3 Macroinvertebrate Identification

Collected samples were sieved to remove large pieces of substrate, vegetation or silt and placed into sorting trays for identification and preservation. Invertebrate family level identification and enumeration identification using standard freshwater keys took place with the use of a binocular dissection microscope with 40x magnification.

2.3.4 River Invertebrate Classification Tool

The current web-based version of RICT was used to derive predictions of taxa and indices based upon physical and chemical parameters in Table 7 above. Settings used for RICT analysis included: 10000 iterations, ASPT, N-Taxa, BMWP, PSI, LIFE calculations, predict and classify run type, summer sampling season and taxonomic group level 4 Great Britain identification.

2.3.5 Analysis of Data

Biological Monitoring and Working Party, Number of Taxa and Average Score Per Taxon

These indices were calculated in order to act as an indicator of general impacts upon the river. Calculation of BMWP was carried out for each sample site by a summing of the scores of the taxa present in each sample. Individual taxa were given a score from 1-10, 1 being a more common or tolerant organism and 10 being a less common or intolerant organism. The rating is assigned as per the table provided in appendix 4 and is calculated by the sum of these scores. N-taxa was calculated by determining the total number of scoring taxa present in each sample. Average Score Per Taxon (ASPT) is calculated using the formula below with example

$$\text{Average Score Per Taxon (7.31)} = \frac{\text{BMWP (95)}}{\text{N} - \text{Taxa (13)}}$$

Whalley, Hawkes, Paisley & Trigg (WHPT)

WHPT developed by Paisely et al. (2014), like other metrics mentioned above, is calculated using family level macroinvertebrate data and is based upon the BMWP scoring system but differs in that scores are dependent upon abundance of each scoring taxa rather than just which taxa are present (Everall et al. 2017). It is primarily developed to fulfil the shortfalls of the BMWP method and provide a more comprehensive reflection of the macroinvertebrate community present at a site. WHPT ASPT is derived from the following equation:

$$WHPT\ ASPT = \frac{Sum\ AB}{WHPT\ N - Taxa}$$

Sum AB: value for each taxon according to abundance (derived from table 8 below and Appendix 5)

WHPT ASPT: Number of taxa in sample

Table 8: Log abundance categories for WHPT calculation.

Abundance category	Log Abundance
AB1	1-9
AB2	10 – 99
AB3	100 – 999
AB4	>1000

Lotic-invertebrate Index for Flow Evaluation

LIFE was calculated to allow an assessment of the extent on which flow-related variation was influencing the communities found. The LIFE index was developed by Extence et al. (1999) and is based upon commonly recognised flow associations of macroinvertebrates, the Environment Agency's estimated abundance data and flow scores (f_s) of different abundance categories combined with the taxa associated with flow table. All Tables below and the LIFE formula are taken from Extence et al. (1999). The relationship between flow groups and flow association are outlined below with the mean current velocity of each flow association also displayed below in Table 9. The designated flow group for each macroinvertebrate family are shown in Extence et al. (1999).

Table 9: Freshwater macroinvertebrate flow groups, flow association and current velocities.

Group	Ecological flow association	Mean current velocity (cm s^{-1})
I	Taxa primarily associated with rapid flows	Typically >100
II	Taxa primarily associated with moderate to fast flows	Typically 20–100
III	Taxa primarily associated with slow or sluggish flows	Typically <20
IV	Taxa primarily associated with flowing (usually slow) and standing waters	-
V	Taxa primarily associated with standing waters	-
VI	Taxa frequently associated with drying or drought impacted sites	-

Table 10: FS score for abundance categories with flow groups and taxa with associated flow.

Flow Groups	Abundance categories			
	A	B	C	D/E
I Rapid	9	10	11	12
II Moderate/ fast	8	9	10	11
III Slow/ sluggish	7	7	7	7
IV Flowing/ standing	6	5	4	3
V Standing	5	4	3	2
VI Drought resistant	4	3	2	1

Table 11: Abundance categories for macroinvertebrates.

Category	Estimated abundance
A	1 - 9
B	10 - 99
C	100 - 999
D	1000 - 9999
E	10000 +

Using the abundance categories present in Table 11, a score can be obtained from Table 10 dependent upon which flow group and abundance category that organism is placed.

From this, the scores can be placed into the LIFE equation below to provide a LIFE score for that sample or site.

$$\mathrm{LIFE} = \frac{\sum \mathrm{fs}}{\mathrm{n}}$$

$\sum$fs: the total sum of taxon flow scores of the whole sample.
n: the number of taxa used to calculate $\sum$fs.

This metric was applied to the riffle and pool sites that are under high diffuse input in order to establish the extent to which the diffuse impacts are having an effect on the macroinvertebrate communities in relation to flow.

PSI Index

The PSI index (Proportion of Sediment-sensitive Invertebrates) was developed by Extence et al. (2011b) for the purpose of assessing the impact of sediment loading on the macroinvertebrate community. This method was applied to all sites sampled in order to assess the levels of diffuse fine sediment inputs present along the riparian corridor with a particular focus on differences between high diffuse input sites and low diffuse input sites. Tables and formulas presented below are taken directly from Extence et al. (2011b) and follows a similar theme used to the create and test the LIFE metric. Designations for what FSSR group macroinvertebrates represent can be found in Extence et al. (2011a).

The PSI score gives a representation of sediment sensitive taxa present within the sample and is calculated by using Table 12 and 13 as well as formula detailed below.

Table 12: PSI percentage and the interpreted river bed condition.

PSI	River bed condition
81 – 100	Minimally sedimented/ unsedimented
61 – 80	Slightly sedimented
41 – 60	Moderately sedimented
21 – 40	Sedimented
0 - 20	Heavily sedimented

Table 13: FSSR (Fine Sediment Sensitivity Rating) outline with Log abundance categories.

Group	FSSR	Log Abundance			
		1 – 9	10 - 99	100 - 999	1000 +
A	Highly sensitive	2	3	4	5
B	Moderately sensitive	1	2	3	4
C	Moderately insensitive	1	2	3	4
D	Highly insensitive	2	3	4	5

$$\text{PSI}\ (\Psi) = \frac{\sum \text{Scores for Sediment Sensitivity Groups A \& B}}{\sum \text{Scores for all Sediment Sensitivity Groups A; B; C \& D}} \times 100$$

The PSI index values give an indication of sedimentation effects at different sites, which may also be expected to correlate with diffuse inputs given the role of sediment in transporting nutrients to the river. This will also be utilised to assess the differences between low and high level diffuse input sites and help to quantify the amount of diffuse inputs into the river Almond.

Ecological Quality Ratio

The observed values for ASPT, N-Taxa, PSI index, Life and WHPT indices obtained from each site were compared to expected values calculated by RICT or from the Biotic package in R (Briers, 2016) to derive Ecological Quality Ratio (EQR) and is calculated using the formula below:

$$\text{Ecological Quality Ratio} = \frac{\text{Observed Index Score of Sampled Site}}{\text{Predicted Index Score Derived by RICT}}$$

The status boundary for each index is shown in the tables below.

Table 14: EQR boundaries for WHPT N-Taxa and WHPT ASPT (WFD-UKTAG, 2014).

Status Boundary	WHPT N-Taxa EQR	WHPT ASPT EQR
High/Good	0.80	0.97
Good/ Medium	0.68	0.87
Medium/Poor	0.56	0.72
Poor/Bad	0.47	0.53

Table 15: EQR values for BMWP, ASPT and N-Taxa quality thresholds (Scottish Government 2009).

Quality	EQR BMWP	EQR ASPT	EQR N-Taxa
High	0.80	0.97	0.85
Good	0.60	0.86	0.71
Moderate	0.40	0.75	0.57
Poor	0.20	0.63	0.47

Nutrient Analysis

Water samples, stored in a 150ml polypropylene bottle, were taken at each sample site on the day of kick sampling. There kept cool in a fridge on return to the laboratory and were analysed using a Seal AQ2+ analyser to assess the levels of dissolved nitrogen (nitrate, nitrite and ammonia) and soluble reactive phosphorus within the sample, using standard colorimetric techniques (APHA, 2012). Statistical analysis in the form of T-tests and Mann-Whitney U were performed on the nutrient values to test for significances between high phosphorus riffle and pool sites as well as high phosphorus input sites and low phosphorus input sites.

3.0 Results

Owing to the availability of nitrogen loading coefficients only for a limited range of very broad CORINE land-use types, the main results were based solely on phosphorus loadings. Nitrogen data were still calculated and are displayed in appendix 2 and 3.

3.1 Catchment Level Total Phosphorus Loading

Figure 18 shows the predicted total loading of phosphorus based on the land use in different years.

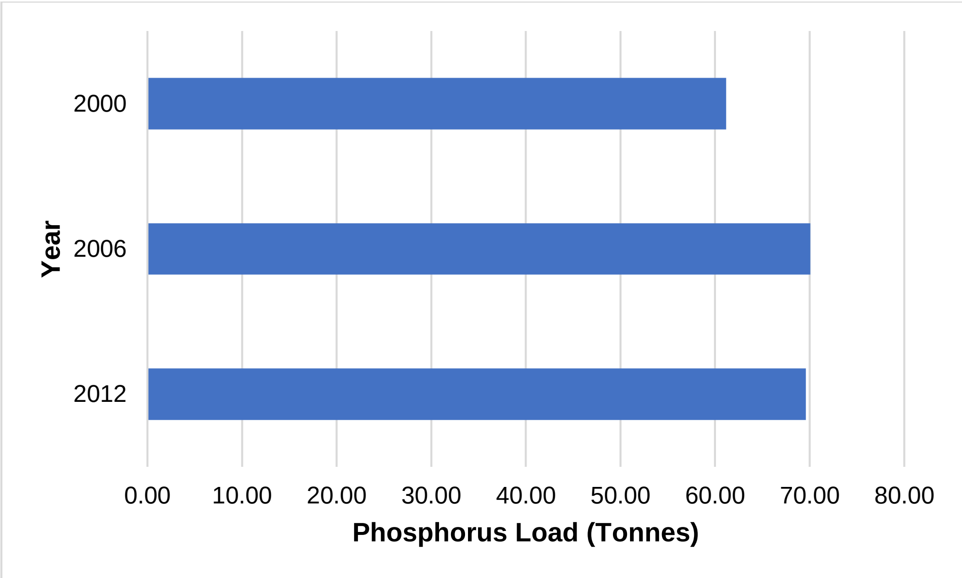


Figure 18: Total phosphorus loadings (tonnes) per year for 2000, 2006 and 2012. Phosphorus export is displayed in tonnes per year and is derived from CORINE Land use data and coefficient data from Smith et al. (2005).

Table 16 below shows data for the total phosphorus load to each designated Almond sub catchment. Loadings vary from a very low score (0.00) tonnes per year at Niddry Burn, to 14.6 tonnes per year at Brox Burn – Ryal Burn Confluence to River Almond. Thirteen out of the 21 sub-catchments contribute over 1 tonne of phosphorus per year whilst seven contribute under half a tonne per year, as shown in Table 16 below. P loadings per hectare show that River Almond – Foulshiels Burn to Breich Water Confluence sub-catchment has the highest P contribution per hectare whilst Drainage Ditch Upstream of Cobbinshaw Reservoir has the lowest.

Table 16: Total phosphorus loading and loading per hectare from sub-catchments within the Almond catchment by using data deriving the catchment boundaries with information supplied by SEPA (2010), phosphorus export is displayed in tonnes per year and is derived from CORINE Land use data and coefficient data from Smith et al. (2005). Sub-catchments are ranked from highest to lowest, with highest contributors at the top of the column.

SEPA Designated Sub catchment	P Loading (Tonnes per Year)	Sub-Catchment Area (Ha)	P Loading (Tonnes per Ha)
Brox Burn - Ryal Burn Confluence to River Almond	14.61	5008.13	0.0029
Gogar Burn - Union Canal to River Almond	11.15	2987.62	0.0037
River Almond - Breichwater Confluence to Cramond	7.64	2022.52	0.0033
Gogar Burn - Source to Union Canal	6.22	2306.05	0.0031
River Almond - Maitland Bridge to Crammond	5.04	1513.17	0.0033
River Almond - Source to Foulshiels Burn Confluence	4.82	5072.47	0.0010
Breich Water/ Darmead Linn	4.19	3427.76	0.0012
Killandean Burn/ Harwood Water	3.23	2495.90	0.0013
Lochshot Burn	2.84	1518.73	0.0019
Murieston Water	2.57	4391.38	0.0006
Linhouse Water, Camility Burn, Green Burn	1.99	1403.39	0.0014
Darmead Linn	1.91	2128.04	0.0009
Bog Burn	1.62	1792.46	0.0009
Brox Burn - Wester Tartraven to Ryal Burn Confluence	0.66	1403.39	0.0043
Union Canal - Craigton to Murrayburn	0.42	249.21	0.0017
Foulshiels Burn/ Bickerton Burn	0.34	190.97	0.0018
How Burn	0.32	353.92	0.0009
Cobbinshaw Reservoir	0.01	12.79	0.0009
Drainage Ditch - Upstream of Cobbinshaw Reservoir	0.00	1.11	0.0001
River Almond - Foulshiels Burn to Breich Water Confluence	0.00	0.03	0.0049
Niddry Burn	0.00	0.00	0.0000

The main stem of River Almond intersects the following sub-catchments, shown in Table 17.

Table 17: The main sub-catchments the river Almond intersects, as outlined by SEPA, ranked with highest contributing sub-catchment at the top of the column.

SEPA Designated Sub catchment
Brox Burn - Ryal Burn Confluence to River Almond
Gogar Burn - Union Canal to River Almond
River Almond - Maitland Bridge to Cramond
River Almond - Source to Foulshiels Burn Confluence
Linhouse Water, Camility Burn, Green Burn

The total phosphorus export to the river Almond for each sub-catchment are displayed below in Figure 19, with the Linhouse Water, Camility Burn Green Burn sub-catchment contributing the lowest amount of phosphorus to the catchment with less than 2 tonnes per year, as derived by the coefficient method previously detailed, and Brox Burn – Ryal Burn Confluence to River Almond sub-catchment contributing the most with over 14 tonnes per year. River Almond – Source to Foulshiels Burn and River Almond - Maitland Bridge to Crammond both contribute over 4 tonnes of phosphorus per year respectively whilst Gogar Burn - Union Canal to River Almond contributes just under 11 tonnes per year.

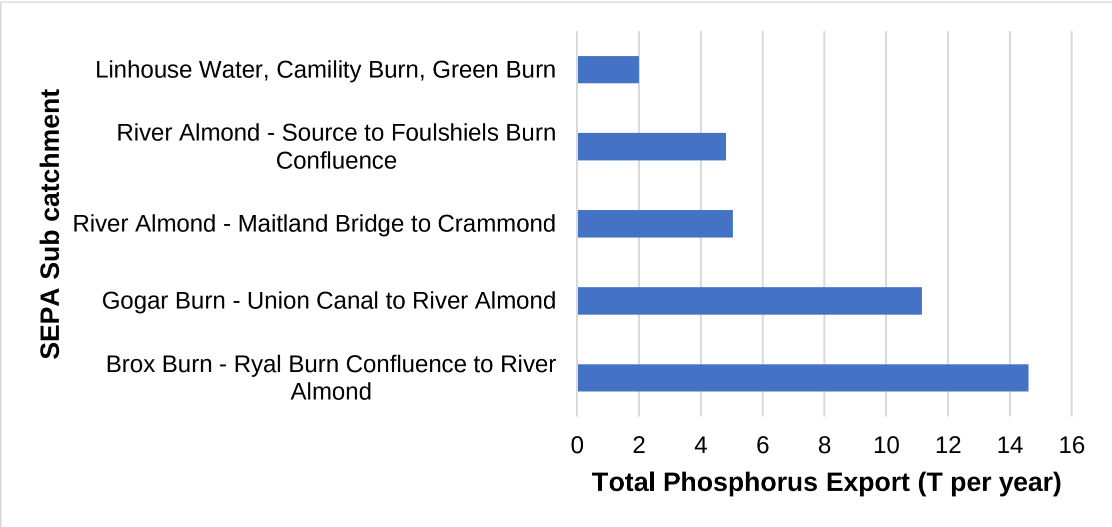


Figure 19: Graph showing the levels of phosphorus export from the five main sub-catchments that the main stem of the Almond intersects within the Almond catchment. The sub-catchments are ranked from highest contributor to lowest, phosphorus export is displayed in tonnes per year and is derived from CORINE Land use data and coefficient data from Smith et al. (2005).

3.1 Riparian Corridor Phosphorus Loadings

Riparian Corridor - 25m and 50m

Sections 1 and 2 contribute the largest levels of phosphorus with over 0.08 tonnes of phosphorus per year entering the river, whilst section 8 contributes the lowest with 0.009 tonnes per year at the 25m corridor level. At the 50m riparian corridor level, section 1 and 2 also contribute the highest whilst section 8 is lowest contributor. At the 25m and 50m distance the average P load per year is 0.043 and 0.086 tonnes per year respectively. Figure 20 below shows the total P load in tonnes per year per section.

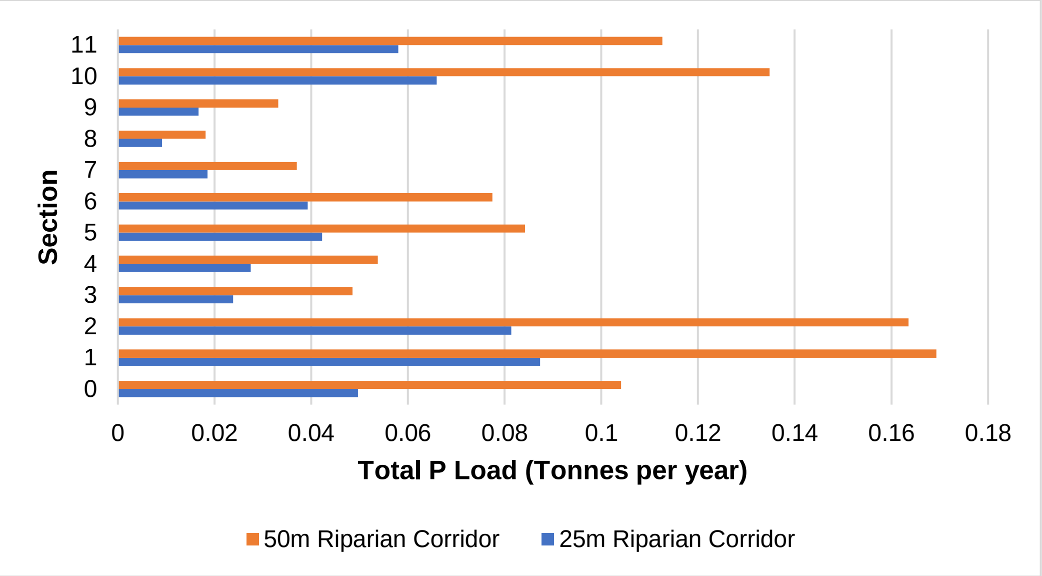
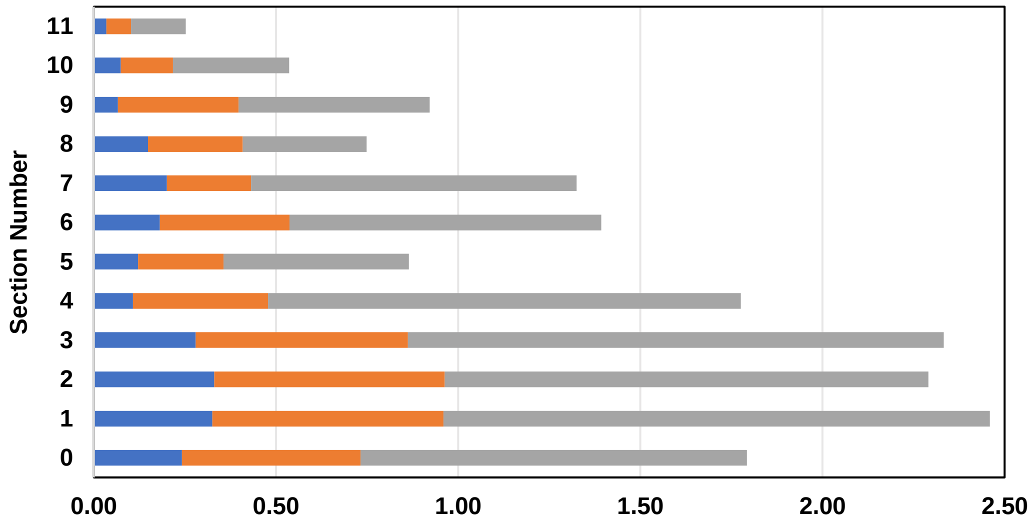


Figure 20: Total phosphorus load in tonnes per year for 25 and 50m riparian corridor with 12 equal sections. Data was calculated using coefficient data from Smith et al. (2005) and land use data from CORINE European Environment Agency (2016). Sections are ordered from sea to source with 0 at sea and 11 being at source. Total phosphorus contribution at 25m is 0.52 Tonnes per year and at 50m 1.04 Tonnes per year.

Riparian Corridor – 100m

The figure and table below provide values of total P export per section per year for the 100, 200 and 500m riparian corridor scales.



	0	1	2	3	4	5	6	7	8	9	10	11
100m	0.24	0.33	0.33	0.28	0.11	0.12	0.18	0.20	0.15	0.07	0.07	0.03
200m	0.49	0.63	0.63	0.58	0.37	0.23	0.36	0.23	0.26	0.33	0.14	0.07
500m	1.06	1.50	1.33	1.47	1.30	0.51	0.86	0.89	0.34	0.52	0.32	0.15

Total Phosphorus Export (Tonnes per year)

Figure 21: Total phosphorus load in tonnes per year for 100, 200 and 500m riparian corridor with 12 equal sections. Data was calculated using coefficient data from Smith et al. (2005) and land use data from CORINE European Environment Agency (2016). Sections are ordered from sea to source with 0 at sea and 11 being at source. Total phosphorus contribution at 100m is 2.11 tonnes per year, 200m is 4.33 tonnes per year and at 500m, is 10.25 tonnes per year.

At the 100m riparian corridor distance from the river sections 2 and 1 have the highest phosphorus loading with over 0.30 tonnes of phosphorus per year entering the river. Sections 9, 10, and 11 contribute the lowest amount of phosphorus to the river, under 0.10 tonnes per year, with section 11 contributing the least as shown in Figure 21. Average phosphorus loading per section per year is 0.18 tonnes.

Riparian Corridor – 200m

When the riparian corridor is increased to 200m, sections 1 and 2 increases to over 0.60 tonnes of phosphorus per year with section 3 just below 0.60 at 0.58 tonnes of phosphorus per year. Section11 is still the lowest contributor with a loading of 0.07 tonnes of phosphorus per year as shown in Figure 21 above. The average phosphorus loading per section per year is 0.36 tonnes.

Riparian Corridor – 500m

Figure 21 above shows results of total phosphorus loading per section when the riparian corridor is increased to 500m from the river. Section 1 is the largest phosphorus contributor with 1.5 tonnes per year with section 0,2, 3 and 4 also contributing above 1 tonne per year – 1.06, 1.33, 1.47 and 1.3 respectively. Section 11 has the lowest contribution with 0.15 tonnes of phosphorus per year with sections 9 and 10 having phosphorus contributions of under 0.5 tonnes per year. The average contribution for each section is 0.85 tonnes of phosphorus per year.

3.2 Catchment Walkover Digitising

Features recorded during the walkover of the main stem of the river Almond are detailed in Table 18 and includes the features recorded with the total or count value of each feature.

Table 18: length or count data for features recorded from digitised maps of the river Almond walkover with feature class and type data.

Feature Class	Feature	Type	Count/Length (km)/ Area (km^2)
Flow Type	Broken Water	Area	68
	Cascade	Area	31
	Chaotic	Area	1
	Chute	Area	1
	Exposed Bedrock	Area	8
	Free Fall	Area	2
	Glide	Area	125
	Impoundment	Area	26
	No Perceptible Flow	Area	14
	Pool	Area	172
	Riffle	Area	242
	Run	Area	293
In-Channel Features	Boulder	Area	26
	Debris Dam	Point	5
	Erosion	Polyline	8.86
	Ford	Point	1
	Island	Point	22
	Large Woody Debris	Point	221
	Mid Channel Bar	Area	71
	Point Bar	Area	32
	Reinforcement	Polyline	15.13
	Side Bar	Area	76
Modification	Leves	Polyline	3.04
	Overhanging Vegetation	Polyline	52.59
	Poaching	Polyline	3.91
	Sand Bank	Polyline	0.40
	Weir	Point	59
Pollution Inputs	Diffuse Pollution Source	Point	602
	Point Source	Point	132
	Mine Leach	Point	13

The most common flow type recorded was 'Run' with the least common being 'Chutes' and 'Chaotic'. 'Erosion' was the most common channel feature with only one 'Ford' being recorded as the least common. 'Reinforcement' is present in large amounts along the river Almond as well as large amounts of 'Weirs' that are natural or artificial. 602 'Diffuse' input points were recorded as well as 132 'Point Source' points and 13 'Mine Leach' points which together with 'Erosion' and 'Poaching' in contributing to diffuse pollution inputs and potential nutrient loadings.

3.2.1 Association between Walkover Features and land-use loadings.
Associations between the frequency or extent of diffuse inputs recorded from the walkover survey and predicted nutrient loadings were examined by calculating Pearson correlations between nutrient loading of each river section and the count (points) or extent (line/area features) of diffuse inputs. This was repeated for each riparian corridor width (25, 50, 100, 200 and 500m).

Diffuse Points

From the digitised maps it highlights that there are 602 diffuse points along the main stem of the Almond. However, no distinct or consistent change occurs in frequency as you move downstream from the source of the river, although highest totals are recorded in the lower reaches. Figure 22 below details the number of diffuse points per designated section.

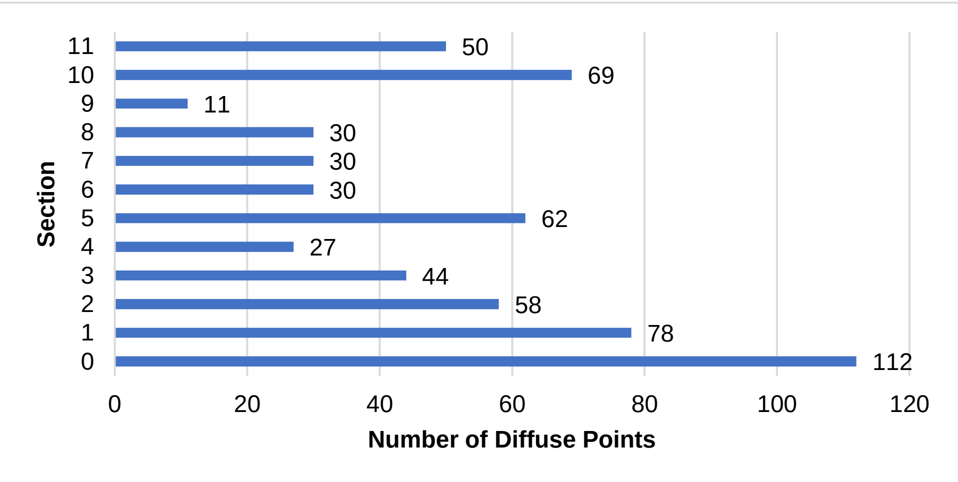


Figure 22: Number of diffuse pollution points per section as derived from digitised walkover.

Pearson's Correlations were calculated in order to assess whether phosphorus loadings per section correlate with number of diffuse points within that section, results are displayed in Table 19 below; there were significant correlations found at the 25m and 50m riparian corridor distances whilst no significant correlations found at any other distance perhaps suggesting that the land closest to the river has the greatest influence on nutrient input.

Table 19: Pearson's correlation results for total phosphorus loadings at different riparian corridor distance with number of diffuse points per section.

Riparian Corridor Distance	R Value/Correlation Strength	P-Value
25m	0.652	0.022
50m	0.674	0.016
100m	0.391	0.209
200m	0.286	0.367
500m	0.196	0.541

Erosion and Poaching

Field observations regarding erosion and poaching were recorded during the walkover of the Almond and by digitising this allows for total length of these features to be calculated. Statistical analysis was undertaken to assess whether phosphorus export per section correlated with length of bank erosion recorded as well as phosphorus export with poaching and the results of these are shown in Table 20 below.

Table 20: Pearson's correlation results for total phosphorus loadings at different riparian corridor distances with length of erosion and poaching per section.

Riparian Corridor Distance	R Value/ Correlation Strength		P-Value	
	Erosion	Poaching	Erosion	Poaching
25m	-0.037	-0.057	0.909	0.859
50m	-0.029	-0.077	0.929	0.812
100m	-0.355	0.059	0.257	0.855
200m	-0.237	0.089	0.458	0.782
500m	-0.096	0.054	0.766	0.867

3.3 River Invertebrate Classification Tool

Utilising RICT predictions based on physical and chemical characteristics from each site allowed for comparisons to be made between sampled data and expected data. These were then averaged as the sample sites form replicates of the categories they belong to, for example, riffles sampled Seafield #1, 2, 3, 4, Newbridge and Kirkliston form the broad category of high phosphorus inputs and Source, Harthill #1 and 2, Polkemet, Almondell and Lin's Mill form category of low phosphorus inputs, as well as distinguishing between flow type within the high phosphorus input category, riffle and pool. This is shown in Table 21 below.

Table 21: Average River Invertebrate Classification Tool predicted values of BMWP, ASPT and N-Taxa for areas of high and low phosphorus input and with different flow conditions.

	RICT Prediction (Average) High Phosphorus Inputs		**RICT Prediction (Average) Low Phosphorus Inputs**
Indices	Riffle	Pool	Riffle
BMWP	122.68	126.32	120.81
ASPT	5.81	5.85	5.78
N-taxa	21.00	21.46	20.89

3.4 Water Quality Classification

BMWP, ASPT, N-taxa, WHPT ASPT and WHPT N-taxa scores were converted to EQR values and compared with the quality bandings for WFD quality assessment outlined by the Scottish government as shown in Tables 22 & 23 below. Appendix 7 provides the number and family of invertebrate taxa data found at each sample site.

Table 22: Ecological Quality Ratio boundaries for WHPT N-Taxa and WHPT ASPT (WFD-UKTAG, 2014).

Status Boundary	WHPT N-Taxa EQR	WHPT ASPT EQR
High/Good	0.80	0.97
Good/ Medium	0.68	0.87
Medium/Poor	0.56	0.72
Poor/Bad	0.47	0.53

Table 23: Ecological Quality Ratio values for BMWP, ASPT and N-Taxa quality thresholds (Scottish Government 2009).

Quality	EQR BMWP	EQR ASPT	EQR N-Taxa
High	0.80	0.97	0.85
Good	0.60	0.86	0.71
Moderate	0.40	0.75	0.57
Poor	0.20	0.63	0.47

3.4.1 Ecological Quality Ratio (EQR) BMWP, Average Score Per Taxon, Number of Taxa, WHPT Average Score Per Taxon and WHPT Number of Taxa

High Phosphorus Inputs – Riffle Versus Pool

EQR BMWP scores for riffle sites obtained ‘good’ whilst pool sites scored ‘moderate’ in when compared to values in Table 23. EQR ASPT score for riffle sites obtained ‘high’ whilst pool sites scored ‘moderate’ when compared to values in Table 23. EQR N-taxa values for riffle sites obtained ‘good’ whilst pool sites obtained 'moderate' as designated from Table 23 above. EQR WHPT ASPT riffle sites attained a 'high/good' score whilst pools attained a 'good/medium' score. EQR WHPT N-taxa riffle sites were scored as 'medium/poor' whilst pool sites were also 'medium/poor'. EQR values for BMWP, ASPT, N-taxa WHPT ASPT and WHPT N-taxa values for flow type and input level are shown graphically in Figure 23 below. Riffle sites for BMWP, ASPT, N-taxa, WHPT ASPT and WHPT N-taxa had higher EQR values and therefore are closer to predictions made by RICT (Figure 23).

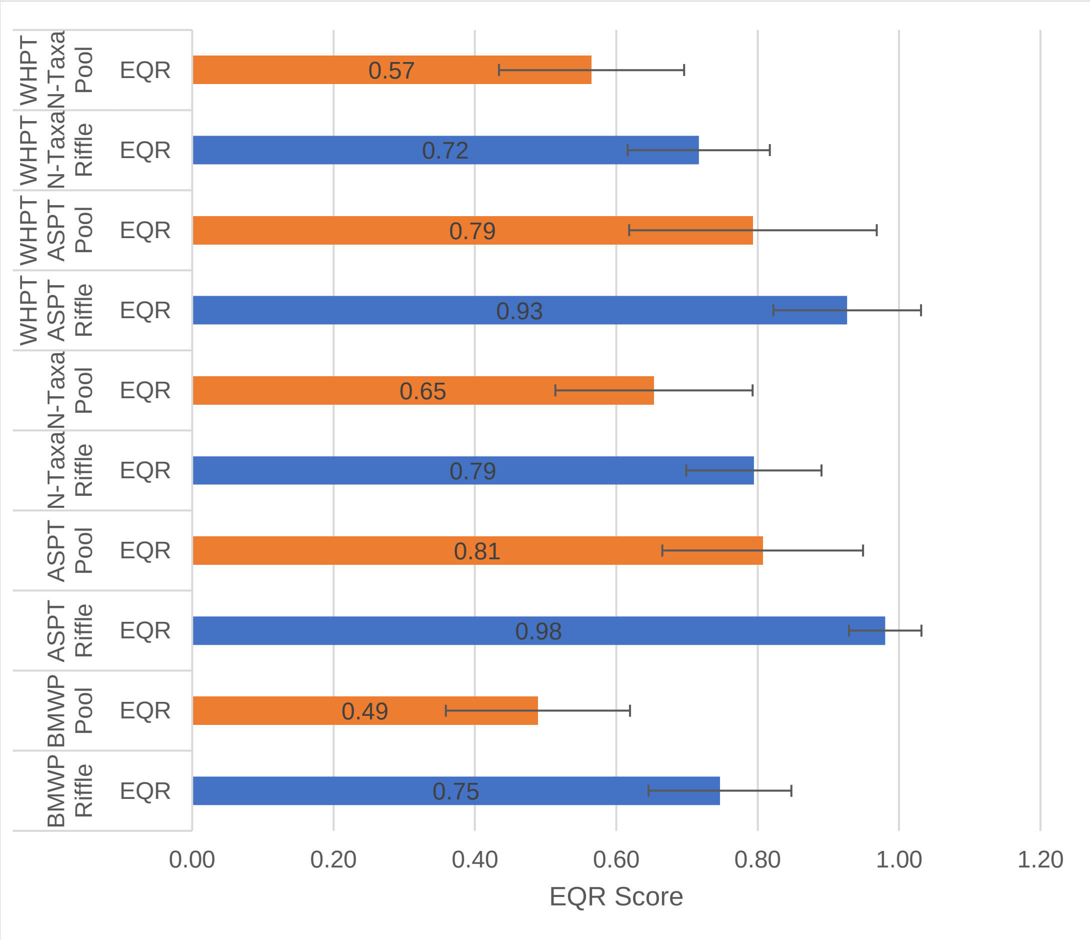


Figure 23: The Ecological Quality Ratio BMWP, ASPT, N-Taxa scores for riffle versus pool sites under high phosphorus input conditions are displayed above with error bars displaying the standard deviations of the sites. Blue bars represent riffle values and orange bars represent pool values.

Paired T-Test results below in Table 24 for EQR BMWP that there is a significant difference between riffle and pool sites (paired T-test, $t = 0.84$, $df = 6$, $P = 0.022$) and the mean was higher at riffle sites. EQR ASPT analysis indicates that there is a significant difference between rifle and pool sites (paired T-test, T= 3.22, df = 6, P= 0.023) and the mean was higher in riffle sites. EQR N-taxa values also showed no significant difference between the riffle and pool sites (paired T-test, t = 1.85, df = 6, p = 0.123) riffle sites have the higher mean. EQR WHPT ASPT values had no significant differences due to the abundance weighted metrics, a weakening of the p-value occurred compared to non-weighted metrics (paired T-test, t = 2.24, df = 6, p = 0.075). EQR WHPT N-taxa values, like EQR N-taxa, had no significant differences between riffle and pools sites but a strengthening of the p-value did occur (paired T-test, t = 2.17, df = 6, p = 0.082).

Table 24: Ecological Quality Ratio BMWP, ASPT, N-Taxa, WHPT ASPT and WHPT N-taxa values for Paired T-Test carried out on EQR values displaying mean, standard deviation and mean for high P loading riffle and pool sites. N = 6 for all samples

EQR Indices	Flow Type	Mean	Standard Deviation	T - Value	df	P-Value
BMWP	Riffle	0.750	0.163	0.84	6	0.022
	Pool	0.490	0.158			
ASPT	Riffle	0.980	0.097	3.22	6	0.023
	Pool	0.810	0.151			
N-Taxa	Riffle	0.790	0.155	1.85	6	0.123
	Pool	0.650	0.151			
WHPT ASPT	Riffle	0.930	0.104	2.24	6	0.075
	Pool	0.790	0.175			
WHPT N-Taxa	Riffle	0.720	0.100	2.17	6	0.082
	Pool	0.565	0.131			

High Phosphorus Versus Low Phosphorus Inputs

EQR BMWP values for high P and low P input sites attained a ‘good’ score (Figure 24). EQR ASPT values for high and low phosphorus input sites had a very similar score with only 0.02 between them, and both obtained a ‘high’ score when compared with data in Table 23 above, but low phosphorus input sites had the higher EQR value. EQR N-taxa data shows a large difference between high and low input sites, with high input sites obtaining a ‘good’ score and low input sites obtaining a ‘good’ score. High phosphorus input sites have a higher EQR BMWP and N-Taxa value whilst ASPT EQR values are higher at low P input levels indicating that these sites are closer to RICT predictions (Figure 24). EQR WHPT ASPT is higher at low P input sites (high/good score) whilst high P input sites scored the same. Both high and low P input sites are close to RICT predictions for WHPT ASPT which perhaps indicates the riffle flow type has a greater influence on community structure than nutrient presence. EQR WHPT N-taxa low P sites scored higher than high P sites, ‘medium/poor’ compared to ‘poor/bad’.

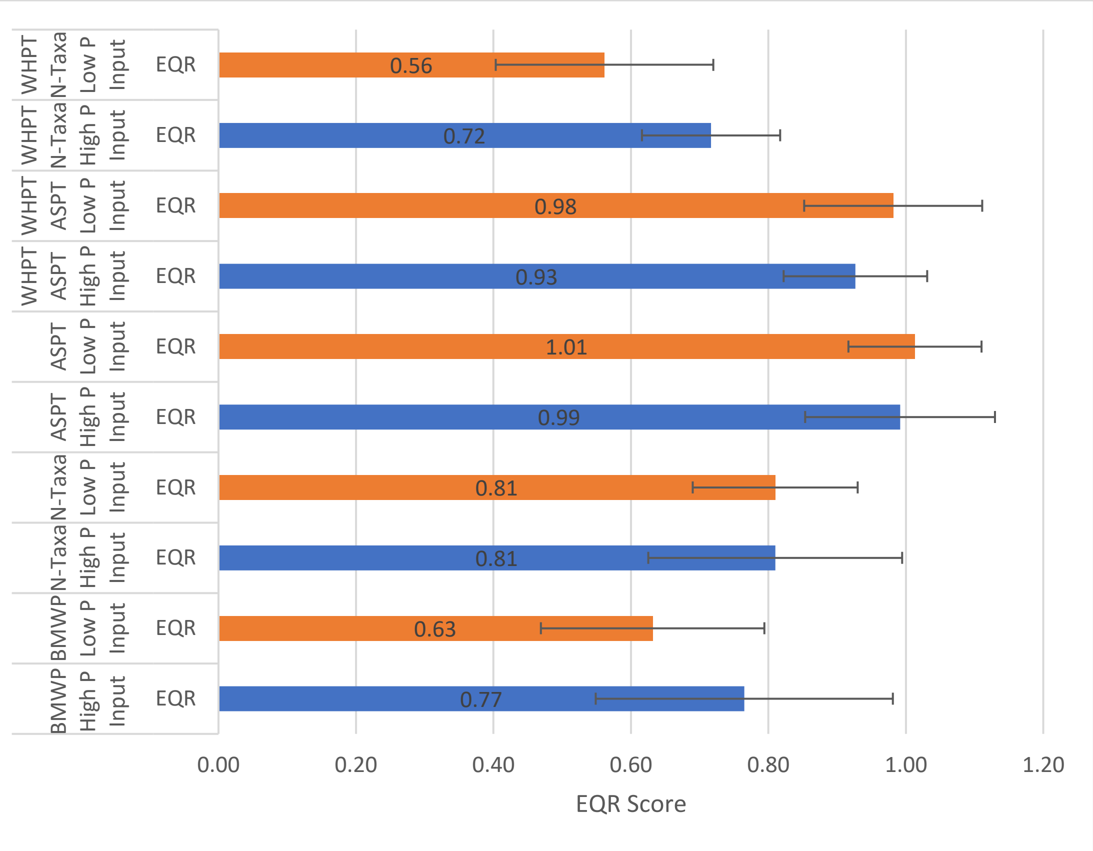


Figure 24: The Ecological Quality Ratio BMWP, ASPT, N-Taxa, WHPT ASPT and WHPT N-taxa scores for high phosphorus V’s low phosphorus riffle sites are displayed above with error bars displaying the standard deviations of the sites. Blue bars represent high P input sites and orange bars represent low P input sites.

Paired T-tests for EQR BMWP, ASPT, N-taxa, WHPT ASPT and WHPT N-taxa were carried out to determine if any significant differences persist between high and low P input sites. There was no significant difference between the means of high and low P input sites for EQR BMWP (Paired sample T-test, $t = 0.93$, $df = 6$, $p = 0.396$) high P load sites have the higher mean. No significant difference existed between EQR ASPT (Paired sample T-test, $t = 0.28$, $df = 6$, $p = 0.792$) low P input sites had the higher mean. EQR N-taxa analysis also showed no significant difference between high and low P input sites (Paired sample T-test, $t = 1.47$, $df = 6$, $p = 0.200$) whilst high P input sites also had a higher mean (Table 25). Abundance weighted WHPT calculations for both ASPT and N-taxa showed that there was still no significant difference between high and low P load for either indices used but a strengthening of p-value did occur (EQR WHPT ASPT, Paired sample T-test, t = -0.80, df = 6, p = 0.459. EQR WHPT N-taxa, Paired sample T-test, t = 1.6, df = 6, p = 0.171).

Table 25: Results from paired sample T-Test for EQR indices of BMWP, ASPT, N-Taxa, WHPT ASPT and WHPT N-taxa with mean values and standard deviation.

EQR Indices	P Load	Mean Values	Standard Deviation	T-Value	df	P-Value
BMWP	High P Load	0.770	0.088	0.93	6	0.396
	Low P Load	0.630	0.216			
ASPT	High P Load	0.990	0.096	0.28	6	0.792
	Low P Load	1.010	0.138			
N-Taxa	High P Load	0.810	0.115	1.47	6	0.200
	Low P Load	0.810	0.185			
WHPT ASPT	High P Load	0.930	0.104	-0.80	6	0.459
	Low P Load	0.980	0.123			
WHPT N-taxa	High P Load	0.720	0.100	1.60	6	0.171
	Low P Load	0.560	0.158			

3.5 Lotic-invertebrate Index for Flow Evaluation

High Phosphorus Inputs - Riffle Versus Pool

High phosphorus riffle sites achieved a higher average LIFE score than pool sites by 0.39 (7.43 riffle, 7.04 pool) and have a lower standard deviation as shown in the Figure 25 below.

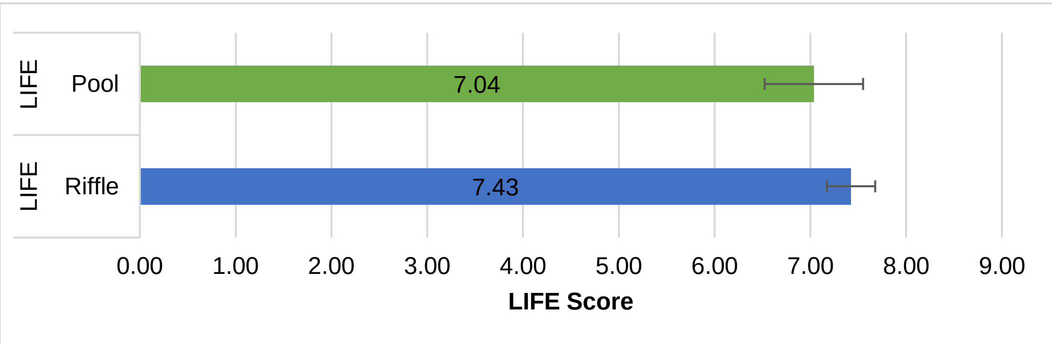


Figure 25: LIFE scores for sites that are under high phosphorus inputs with differing flow types, riffle and pool, with standard deviations displayed by error bars. Riffle LIFE score is depicted by the blue bar and pool score by the green bar.

High phosphorus pool sites achieved a higher LIFE EQR score than riffle sites (0.94 riffle, 1.04 pool) as shown in the Figure 26 below. Pool sites achieved better than predicted LIFE scores whilst riffle sites fall just short of this.

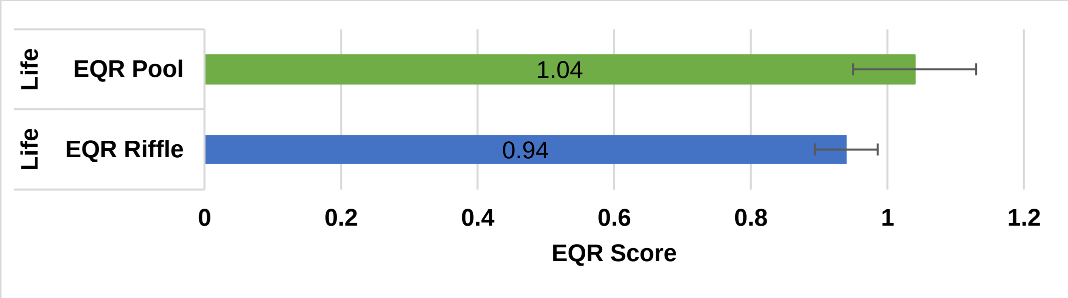


Figure 26: LIFE Ecological Quality Ratio scores for riffle and pools at high P input sites with standard deviations at error bars. Blue bar represents riffle Ecological Quality Ratio values and green represents pool values.

Analysis of LIFE EQR values indicates that a significant difference exists between riffle and pool sites (paired T-test, $t = 3.93$, $df = 6$, $p = 0.011$) with riffle sites having the higher mean.

3.6 Percentage of Sediment-sensitive Invertebrates

High Phosphorus Inputs - Riffle Versus Pool

The river bed condition of riffle and pool sites are both rated as 'moderately sedimented' but pool sites have a lower score than riffle sites indicating higher levels of sedimentation. Riffle sites have a smaller standard deviation than pool sites represented by error bars on Figure 27 below.

The EQR values for PSI indicate that riffle sites are closest to the predictions made and that pool sites are below (Figure 28). Paired T-test for EQR PSI high P input riffle and pool sites indicate that a significant difference exists (paired T-test, $t = 4.85$, $df = 6$, $p = 0.005$) with riffle sites having the higher mean.

High Phosphorus Versus Low Phosphorus Inputs

Low P input riffle sites obtained a higher PSI percentage score (Slightly Sedimented) than high phosphorus riffle sites which were 'moderately sedimented' category meaning high P input sites are more sedimented. The low phosphorus input riffle sites had a larger standard deviation, as shown in Figure 27 below.

EQR PSI results for high and low P input sites indicate that low P input sites have a higher EQR score than high P input sites (Figure 28) showing that high P input sites are closer to predictions. EQR PSI statistical analysis for high and low loads indicated that no significant difference exists (Paired T-test, t = 0.28, df = 6, p = 0.776) with low P input sites having a higher mean.

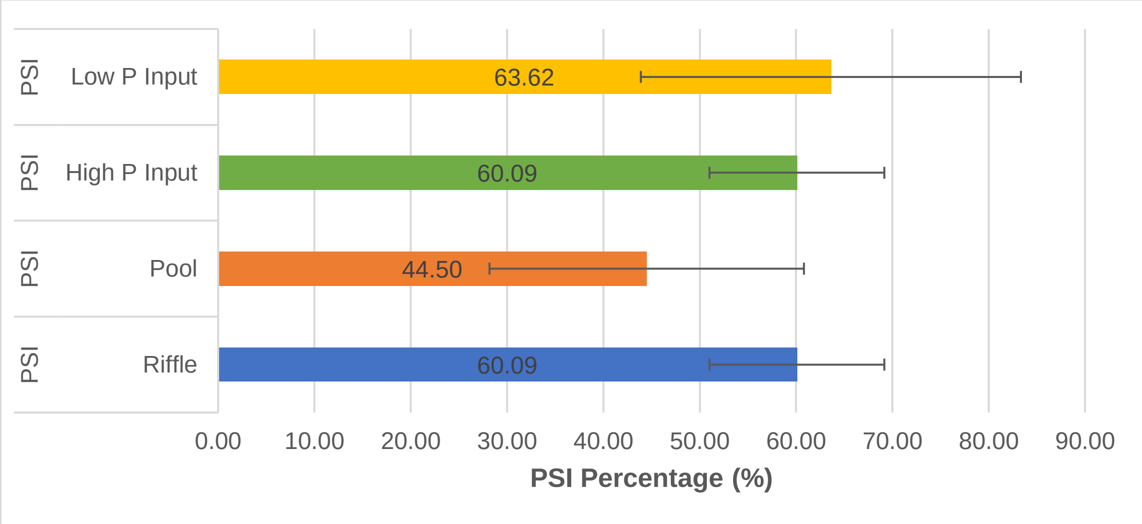


Figure 27: PSI percentage high phosphorus input riffle and pool sites, high and low phosphorus input sites, with standard deviations displayed by error bars. Riffle sites are represented by the blue bar, pool sites by the orange bar, high P sites represented by the green bar and low P input sites are represented by the yellow bar.

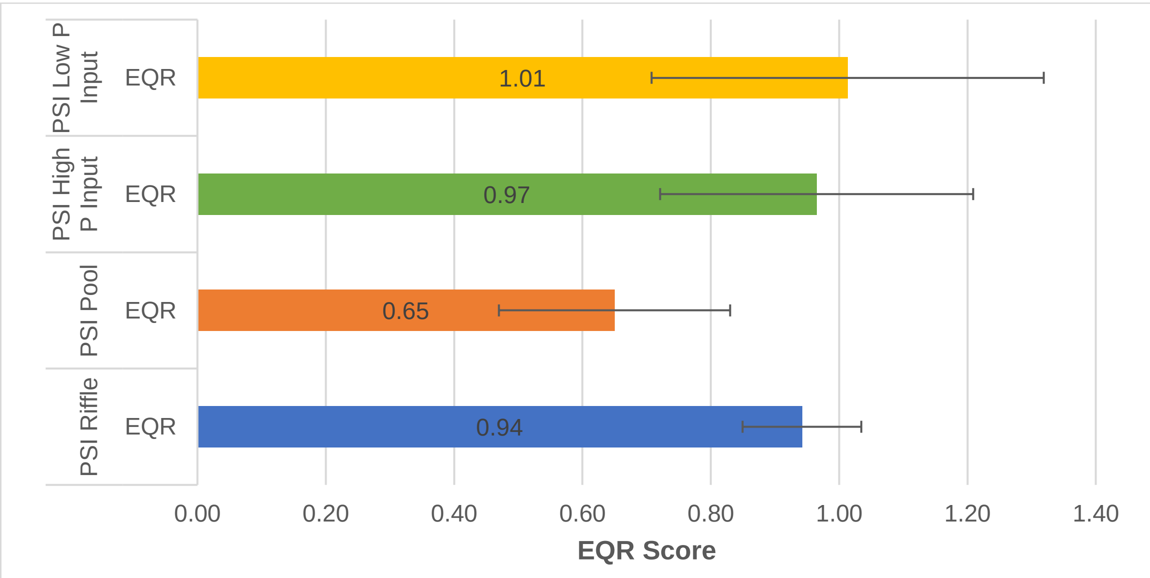


Figure 28: Ecological Quality Ratio PSI scores for high phosphorus input riffle and pool sites and high and low phosphorus input sites. Error bars show the standard deviation. Riffle sites are represented by the blue bar, pool sites by the orange bar, high P sites represented by the green bar and low P input sites are represented by the yellow bar.

3.7 Nutrient Analysis

Nutrient levels were compared between site groups, following appropriate transformation of values (log10) where required to satisfy the assumptions of analysis. Nitrate data could not be transformed to normality, so a Mann-Whitney U test was used instead. Results are shown in the Table below and accompanying plots.

3.7.1 High Phosphorus Inputs – Riffle Versus Pool

Nitrate

There was no significant difference between riffle and pool water samples (Paired T-test: T = 0.01, DF = 5, P-Value = 0.993) with pool sites having the higher mean value (0.6060) – there was almost no difference between the mean values of nitrate data between riffle and pool sites (0.001). Table 26 below shows details on standard deviation and means.

Nitrite

There was no significant difference between riffle and pool water samples (Paired T-test: T = 0.10, DF = 9, P-Value = 0.920) with riffle samples having the higher mean value (0.0267). Table 26 below shows details on standard deviation and means.

Ammonium

No significant difference was found between riffle and pool water samples (Paired T-test: T = 0.97, DF = 7, P-Value = 0.363) with riffle sites having more than double the greater mean than pool sites (1.0800). These results along with standard deviation and means are shown below in Table 26.

Phosphorus

Results of the paired T-test show no significant difference exists between riffle and pool sites (Paired T-test: T = 2.26, DF = 5, P-Value = 0.073) with pool sites having the greatest mean (1.3520), Table 26 below provides results from the T-test.

Table 26: Results of paired sample T-tests for chemical constituents present in water samples taken at high phosphorus inputs at riffle and pool sites along the river Almond.

Chemical Constituent	Site/ Flow Type	N	Mean	Standard Deviation	T-Value	DF	P-Value
Nitrate	Riffle	6	0.6050	0.2070	0.01	5	0.993
	Pool	6	0.6060	0.0530			
Nitrite	Riffle	6	0.0221	0.0221	0.10	9	0.920
	Pool	6	0.0189	0.0189			
Ammonium	Riffle	6	1.0800	1.2000	0.97	7	0.363
	Pool	6	0.5300	0.6460			
Phosphorus	Riffle	6	0.5030	0.8940	2.26	5	0.073
	Pool	6	1.3520	0.2170			

3.7.2 High Phosphorus versus Low Phosphorus

Nitrate

The results of the Mann-Whitney test reveal there is no significant difference between the median values of high phosphorus and low phosphorus input water samples (Man-Whitney U Test: $W = 52$, $n_1 = 6$, $n_2 = 6$, $P = 0.0543$) with high phosphorus input sites having a higher median value. High P input sites have a higher Nitrate level than low P input sites, with an almost significant difference occurring ($p = 0.0543$). This data is shown graphically in Figure 29 below.

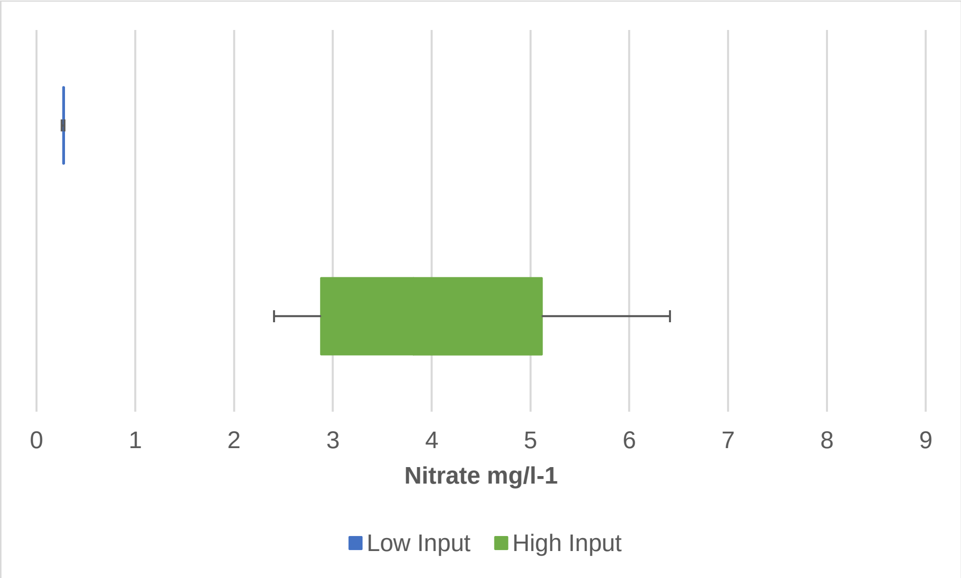


Figure 29: Boxplot showing the median value, 25th quartile and 75th quartile of log nitrate values from high phosphorus (Green) and low phosphorus (Blue) input sample sites from the main stem of the river Almond.

Table 27: Results of paired sample T-tests for chemical constituent's present in water samples taken at high and low phosphorus inputs at riffle sites along the river Almond.

Chemical Constituent	P Load	N	Mean	Standard Deviation	T-Value	DF	P-Value
Nitrite	High P Load	6	0.0267	0.0221	0.19	5	0.855
	Low P Load	6	0.0250	0.0018			
Ammonium	High P Load	6	1.0800	1.2000	0.73	5	0.500
	Low P Load	6	0.7160	0.1480			
Phosphorus	High P Load	6	0.5030	0.8940	0.38	8	0.717
	Low P Load	6	0.3420	0.5540			

Nitrite

There was no significant difference between high and low P input water samples (Paired T-test: T = 0.19, DF = 5, P-Value = 0.855) with high P input samples having the higher mean value (0.0267). Table 27 above provides details on standard deviation and means.

Ammonium

No significant difference was found between high and low P input water samples (Paired T-test: T = 0.73, DF = 5, P-Value = 0.500) with high P input sites having a greater mean than pool sites (1.0800). These results along with standard deviation and means are shown below in Table 27.

Phosphate

Results of the paired T-test show no significant difference exists between high and low P input sites with high phosphorus inputs (Paired T-test: T = 0.38, DF = 8, P-Value = 0.717) with high phosphorus input sites having the greatest mean (0.5030), Table 27 above provides results from the T-test.

4.0 Discussion

4.1 Catchment Coefficient Data – Phosphorus Loading

Initially, an assessment of the changes in phosphorus loading from the river Almond catchment revealed how the levels of phosphorus export had increased from the year 2000 to 2006 but decreased slightly between 2006 and 2012 (Figure 18). Most of the change is likely to be linked to an increase in non-irrigated arable land (Appendix 6) between 2000-2006 (Figure 3a&b). There was not a substantial reduction between 2006 and 2012 but the lack of further increase at least suggests that the situation with regard to land-use driven patterns of phosphorus loading is not getting any worse. A breakdown of land-use changes between from the year 2006 to 2012 can be found in Appendix 6.

The estimation of P loading from sub-catchments that the main stem of the river intersects (Figure 19) shows some pronounced variation. Of the sub-catchments present, the 'Union canal – Craigton to Murrayburn' has the largest phosphorus export to the river Almond and 'River Almond – Breichwater Confluence to Maitland Bridge' has the lowest phosphorus contribution. These data, however, are slightly misleading as the sub-catchments vary in total area, which contributes to the differences in total loading. The data presented in Table 13 under the column P Loading (Tonnes per ha) therefore provide a complementary representation of which sub catchment provides the greatest phosphorus load per unit area to the main stem of the river. The utility of different approaches depends upon the purpose to which the data are to be used, but in this case, the total loadings are more significant in terms of comparing diffuse inputs to different parts of the river.

Non-Irrigated arable land has the highest P loading coefficient (Smith et al. 2005) and often has the largest area within the sub-catchments. Eight sub-catchments in total have this as the largest land use area, whilst Broad-Leaved forest was the most common low input land use with five sub-catchments having this as the lowest contributing land use. Agricultural areas are often regarded as being one of the main contributors of diffuse N, P and fine sediment to a river system (Owen et al. 2012) and as a result when this land use covers a large proportion of a catchment, the estimated loadings are high. Forested buffer strips have been shown to be effective in reducing the levels of sediment that enter water bodies with the primary mechanism of binding and solidifying bank soils (Stutter et al. 2012). Nutrients, in particular phosphorus, have an affinity to the upper layers of loose, mobile top soil and the movement of this section of soil is a main transport mechanism, especially under high flow conditions, for nutrients to enter rivers and water bodies (Schulte et al. 2010). Sub-catchments with the highest proportion of forest related land use, where sediment movements may be reduced, are therefore correlated with a reduced total P load to the main stem of the river.

4.1.1 Riparian Corridor

Whilst examining nutrient loading over an entire sub-catchment helps to indicate broad patterns of variation, it is the land most closely associated with the river that is likely to influence inputs most strongly (Tufford et al. 1998). Unsurprisingly, as the size of the riparian corridor increases so does the loading to the river. However, it does not increase proportionately which suggests that variation in the scale over which such assessments occur can influence patterns observed.

The sections towards the source of the river were the lowest contributor to nutrient loading across the different riparian corridor distances assessed. This is consistent with variation in predicted nutrient loading contributions from sub-catchments. A variety of factors could be responsible for this change but this level of change likely to be primarily land use drive (Boyer 2002; Castillo 2010; Kemp & Dodds 2001). The sub-catchment and riparian corridors in the lower reaches of the river have a higher proportion of area consisting of high-loading land-uses, such as non-irrigated arable land. In comparison, upper sections of the catchment have a greater preponderance of low contributing land uses such as moors and heathland or peat bogs. Other studies of temperate streams have found similar patterns (Tufford et al. 1998), although the pattern of change from high to low sections of the river is variable across different biomes, depending on catchment topography and land-use (Castillo 2010).

4.2 Catchment Walkover Survey

The aim of the walkover was to assess the extent of diffuse and point source pollution sources along the main stem of the Almond, and allows a direct comparison to be drawn between the predicted loading from the riparian corridors and the observed inputs. The catchment walkover was also integral in identifying sampling areas for macroinvertebrate kick sampling as details on the number and frequency of not only diffuse sources but point sources were collected. In order to assess the potential impact of diffuse sources sites had to be selected that were not under the influence of point source pollution, which ultimately could distort the results found. Agricultural areas provide large and often significant levels of nutrient input into river systems (Posthumus & Morris 2010). It is known for waste water treatment plants (point sources) to provide pollution which dominates hydrological characteristics, overwhelm receiving waters and can regulate instream nutrient processing (Carey & Migliaccio 2009). As well as using data on the locations of point sources from the catchment walkover, a map of combined sewer overflow (CSO) points was used that was provided by SEPA to try and minimise the effect of these on the observed patterns. The Almond is however a river with a large number of diverse point sources, which are known to have a strong influence on water quality (D'Arcy & Frost 2001) so it is unlikely that the influence of these sources can be entirely removed from the assessment undertaken.

4.2.1 Catchment Walkover – Diffuse Inputs

Diffuse Points

Significant correlations existed between the number of diffuse points located and the total nutrient loadings per section at the 25 and 50m riparian corridor distance assessed. This indicates that as the number of diffuse points increases the level of phosphorus entering the river increases. These results found at the smaller riparian corridor level concur with findings from Dodds (2009), where stronger correlations between land use and phosphorus loadings occurred. Even though a lack of correlation at higher riparian corridor levels suggests no direct relationship exists between the two variables assessed, they represent different ways of assessing diffuse pollution inputs and both have their limitation, so together still provide complementary information for assessment.

Diffuse inputs recorded as discrete points only represent one pathway for diffuse pollution to enter river environments and other additional pathways are also likely to contribute to overall loading. Inputs such as erosion and poaching of banks, the extent of bank modification present and amount and composition of riparian and bankside vegetation all combine and interact so it is difficult to separate these individually to assess correlations (Dunn et al. 2012). Whilst these were also assessed, the different nature of the recording (points versus lengths or river bank or areas) make combining them together in a comparable fashion difficult. Additionally, the size and frequency of diffuse inputs can alter the nutrient loading to a river. During the survey work, diffuse inputs of different magnitudes were not distinguished, as these are often highly temporally variable (Panagopoulos et al. 2011) and thus not easy to categorise when survey work is undertaken over an extended period with different preceding weather conditions. Due to excessive riparian vegetation in some sections, not all diffuse points may have been recorded within a particular area. This could also reduce the probability of a correlation being found.

The variation within the catchment in terms of land use, highlights the need for a spatially distributed approach to sampling and modelling, hence making the implementation of sections more pertinent (Dunn et al. 2012). The application of this method maximises the chances of capturing patterns of variation by looking at a larger scale. Diffuse inputs are likely varying in total nutrient contribution; therefore, the number of diffuse points may not correlate to the level of nutrient loading. The basis of the approach used here assumes that each diffuse input identified contributes equally to the overall loading. In turn, if this was occurring then a significant correlation would be found between the P load and the number of diffuse points. In an ideal situation, an assessment quantifying the loading coming from each diffuse input would be undertaken in order to make the best assessment of diffuse inputs and the respective nutrient loading, this however, would not be feasible due to the continued monitoring required and the excessive number of diffuse points identified. The results from this study suggest the walkover assessment approach proves useful in identifying the number of diffuse inputs but does not provide an indication of diffuse pollution load.

Poaching

Much of poaching identified from the catchment walkover was caused by human influence, often found within areas popular with dog walkers. In the more remote reaches of the catchment livestock poaching is the main form. As outlined in the Controlled Activities and Regulations documentation (Scottish Environment Protection Agency 2014), these issues can be easily resolved with the addition of gates and fences to reduce the risk of sediment entering waterways in high-risk areas where high mobility top soils persist (Posthumus & Morris 2010). Poaching often removes the more mobile layers of soils and with that facilitates the movement of sediment bound nutrients into waterbodies (Søndergaard et al. 2003) indicating that the poaching mechanism can contribute large levels of diffuse pollution to water bodies. At the 25m, 50m, 100m, 200m and 500m riparian corridor levels no significant relationship was found indicating no relationship exists between poaching and total P load at these distances. However, as the presence or severity of poaching is dictated by the land use closest to the river, in areas where the dominant land use is within the urban category, for example discontinuous urban fabric, sediment bound P may not enter waterbodies as easily. In contrast, in a rural environment the likelihood of sediment entering via poaching increases. The P load delivered by poaching may also vary due to the characteristics of the sediment (Miller et al. 2009) and the amount of P stored within it and is known to be directly related to rainfall and flow – less contribution via poaching may persist in areas with less rainfall and lower flows as the conditions do not facilitate movement (Cuttle et al. 2007).

Erosion

Erosion was found along the majority of the main stem in varying severity and was particularly prevalent in more rural reaches of the river where limited urbanisation and small amounts of reinforcement is present. The main stem of the river Almond is heavily reinforced with 15.13 km of reinforcement measures present, mostly used to protect meanders, properties and populations closest to the river banks (Brainwood et al. 2006) from the erosion of the river under high flows. The results found no correlation between total P load and area of erosion at any riparian corridor distance. This is most likely primarily due to the complex interactions between all forms of diffuse pollution inputs as well as interactions with rainfall and the type of land use present. Overhanging vegetation present along most of the main stem, with 52.59 km in total across both banks. This has been documented to stabilise bank materials, including the stabilisation of transport sediments, with this colonisation eventually facilitating the construction of large land forms such as river banks, island and flood plains (Gurnell et al. 2012). Whilst reinforcement and modification reduces the movement and erosion of sediment it can often be detrimental to the natural processes within a river – sediment erosion and deposition are considered a natural phenomenon which has importance for channel processing and ecological functioning (Jones et al. 2012a). Humans have influenced the rate at which sediment enters rivers and are therefore speeding up this natural process which could have detrimental effects to salmonid spawning areas and the macroinvertebrate communities (Extence et al. 2011b).

4.3 River Invertebrate Classification Tool

The WFD adopted an approach of using biological communities, in conjunction with chemical and hydro-morphological measures to assess and measure quality. The implementation and the need for all member states to achieve the same status lead to widespread standardisation of not only methods but reference states which in turn has led to the development of intercalibrated metrics and indices to assess water quality (Acreman & Ferguson 2010) – prior to this no widely implemented methods or procedures existed (Hering et al. 2010). Macroinvertebrates are still used extensively within the WFD to monitor and asses the water quality and overall health of waterbodies due to their ability to show acute and chronic pressures as well as occupying an important position in food webs (Kartikasari et al. 2013).

4.4 Water Quality

BMWP and Associated Indices

Riffle Versus Pool

Riffle sites have an increased flow in comparison to pools and the movement of water is a defining characteristic of rivers and streams which influences aquatic organisms (Poff & Zimmerman 2010). The influences of flow affect organisms indirectly by altering the delivery rate of nutrients, altering water chemistry, substrate composition, organic particles and the presence of available habitat (James et al. 2007). As a result of these differences in flow type, scores for BMWP, ASPT and N-Taxa were higher at riffle than pool sites indicating more favourable conditions for a wider range of pollution sensitive macroinvertebrates to establish. The observed values for riffle sites were also closest to predictions and were in higher water quality categories. For EQR BMWP sites and ASPT scores, a significant difference existed, whereas for N-Taxa values, no significant difference was found between mean EQR values. Abundance weighted WHPT ASPT calculations changed the relationship to non-significant, the value however was very close to being significant (Table 24). WHPT N-taxa calculation revealed that no significant relationship between riffle and pool existed but a strengthening of the p-value did occur. ASPT is thought to be less specific than BMWP especially when investigating seasonal variation (Zamora-Muñoz et al. 1995) but is often calculated to fill the 'gap' or shortfalls not calculated from BMWP. N-Taxa is an indicator of diversity at each site – higher the N-Taxa score, the more taxa are present.

As the physicochemical characteristics entered into RICT predicted higher BMWP scores it can be assumed that other factors such as degraded water quality or an overall poor ecosystem health have caused a reduced BMWP score. This suggests that the high nutrient loading entering the river was having an effect on the macroinvertebrate communities at both riffle and pool sites. The link between nutrient inputs and photosynthetic biomass is well established (Lewis & McCutchan 2010) and relatively high levels of algal material and moss were evident at some sites, notably those at Seafield which correlates with the reduced BMWP score. Flow type is seen as a key determination for a number of abiotic factors: depth, sediment deposition and mobilisation rates, temperature, turbulence and water chemistry, which ultimately alter macroinvertebrate community composition, abundance and diversity (Graeber et al. 2013). Mayflies, stoneflies and caddisflies are regarded as sensitive taxa to stressors including flow regime, nutrient load and water quality, meaning they are used as good indicators of environmental quality (Ferreira et al. 2014). These groups, the EPT fraction of the community, were all found in greater numbers in samples from riffle sites than pool sites suggesting that, when habitat effects have been taken into account, overall environmental quality at these sites is greater. Interestingly based on the physicochemical characteristics entered into RICT, pool sites have higher predicted BMWP values. This indicates that pool sites are being degraded possibly by nutrient input, as the physical and chemical characteristics suggest that the pool sites should have a higher score. Factors specific to sites can influence a macroinvertebrate community whether that it is hydrological or the substrate characteristics present. It is often difficult to compare the communities present at each site due to the preferred method of sampling in each flow type – shallow riffles are generally preferred with an emphasis on kick sampling, whilst at deeper pooled areas sweep sampling if often employed (Brabec et al. 2004). Despite this, Brabec et al. (2004) found that despite riffles and pools having differing microhabitat characteristics, the presence of organic pollution actually reduces the difference in invertebrate fauna between the two differing flow types. This perhaps explains why no statistically significant difference was found between riffles and pools for ASPT and N-taxa EQR values as both riffle and pool sites are subjected to high P inputs.

High flows are likely to reduce the effects of excess nutrient exposure (Hutchins et al. 2010) owing to reduced retention time, and thus lower uptake by photosynthesisers. Conversely, the pool sites, with lower flows, have a longer potential retention time, allowing for greater local uptake or effects to be felt, potentially including enhanced levels of benthic algae or other vegetation growth. Similarly, pools will also potentially retain more particulate organic matter and preferences of particular taxa, such as caddisflies – e.g.

Urbanic et al. (2005) may reflect these varying conditions. It is possible that the reduced ASPT score at pool sites is an indication of habitat preference rather than nutrient pressures being magnified. Furthermore, riffles do not allow for the effects of sediments to act as quickly or as intensely due to faster flows occurring; this reduces the settlement of suspended sediments. EPT families are particularly sensitive to ecological pressures and all score highly in the BMWP score sheet and are usually found in more ‘pristine’ habitats (Kenney et al. 2009) with some families within the EPT groups preferring the high flows of riffles. Mayfly organisms, particularly Heptageniidae are recognised indicators of substrate changes, particularly those associated with lentic habits which are more likely to be characteristic of pool sites (Courtney & Clements 2002).

Macroinvertebrates are most directly affected by changes in flow types when it affects feeding methods, for example filter feeders require the flow of water to catch and provide organic material for consumption. Organisms such as Simuliidae utilise filter feeding (Rivers-Moore et al. 2006) whilst caddisflies prefer faster moving streams and rivers for respiratory reasons (Okano & Kikuchi 2012). As a result, Simuliidae and caddisfly organisms are found in greatest abundance at riffle sites, and seeing as caddisflies are a diverse and large group, this leads to a greater N-Taxa score. The high flow rates reduce the ability of nutrients and organic material to settle and persist within the sediment (Graeber et al. 2013), therefore reducing the impact posed by increased nutrients (Jones Jr 1997) - this factor possibly leads to a greater abundance of macroinvertebrates at riffle sites as found in. Low flows increase the likelihood of substrate smothering by sediment settling out of the flow and in turn provides less favourable conditions for sediment-sensitive fauna that respond unfavourably to the presence of sediment (Extence et al. 2011b), meaning organisms that prefer stone and gravel streambed composition will be impacted. Logan & Brooker. (1983) found that increased levels of particulate solids had a greater effect on pool sites than riffle sites, which in turn leads to reductions in the number of organisms, such as stoneflies present at pool or low flow sites, which contributes to the reduced N-Taxa score.

Macroinvertebrate drift occurs when taxa move from one area to another due to unfavourable conditions brought upon by the flow type within the current area. James et al. (2007) found that once flow conditions changed, drift occurred in favour of the movement from low flow areas to high flow areas. This could further explain why the N-Taxa score is higher at riffle sites than pool sites. With the N-Taxa values being higher at riffle sites this provides an indication that the habitat characteristics brought about by the riffle flow type are able to support the cohabitation of a wider range of macroinvertebrate taxa than the pool sites. With the presence of EPT groups also at riffle sites adding further diversity it may also lead to a greater number of organisms inhabiting riffle sites than pool sites. Furthermore, this highlights the potential for organisms inhabiting pool sites having traits that are specialised to low flows as a result.

High Phosphorus Versus Low Phosphorus Inputs

There was no significant difference found between high and low P input sites for any aforementioned metric but for EQR BMWP higher scores were found at high P input sites, for ASPT similar scores were found at low P input sites and for N-Taxa EQR scores were higher at high P input sites. Abundance-weighted WHPT calculations for ASPT and N-taxa revealed no significant relationship exists between high and low P sites but a strengthening of both p-values does occur.

A report published by Howieson (2011) highlighted the historical mining activity within the Almond catchment and the long lasting effects they have caused upon within the river Almond catchment. Appendix section 1 shows evidence of metal leaching close to some of the riffle and pool sites possibly leading to the reduced BMWP scores. The presence of mayflies, stoneflies and caddisflies at the high and low P inputs sites indicates that flow conditions are favourable for all organisms but mayflies, in particular Baetidae, are found in greater numbers in high P input sites whilst Perlodidae, and Limnephilidae were present in greater numbers at low P input sites. These groups are regarded as being key for nutrient cycling and processing of coarse organic matter (Ferreira *et al.* 2014) suggesting that their preferred habitats may be associated with areas with relatively high levels of detritus. High levels of nutrient input is found to increase detritus quality but not quantity (Small et al. 2016) and could contribute to the community structure present at both high and low P input sites and provide a reason as to why BMWP scores and N-taxa scores were higher at high P input sites.

Many of the low and high P input samples sites have overgrown banks potentially providing cover and habitat for invertebrate organisms such as caddisflies, and also the organic matter that these organisms are closely associated with (Colburn & Garretson Clapp 2006). Whilst both high P and low P input sites generally have a diverse and functioning invertebrate community, Small et al. (2016) found that higher diversity was found in more open, less covered streams than more closely-knit streams despite, like high and low P sites, having similar physicochemical properties. The low P input sites have more tree cover and thicker bankside vegetation than high P input sites which perhaps contributes to the increased BMWP and N-taxa scores. It is noted that caddisflies are often found within fringe vegetation and slow-moving waters, often pooled. Baetid mayflies are most commonly found in streams or rivers with gravel or stone based substrate (Buss & Salles 2007) indicating that the substrate present at the high P input sites is more favourable and contains more gravel and stone based substratum.

The EQR ASPT score achieved at the low P input sites suggest that these sites are under lower levels of nutrient input than high P input sites which are further highlighted by high numbers of leeches, in particular, Erpodellidae. This conclusion can be drawn as raw ASPT scores indicate a combination of nutrient and habitat impact whereas EQR ASPT indicates the effect of nutrient impact only. Leeches in general are regarded as being abundant under areas with low dissolved oxygen conditions (DO), often below 5ppm (Kaller & Kelso 2007), with excess vegetation growth, caused by increased nutrient levels, the primary cause. Eutrophication is primarily linked to organic pollution and nutrient input. An increase in P and N inputs with a combination of sunlight leads to increased or enhanced plant growth – during these conditions the primary change to invertebrate fauna is a shift in feeding strategy to herbivory (Brabec et al. 2004). Coleoptera are primarily found in areas with high P input suggesting an affinity to more enriched areas. Coleoptera and Hemiptera respond favourably to high input P conditions, an increase in abundance is observed (King et al. 2004), and so do grazing macroinvertebrates such as some mayfly organisms and gastropods.

Interestingly, as illustrated in Wagenhoff et al. (2012) macroinvertebrate diversity and richness are generally negatively affected by the presence of increased nutrients but high P input sites scored the same as low P input sites. Melo & Froehlich (2001) found that macroinvertebrate diversity is positively related to stream size. With regards to the river Almond, the size of the river fluctuates and generally increases the further downstream from the source the sample sites are located. Four of the six sites sampled under low P input conditions are located upstream of the first high P input sample site and as stated. Confounding variation in effects of stream size and the location of the high P sites sampled could in turn explain why a similar N-Taxa value is found at low P input sites - with smaller stream size, reduced species diversity occurs leading to a lower N-Taxa score. The majority of high P sites sampled are of a larger stream size than low P sites, thus a higher number of taxa could be found at high P sites whilst low P sites' N-Taxa score is reduced by reduced stream size.

It is estimated that in the United Kingdom that around 700km of streams and rivers are affected by the legacy left by the mining industry, with drainage discharge and iron precipitation causing alterations in the chemical and ecological status of waterways (Bradley 2010). Figures 14a and 14b show low P input sample sites with evidence of heavy metal inputs indicated by the orange turbidity of the water and the orange presence on the substrate. These inputs are known to reduce diversity (Hogsden & Harding 2012) as the less tolerant taxa either die or colonise different areas of the river, whilst the more tolerant taxa survive and become dominant due to reduced pressure caused by predation and competition (Bradley 2010). The indices could therefore perhaps be skewed by these influences and indeed high P inputs or other physicochemical factors may act in a positive fashion under these circumstances allowing for more diversity to persist in high P input sites. This brings into question the appropriateness of site selection criteria for the sites in question and whether this influences the ability to draw valid conclusions when comparing with sites where this form of pressure is not present. It is worth considering whether this is an anomalous result and if the outcome of the current sites sampled were monitored over a longer period of time if the results would still indicate the high P sites having higher diversity. This study of course only provides a small snapshot into the macroinvertebrate communities at the time of sampling so monitoring on a monthly basis would capture trends and allow anomalous data to be excluded and highlighted more prominently. Cardinale (2011) studied the use of algal biofilms and suggested that the more diversity exists in an area the better the ecosystem is at removing nutrients which could explain why a similar number of taxa persist at high P input sites and low P input sites. As noted in section 2.3.1, algae were abundant at the high P input sites, especially at Seafield sample sites. This in turn could account for the high numbers of taxa at high P input sites as a larger level of diversity persists. The CORINE land use and phosphorus coefficient data indicate where high levels of P nutrient input could be entering the river but due to the nutrient cycling occurring by the large diversity present the degrading effects may not be evident.

4.5 Lotic-invertebrate Index for Flow Evaluation

The LIFE metric (Extence et al. 1999) uses the current velocities preferred by macroinvertebrates to indicate the flow rate within rivers and streams (Bunn & Arthington 2002) thus providing a metric that is suitable to assess changes in aquatic fauna due to flow regime variations. The LIFE metric may not be able to distinguish fully the effects of flow variation and the effect the differences between riffles and pools in terms of nutrients, but was calculated in order to assess the manner in which differing flow regimes ‘handle’ the diffuse inputs and partly to fulfil objective 3: Quantify the impact of diffuse pollution inputs and associated impacts on environmental quality of the river.

High Phosphorus Inputs – Riffle Versus Pool

The results presented in section 3.5 indicate that high P input riffle sites have a higher LIFE score than pool sites with similar inputs. However, high P input pool sites have a higher EQR LIFE score than riffle sites which would indicate that the communities in riffle sites are deviating more strongly from those expected based on the prevailing flow conditions than the pool communities. As the equation used to calculate LIFE adjusts scoring depending on abundance (Dunbar et al. 2010), the variety of organisms' sensitive to flow as well as the abundance dictates the score achieved at the input sites. Pool sites have fewer organisms overall than riffle sites and have a lower average N-Taxa score (Figure 23) but despite this have a higher LIFE score. When conditions that taxa prefer exist, such as low flows for example, the number of organisms tailored to those conditions will increase whilst the organisms that are not adapted will decrease, and vice versa (Extence et al. 1999). This could explain the difference in N-Taxa values as a wider range of taxa prefer the conditions at the riffle site than the pool sites, but not only flow conditions dictate this a wider range of abiotic factors also determines the conditions present.

Assessing the physical and chemical data collected from sample sites reveals that on average pool sites had higher recorded velocity than riffle sites. This could be due to the sampling of riffle and pool sites taking place in different sample sessions meaning flow conditions, despite best efforts to carry out sampling under the same conditions, may have been different. The movement of macroinvertebrates in response to flow, otherwise known as drift, can occur after low flow events (Dewson et al. 2010) or high flow events (Lancaster & Hildrew 1993) and can happen actively or passively which aids in the removal from unfavourable conditions. Active drift can occur in response to conditions presented due to low flow which in turn reduces the number of refuge areas in the area for not only avoidance of unfavourable conditions e.g lack of food, reduced oxygen levels but also the avoidance of an increased concentration of predators (Dewson et al. 2010). Fish species, in particular Atlantic salmon (*Salmo salar*) predate on macroinvertebrates and inhabit areas with high flows – these are often relied upon as an indicator of when to start migratory periods from sea to freshwater environments to begin spawning (Bardonnet & Bagliniere 2000). Brown trout (*Salmo trutta*) most commonly exploit pools and slower moving reaches during the night whilst the most primary behaviour during the day is feeding (Klemetsen et al. 2003). Brown trout are more commonly found on the river Almond than Atlantic salmon and in riffle sites during summer months makes up the bulk of the predation on macroinvertebrate organisms (Klemetsen et al. 2003).

Naturally occurring material is distributed all along the river in form of large woody debris and debris dams (a combined total of 226 recorded) also altering flow patterns. Weirs are often the primary focus of connectivity papers as their presence creates two very contrasting flow types and therefore habitat types. Hydrological changes brought about by weirs can lead to changes in hydrological characteristics which therefore alters macroinvertebrate community diversity and abundance (Mueller et al. 2011). Naturally occurring vegetation and wooded debris form habitat for macroinvertebrates whilst also altering flow type (Miller et al. 2010) and can be found at, either above or below, riffle and pool sample sites. These changes could facilitate the differences in macroinvertebrate assemblages found at riffle and pool sites whilst also providing refuge areas during high flow periods. Research conducted by Dunbar et al. (2010) found a correlation between LIFE scores and the presence of modification in rivers, specifically reinforcement/ re-sectioning, which lead to degraded LIFE scores. This is perhaps applicable in urbanised areas where large-scale infrastructure is present to protect buildings and developments from flooding, but also within the rural environment where farmers protect lands and livestock from potential flooding. At the scale of the main stem of the river, a significant relationship between reinforcement/ modification and total P input was not established but perhaps at the catchment scale, where the modifications of tributaries and all sub-catchments are mapped a significant relationship may exist. Certainly, areas with modification would facilitate the movement of rainwater or run off from the land into the river as more potentially impermeable surfaces cater for this.

4.6 Percentage of Sediment-sensitive Invertebrates

High Phosphorus Input – Riffle Versus Pool

The results presented in Figure 28 show that on average riffle sites have a higher PSI score than pool sites meaning that organisms at riffle sites are less impacted by sediment. Riffle sites are in the 'moderately' sedimented category as well as pool sites but the difference between the two PSI scores is 15.6, making the PSI values obtained very different. A significant difference was observed between riffle and pool sites with riffle sites being significantly closer to the predicted scores, although neither riffle or pool sites achieve the predicted PSI scores calculated by RICT. Sediment can impact upon macroinvertebrate organisms in various ways. The main physical impacts that can be caused by fine sediment include: abrasion, clogging, burial and can alter substrate composition (Jones et al. 2012). Abrasion of body parts, particularly in filter feeding organisms such as Hydropsychidae, can cause damage to feeding appendages therefore reducing the likelihood of filter feeders and organisms susceptible to damage inhabiting areas with high fine sediment input. Clogging of feeding and respiratory apparatus can cause difficulties in these biological processes as ingestion of particles can not only cause damage to internal organs but further wastes time and energy in the expulsion of unwanted particles. Simuliidae (black flies) are particularly susceptible to clogging as they are unselective of food meaning energy is spent removing the unwanted material leading to a decreased feeding rate (Jones et al. 2012). Caddis flies and Simuliidae species are often not found within areas of high sediment inputs - with both groups predominantly being more common at riffle sites this suggests pool sites have larger levels of fine sediment input (Murphy *et al.* 2015). As riffle sites have a higher PSI score it could be that pool sites are high in sediment-bound nutrients (Murphy et al. 2015) as demonstrated in Table 26 where phosphate showed a trend towards higher values for pool sites, even if the difference was not significant. Caddisflies as a group are generally high scoring on the BMWP index meaning the reduced presence at pool sites could be a factor in the reduced ASPT scores when compared to riffle sites. Burial of organisms through the accumulation of sediments more common within sessile invertebrates such as

Unionidae. Baetidae were found in riffle sites with faster moving water, which indicates they prefer riffle sites with low sediment inputs. Research by Wood et al. (2005) suggested that Baetidae prefer less sedimented areas as they are highly sensitive to burial from sediment, which is consistent with the abundance data collected in this study.

Murphy et al. (2015) categorised sediment impacts into two main components: organisms are sensitive to the quantity of fine sediment and the quality of organic fine sediment. Using this, Murphy found that organisms such as Ephemerellidae and Nemouridae are sensitive to organically enriched sediment and Dytiscidae beetles are sensitive to the overall sediment totals and are tolerant of organically enriched sediments. Ephemerellidae and Nemouridae were found in greater numbers at riffle sites than pool sites and Dytiscidae are present at pool sites in greater numbers indicating that perhaps pool sites have sediment inputs that are organically enriched whilst riffle sites have reduced levels. Limnephilidae were also classified by Murphy et al. (2015) as sensitive to the total sediment input and tolerant of organically enriched sediment but were found in riffle sites in this study, which conflicts with the idea that riffles contain less of this material. The presence of Limnephilidae could be due to the preference for flow type outweighing the drawbacks of sediment presence – the requirement for oxygenated water possibly being a contributing factor.

High Input Versus Low Input

The data suggested that high P input sites were 'moderately' sedimented and low P input sites were 'slightly' sedimented as determined by EQR calculations, but no significant difference was found between high and low P input EQR scores. However, a small difference in score exists between the two sites, 3.53 in the favour of low P input sites (Figure 30) whilst EQR values indicate that PSI EQR values are close to predictions, with low P input sites surpassing expected scores (Figure 28). As the flow regime is the same at both high and low P input sites, by inference, it is potentially sediment presence associated with diffuse inputs of P that causes the disparity between the PSI results obtained (Extence et al. 2011b). Nutrient and sediment inputs ultimately become distributed throughout a river system as flow facilitates this movement. However, under low flow conditions such as those resulting from impoundment from weirs or large pooled areas, the flow is reduced leading to a change in distribution (Vörösmarty et al. 2003). This can have negative impacts upon invertebrate taxa and the general habitat within the area as demonstrated by the reduced PSI score at high P input sites. The Almond is impacted by a large number of weirs (34 based on a survey conducted in 2010 (Howieson 2011), locations marked on the digitised walkover catchment data), indicating that there is potential for sediment to be retained within the impoundment zone. Furthermore, large pools are located to the west of Livingston, as shown in the digitised walkover, that could contribute further to the retention of fine sediment which further reduces impacts upon biota. Upstream reaches of the high P input sites have high levels of natural weirs, large woody debris and artificial weirs that create areas of slow moving or impounded water which facilitates the deposition of sediment. In comparison, low P input sites are spread further across the main stem of the river, not in as close a proximity which could increase the potential impacts caused by sediment inputs. The levels of erosion and poaching are higher around sites with low P inputs; such inputs have been found previously to be linked to variation in LIFE scores though the flow conditions present (Dunbar et al. 2010).

In essence, high P input sites may be more influenced by any sediment retention effects of the weirs present than low P sites, which were mostly located upstream as shown in the digitised walkover surveys. Riverine environments are complex systems hydromorphologically and such influences can confound the ability to establish patterns in relation to diffuse pollution. The site selection process was established to attempt to reduce the impact of these effects, but in heavily impacted rivers, such as the Almond, background ‘noise’ from other factors such as different pollutants and hydromorphological modification appears to obscure any distinct signal caused by the diffuse pollution.

4.7 Nutrient Analysis

High Phosphorus Input- Riffle versus Pool

There was no significant difference between riffle and pools for all assessed nutrients (section 3.7.1). This is not altogether surprising as the pools sampled were located directly below and within close proximity to the riffles sampled. The results indicate that both riffle and pools have the same or similar levels of dissolved nutrients but it should be noted that the levels of these nutrients not only vary with the type of land use around them but also the levels of rainfall. It is generally accepted that nitrogenous inputs enter river environments during winter months under high flow conditions (Goodale et al. 2009) meaning that at the time of sampling, summer period, these would be at the lowest due to a high demand from aquatic plants. The balance of P inputs from diffuse and point sources is variable and is evidently in the case of the river Almond, the effects caused by inputs from CSO's and waste-water treatment works may have masked subtle fluctuations and effects from diffuse points. As a result, P levels should be highest during low flow summer periods intimating that significant relationships should exist between the flow types if this is abiotic factor interacts with P differently. As water sampling only occurred once it is difficult to assess whether an increased nutrient presence occurs at either riffle or pool sites. Long term monitoring data of nutrient concentrations at each site and also determining levels of bound, sediment-associated forms, would be beneficial in assessing whether a relationship exists between nutrient levels and the flow type experience at each site but this can be logistically impractical and costly.

High Phosphorus Inputs versus Low Phosphorus Inputs

Similar to the above results, there were no significant differences in the nutrient levels recorded at high and low P input sites. This could mean that using the method of sample site identification detailed in section 2.1 is inaccurate at predicting nutrient loadings. However, there are strong limitations on the conclusions that can be drawn from a single water sample, and in order to get a more robust picture of any differences, longer-term regular monitoring would be required. This would help to establish whether the land use nutrient loading coefficient method of Smith et al. (2005) is applicable for estimating loadings of nutrients in general. There may also be unaccounted for differences between Ireland and Scotland (Bowes et al. 2008). As detailed in Bowes et al. (2008) quite specific information on the land use, number of livestock and locations and numbers of all point source and non-point source areas need to be accounted for to enable catchment managers to quantify and mitigate P loads to the catchment. This to a certain extent was incorporated into the site selection by including data from the walkover surveys but this again does not guarantee that all point sources were noted as some could be masked by vegetation or were below the water level on the day of the walkover. Taking this into consideration, it is therefore not a surprise that there were no strong differences in nutrient levels.

5.0 Conclusion

The project set out to provide an assessment of diffuse pollution inputs and the associated impacts on invertebrate communities by attempting to assess and quantify the nutrient loading and diffuse inputs at a catchment and local scale. A land-use based approach allowed for total P and N loadings to be calculated at a catchment, sub-catchment and different riparian corridor levels and was combined with a full catchment walkover of the main stem of the river. Initially P and N loadings were calculated using areas of different land uses and coefficients derived by Smith et al. (2005) and Foy and Girvan (2004), but due to the high category level and reduced specificity of N coefficients and the relative specificity of P coefficients, calculated P loadings were used to inform sampling areas. The walkover assessment of the main stem highlighted key habitat features including in channel features, channel modification, flow type and pollution inputs which facilitated, in conjunction with riparian corridor nutrient loadings, the selection of macroinvertebrate sampling areas. However, whilst a full main stem walkover was completed, some sections of the river were inaccessible meaning some features such as diffuse points may not have all been recorded, leading to an under-representation to some extent. To determine how land use and diffuse pollution interact, targeted macroinvertebrate sampling occurred at riffle and pool sites under high P input pressure, as determined by land use, coefficient data and walkover survey, as well as riffle sites under low P input conditions. From this macroinvertebrate sampling took place at 18 sites in total, to quantify the effect of diffuse pollution at areas of differing flow type and differing nutrient input level.

Water quality was assessed at each sample site using metrics such as PSI, LIFE, BMWP, ASPT and N-Taxa WHPT ASPT and WHPT N-Taxa. Statistical analysis showed no significant differences existed for any metrics calculated between riffles under high and low P input suggesting that the level of nutrients a community is exposed to has no significant effect on community composition or that site selection did not provide sufficient contrast to allow accommodate detection of any effects. Using abundance weighted WHPT metrics did show that the p-value moved closer to significance. It is not surprising no significant difference exists for LIFE indices as the flow type at each the sites are the same. The survey and site selection primarily considered features and conditions at the riparian corridor level; given the complexity of the river system, with numerous tributaries that are also contributing diffuse inputs the ability to detect a clear signal between areas with predicted high and low diffuse inputs may be compromised. This is an important consideration for the utility of the different approaches (catchment-scale land-use modelling or walkover surveys) in assessing areas likely to show impacts of diffuse inputs in systems where a range of other interacting factors may also be operating. The sample frequency could also have an effect as the results provided an insight into macroinvertebrate abundances at one-time point and occurs in only one season.

The nutrient data collected at the time of sampling indicated that no significant relationship existed between any dissolved levels for any of the comparisons undertaken. This is not particularly surprising as nutrient data was only collected from a water sample at one time-point and this is perhaps unrepresentative and unreliable due to the rapidly changing levels that can occur within water bodies. This time could have been used in more targeted macroinvertebrate sampling due to the unrepresentative nature of the samples. Increasing the number of macroinvertebrate sampling sites would increase the precision of the mean values therefore providing more accurate comparisons between mean values of the indices used. Increasing the sample size used would in effect reduce the margin of error and make the results more accurate. In order to gain an accurate representation of the nutrient levels persisting within the river, multiple samples over long time periods would be beneficial in gaining an insight into nitrogen and phosphorous levels due to different release patterns and differing rates in biogeochemical cycling. Again, seasonality and the frequency of sampling could be causing the non-significant results found from assessing the water nutrient data and a periodical sampling and analysis of water chemistry at all sites would provide data on any trends and potential significant differences.

However significant differences did exist between high P input riffle and pool sites for all indices calculated, except N-Taxa, inferring that flow type has a greater effect on macroinvertebrate community structure than levels of nutrient input. It is documented within the literature that macroinvertebrates inhabit areas based upon a number of factors but that flow type is the determining factor for colonisation as it has the ability to alter abiotic factors within the river (Dunbar *et al.* 2010a). The slower moving water found at pool sites may allow for higher levels of settlement of fine sediment and chemical transformation and retention of nutrients within the habitat and thus having more pronounced effects on the associated organisms, causing the significant differences found between metrics and community composition. The significant differences found could be most strongly driven by the difference in flow type and the level of nutrient input is less relevant. In order to test this further targeted sampling of riffles and pools of low P input areas could be undertaken and if no significant relationship exists then this could mean that it is the cumulative effects of both flow type and nutrient loading causing the difference in community. A more prolonged sampling period encompassing a range of seasons and conditions would help to establish the validity and generality of this result.

In essence, the methodology of combining land use data, nutrient loading coefficients and walkover surveys was developed to assess the extent to which these provide complimentary or contrasting information and was also used to help inform decisions with regards to where sampling should take place as well as attempting to fulfil the first and second study aims. To gain a true insight into how diffuse pollution affects macroinvertebrate communities periodic sampling incorporating all seasons would provide the best overall picture of community response to pollution. This is in most cases not possible due to high flows caused by weather and the logistics required for extended sampling efforts. A large-scale walkover including all tributaries of the Almond would allow for quantification into the amount of diffuse points contributing to diffuse pollution as well as including a greater level of specificity in N coefficient data, potentially aiding in the site selection process. Sampling with newly identified sites, whether on the main stem or tributaries, would provide a more conclusive and inclusive view on the interaction between diffuse pollution, land use and the effects on the macroinvertebrate community.

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Appendices

Appendix 1

The figures above indicate the presence of mine leaching points in areas along the main stem of the river. These inputs can severely impair the habitat quality for benthic organisms and provides unfavourable conditions such as changes in pH, substrate smothering whilst reducing the aesthetics of the area.

Visible discoloration of water entering the main stem of the river Almond from Gogar burn near Edinburgh Aiport. This is an inidication of some form of enrichment or pollution causing the discolouration of the water body.

This set of figures provide evidence of possible diffuse pollution directly entering the river with the tarmaced footpath excentuating and facilitating the movement of diffuse pollution into the river. The figure on the right shows the movement of excess water from grassland into the main stem.

Visible severe bank side erosion caused by high flows which provides sediment input into the river.

Evidence of bank side poaching caused by cattle providing increased sedment input into the river.

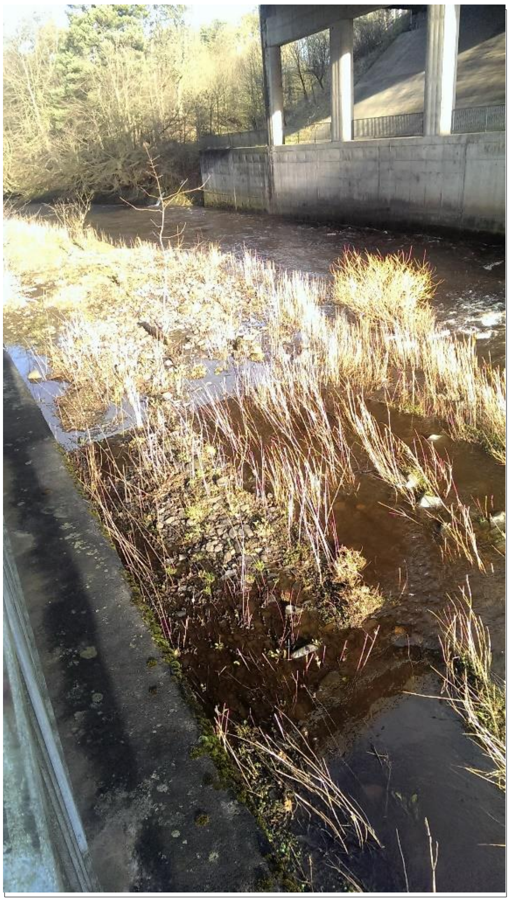
Further evidence of poaching this time most likely caused by human traffic predomonantly walkers as it is located on a popular walking section of the river near Cramond.

Bank side modification in the form of boulder reinforcement is present to protect bank side from erosion. As part of the undergoing construction work during the walkover in February the footpath was upgraded causing the removal of the modification leading to potential sediment inputs. Also present is the run flow type.

An example of broken water and white water flow type located below a major weir near Cramond along with a fish pass. The figure on the right shows boulder and cobble bank side reinforcement.

Pool flow type with railway sleepers as bank reinforcement



Bank protection and resectioning along the main stem of the river with hard engineering work present in both figures to protect a railway bridge and the M9 motorway bridge.

Construction work taking place on the banks of the river Almond in Livingston. Major construction took place including pathway construction, bank side redevelopment and implementation of a children's play park. This may have allowed for the uncontrolled release of sediment and diffuse pollution to occur as bare soil was in close proximity to the pathway as well as new artificial surfaces facilitating the movement of diffuse inputs.

Appendix 2

Nutrient export coefficients for Nitrate, Ammonium and Dissolved reactive phosphorus with corresponding land use type.

		Land use type				
		Urban land	Rough grazing	Forest	Other land	Agricultural land
Nutrient	Units	Nutrient export coefficient				
Nitrate	Tonnes N km-2 yr-1	0.20	0.20	0.20	0.20	0.05
Ammonium	Tonnes N km-2 yr-2	0.03	0.03	0.03	0.03	
DRP	Tonnes P km-2 yr-3	0.03	0.01	0.02	0.03	

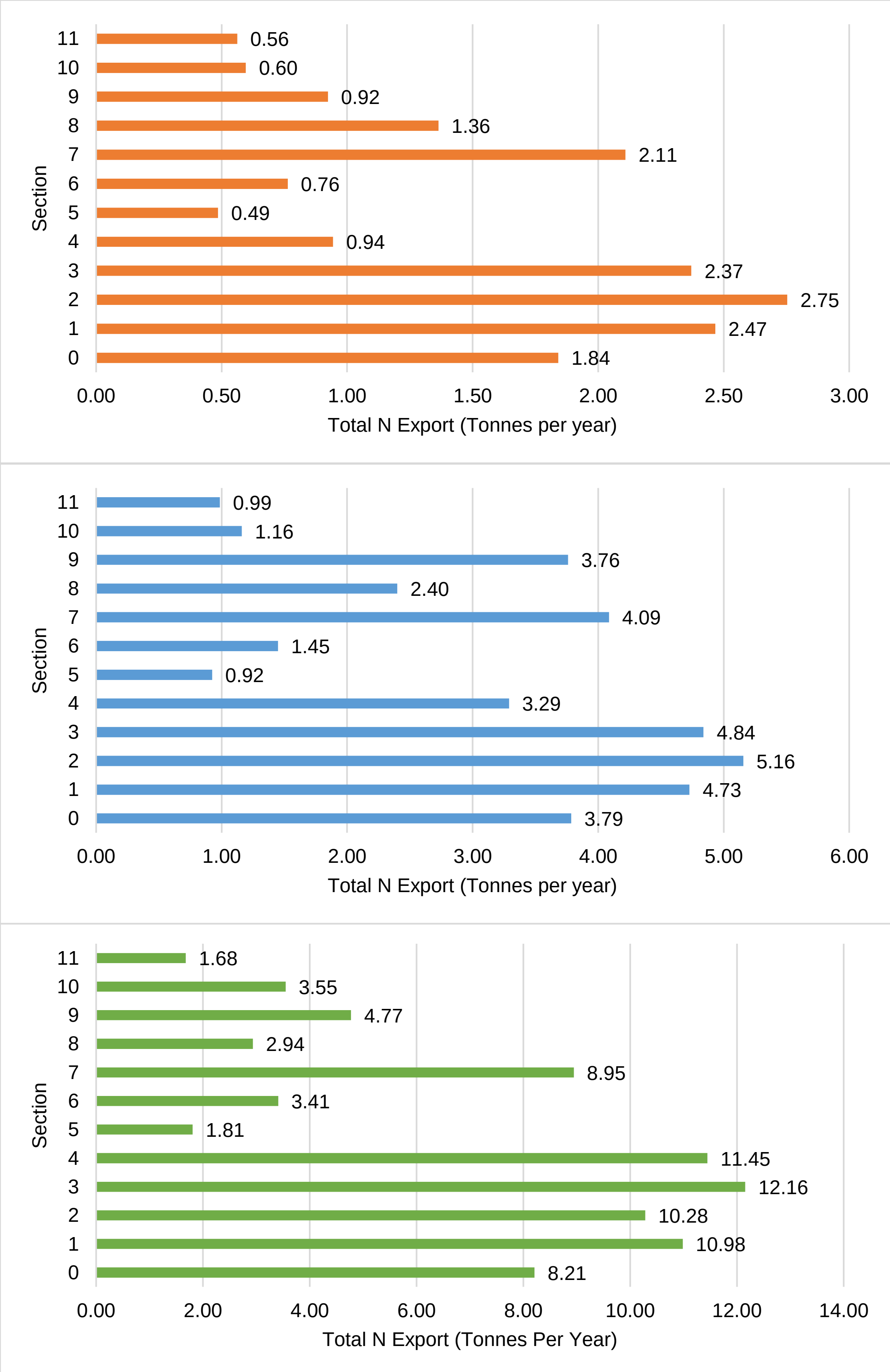
Nitrate with dissolved reactive phosphorus coefficients for agricultural land use types.

Corine Level	Corine land use type	Export coefficient	
		DRP	Nitrate - N
		Tonnes km^{-2} yr^{-1}	
2.1.1.	Non irrigated arable land	0.229	4.367
2.4.1.	Annual crops associated with permanaent crops	0.030	2.036
2.3.1.1.	Good pasture	0.030	2.036
2.3.1.2.	Poor pasture	0.036	0.747
2.3.1.3.	Mixed pasture	0.033	1.511
2.4.2.	Complex ultivation patterns	0.104	2.759
2.4.3.	Land principally occupied by agriculture	0.025	0.829

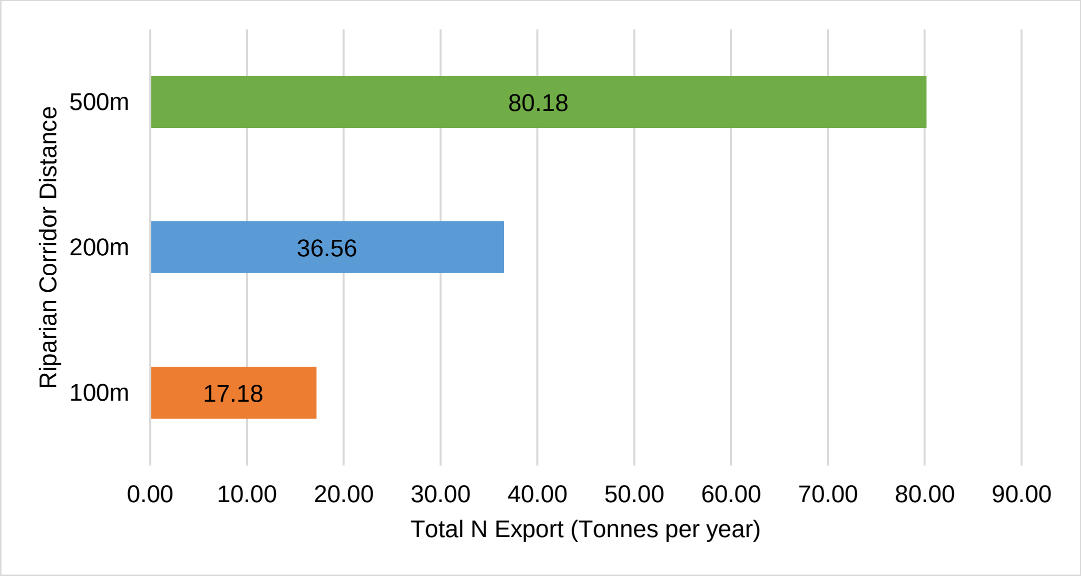
Nitrate with dissolved reactive phosphorus coefficients for forestry land use types.

Corine Level	Corine land use type	Export coefficient	
		DRP	Nitrate - N
		Tonnes km^{-2} yr^{-1}	
3.1.1.	Broad-leaved forest	0.013	0.000
3.1.2.	Coniferous forest	0.005	0.086
3.1.3.	Mixed forest	0.013	0.000
3.2.4.	Transitional woodland-scrub	0.013	

Appendix 3



Section breakdown data for 100m (Orange bars), 200m (Blue Bars) and 500m (Green bars) riparian corridor distance for total N export (Tonnes per year).



Total N export in tonnes per year at different riparian corridor levels.

Appendix 4

BMWP Score sheet for benthic samples

Score	Taxon group	Family
10	Stoneflies	Capniidae, Chloroperlidae, Leuctridae, Perlidae, Perlodidae, Taeniopterygidae
	Mayflies	Ephemerellidae, Ephemeridae, Heptageniidae, Leptophlebiidae, Potamanthidae, Siphlonuridae
	Caddisflies	Beraeidae, Brachycentridae, Goeridae, Lepidostomatidae, Leptoceridae, Molannidae, Odontoceridae, Phryganeidae, Sericostomatidae
	True bugs	Aphelocheiridae
8	Caddisflies	Philopotamidae, Psychomyiidae
	Dragonflies	Aeshnidae, Cordulegasteridae, Corduliidae, Gomphidae, Libellulidae
	Damselflies	Calopterygidae, Lestidae
	Crustaceans	Astacidae
7	Stoneflies	Nemouridae
	Mayflies	Caenidae
	Caddisflies	Limnephilidae, Polycentropodidae, Rhyacophilidae (includes Glossosomatidae)
6	Caddisflies	Hydroptilidae
	Damselflies	Coenagriidae, Platycnemidae
	Crustaceans	Corophiidae, Gammaridae
	Snails	Ancylidae, Neritidae, Viviparidae
	Mussels	Unionidae
5	Caddisflies	Hydropsychidae
	Beetles	Chrysomelidae, Clambidae, Curculionidae, Dryopidae, Dytiscidae, Elmidae (Elminthidae), Gyrinidae, Haliplidae, Hydrophilidae, Hygrobiidae, Scirtidae
	True-bugs	Corixidae, Gerridae, Hydrometridae, Mesoveliidae, Naucoridae, Nepidae, Notonectidae, Pleidae
	Flatworms	Dendrocoelidae, Planariidae
	True-flies	Simuliidae, Tipulidae
4	Mayflies	Baetidae
	Alderflies	Sialidae
	Leeches	Piscicolidae
3	Crustaceans	Asellidae
	Snails	Hydrobiidae, Lymnaeidae, Physidae, Planorbidae, Valvatidae
	Mussels	Sphaeriidae
	Leeches	Erpobdellidae, Glossiphoniidae, Hirudididae
2	True-flies	Chironomidae
1	Worms	Oligochaetae

Calculation of scores

BWMP score = sum of BMWP values for each family recorded (don't multiply by abundance, whether there is one individual or 1000, the score is the same)

N-taxa = number of taxa recorded that have a BMWP score (i.e. do not count those which do not have a score)

ASPT (average score per taxon) = BMWP/N-taxa

Appendix 5

WHPT abundance category groups and scores.

	AB1	AB2	AB3	AB4
TRICLADA (Flatworms)				
Dendrocoelidae	**3.0**	**2.6**	**2.6**	**2.6**
Dugesiidae	**2.8**	**3.1**	**3.1**	**3.1**
Planariidae	**4.7**	**5.4**	**5.4**	**5.4**
MOLLUSCA (Snails, Limpets and Mussels)				
Neritidae	**6.4**	**6.5**	**6.9**	**6.9**
Viviparidae	**5.2**	**6.7**	**6.7**	**6.7**
Unionidae	**5.2**	**6.8**	**6.8**	**6.8**
Sphaeriidae (Pea mussels)	**4.4**	**3.5**	**3.4**	**2.3**
Lymnaeidae	**3.6**	**2.5**	**1.2**	**1.2**
Planorbidae (excl. *Ancylus* group)	**3.2**	**3.0**	**2.4**	**2.4**
Valvatidae	**3.3**	**3.1**	**2.7**	**2.7**
Physidae	**2.7**	**2.0**	**0.4**	**0.4**
Acroloxidae	**3.6**	**3.8**	**3.8**	**3.8**
Ancylus group (= Ancylidae)	**5.8**	**5.5**	**5.5**	**5.5**
Bithyniidae	**3.6**	**3.8**	**3.3**	**3.3**
Dreissenidae	**3.7**	**3.7**	**3.7**	**3.7**
Hydrobiidae	**4.1**	**4.2**	**4.6**	**3.7**
OLIGOCHAETA (worms)				
Oligochaeta	**3.6**	**2.3**	**1.4**	**-0.6**
HIRUDINIA (Leeches)				
Piscicolidae	**5.2**	**4.9**	**4.9**	**4.9**
Glossiphoniidae	**3.4**	**2.5**	**0.8**	**0.8**
Erpobdellidae	**3.6**	**2.0**	**-0.8**	**-0.8**

	AB1	AB2	AB3	AB4
Hirudinidae	**-0.8**	**-0.8**	**-0.8**	**-0.8**
CRUSTACEA (Crayfish, Shrimps and Slaters)				
Astacidae (including non-native species)	**7.9**	**7.9**	**7.9**	**7.9**
Corophiidae	**5.7**	**5.8**	**5.8**	**5.8**
Asellidae	**4.0**	**2.3**	**0.8**	**-1.6**
Crangonyctidae	**3.8**	**4.0**	**3.6**	**3.6**
Gammaridae	**4.2**	**4.5**	**4.6**	**3.9**
Niphargidae	**6.3**	**6.3**	**6.3**	**6.3**
EPHEMEROPTERA (Mayflies)				
Siphlonuridae (including Ameletidae)	**11.3**	**12.2**	**12.2**	**12.2**
Heptageniidae (incl. Arthropleidae)	**8.5**	**10.3**	**11.1**	**11.1**
Ephemeridae	**8.3**	**8.8**	**9.4**	**9.4**
Leptophlebiidae	**8.8**	**9.1**	**9.2**	**9.2**
Ephemerellidae	**7.9**	**8.5**	**9.0**	**9.0**
Potamanthidae	**9.8**	**10.4**	**10.4**	**10.4**
Caenidae	**6.5**	**6.5**	**6.5**	**6.5**
Baetidae	**3.6**	**5.9**	**7.2**	**7.5**
PLECOPTERA (Stoneflies)				
Perlidae	**12.6**	**13.0**	**13.0**	**13.0**
Chloroperlidae	**11.4**	**12.2**	**12.2**	**12.2**
Taeniopterygidae	**11.0**	**11.9**	**12.1**	**12.1**
Perlodidae	**10.5**	**11.5**	**11.5**	**11.5**
Capniidae	**9.7**	**9.4**	**9.4**	**9.4**
Leuctridae	**9.3**	**10.6**	**10.6**	**10.6**

Nemouridae	**8.7**	**10.7**	**10.7**	**10.7**

	AB1	**AB2**	**AB3**	**AB4**
ODONATA (Damselflies)				
Calopterygidae (= Agriidae)	**5.9**	**6.2**	**6.2**	**6.2**
Platycnemididae	**6.0**	**6.0**	**6.0**	**6.0**
Coenagrionidae (= Coenagriidae)	**3.4**	**3.8**	**3.8**	**3.8**
ODONATA (Dragonflies)				
Cordulegasteridae	**9.8**	**9.8**	**9.8**	**9.8**
Aeshnidae	**4.7**	**4.7**	**4.7**	**4.7**
Libellulidae	**4.1**	**4.1**	**4.1**	**4.1**
HEMIPTERA (Bugs)				
Aphelocheiridae	**8.6**	**8.5**	**8.0**	**8.0**
Hydrometridae	**4.3**	**4.3**	**4.3**	**4.3**
Gerridae	**5.2**	**5.5**	**5.5**	**5.5**
Mesoveliidae	**4.7**	**4.7**	**4.7**	**4.7**
Nepidae	**2.9**	**2.9**	**2.9**	**2.9**
Naucoridae	**3.7**	**3.7**	**3.7**	**3.7**
Pleidae	**3.3**	**3.3**	**3.3**	**3.3**
Notonectidae	**3.4**	**3.9**	**3.9**	**3.9**
Corixidae	**3.7**	**3.9**	**3.7**	**3.7**
Veliidae	**4.5**	**3.9**	**3.9**	**3.9**
COLEOPTERA (Beetles)				
Gyrinidae	**8.1**	**9.0**	**9.0**	**9.0**
Scirtidae (= Helododae)	**6.9**	**6.8**	**6.8**	**6.8**
Dryopidae	**6.0**	**6.0**	**6.0**	**6.0**

Elmidae	**5.3**	**7.4**	**8.3**	**8.3**
Haliplidae	**3.6**	**3.4**	**3.4**	**3.4**

	AB1	AB2	AB3	AB4
Paelobiidae (= Hygrobiidae)	**3.8**	**3.8**	**3.8**	**3.8**
Dytiscidae	**4.5**	**4.8**	**4.8**	**4.8**
Hydraenidae	**8.5**	**10.5**	**10.5**	**10.5**
Hydrophilidae	**5.8**	**8.8**	**8.8**	**8.8**
Noteridae	**3.2**	**3.2**	**3.2**	**3.2**
MEGALOPTERA				
Sialidae	**4.2**	**4.4**	**4.4**	**4.4**
NEUROPTERA, PLANIPENNIA				
Sisyridae	**5.7**	**5.7**	**5.7**	**5.7**
TRICHOPTERA (Caddis-flies - caseless)				
Philopotamidae	**11.2**	**11.1**	**11.1**	**11.1**
Polycentropodidae	**8.2**	**8.1**	**8.1**	**8.1**
Hydropsychidae	**5.8**	**7.2**	**7.4**	**7.4**
Glossosomatidae	**7.8**	**7.6**	**7.2**	**7.2**
Psychomyiidae	**5.8**	**5.7**	**5.7**	**5.7**
Rhyacophilidae	**8.1**	**9.2**	**8.3**	**8.3**
TRICHOPTERA (Caddis-flies - cased)				
Odontoceridae	**11.1**	**10.3**	**10.3**	**10.3**
Lepidostomatidae	**9.9**	**10.3**	**10.2**	**10.2**
Goeridae	**8.8**	**8.8**	**9.4**	**9.4**
Brachycentridae	**9.6**	**9.5**	**8.9**	**8.9**
Sericostomatidae	**8.9**	**9.4**	**9.5**	**9.5**
Beraeidae	**8.8**	**7.3**	**7.3**	**7.3**

Molannidae	**6.5**	**7.6**	**7.6**	**7.6**
Leptoceridae	**6.7**	**6.9**	**7.1**	**7.1**

	AB1	**AB2**	**AB3**	**AB4**
Phryganeidae	**5.5**	**5.5**	**5.5**	**5.5**
Limnephilidae (including Apataniidae)	**5.9**	**6.9**	**6.9**	**6.9**
Hydroptilidae	**6.1**	**6.5**	**6.8**	**6.8**
DIPTERA (True flies)				
Simuliidae	**5.5**	**6.1**	**5.8**	**3.9**
Tipulidae (including Cylindrotomidae, Limoniidae & Pedicidae)	**5.4**	**6.9**	**6.9**	**7.1**
Chironomidae	**1.2**	**1.3**	**-0.9**	**-0.9**
Athericidae	**9.3**	**9.5**	**9.5**	**9.5**
Ceratopogonidae	**5.4**	**5.5**	**5.5**	**5.5**
Chaoboridae	**3.0**	**3.0**	**3.0**	**3.0**
Culicidae	**2.0**	**1.9**	**1.9**	**1.9**
Dixidae	**7.0**	**7.0**	**7.0**	**7.0**
Dolichopodidae	**4.9**	**4.9**	**4.9**	**4.9**
Empididae	**7.0**	**7.6**	**7.6**	**7.6**
Ephydridae	**4.4**	**4.4**	**4.4**	**4.4**
Muscidae	**4.0**	**2.6**	**2.6**	**2.6**
Psychodidae	**4.5**	**3.0**	**3.0**	**3.0**
Ptychopteridae	**6.4**	**6.4**	**6.4**	**6.4**
Rhagionidae	**9.6**	**9.6**	**9.6**	**9.6**
Sciomyzidae	**3.4**	**3.4**	**3.4**	**3.4**
Stratiomyidae	**3.6**	**3.6**	**3.6**	**3.6**
Syrphidae	**1.9**	**1.9**	**1.9**	**1.9**

Tabanidae	**7.1**	**7.3**	**7.3**	**7.3**

Appendix 6

Land use categories with total area in Hectares for 2006 and 2012, as designated from CORINE land use data. Total area is 36606.535 Hectares for 20016 and 37229.235 hectares for 2012. Total area has increased by 622.7006 hectares from 2006. This is mainly due to Construction sites and Sparsely vegetated areas being designated an area value. * denotes a non-designated land use for the year 2006. Percentage change between land use categories, as designated by CORINE, from 2006 and 2012 is shown in the final column.

2012 CORINE Land Use	Total Area (Hectares) for 2006	Total Area (Hectares) for 2012	% Change (Area)
Airports	471.0442	463.4982	-0.016019728
Broad-leaved forest	1180.4594	551.5762	-0.532744455
Coniferous forest	1162.5042	3050.967	1.62447826
Construction sites*	0	13.2466	0
Discontinuous urban fabric	3008.6476	3894.1845	0.294330549
Dump sites	559.3378	471.3432	-0.157319244
Green urban areas	150.5088	177.4742	0.179161617
Industrial or commercial units	1659.6954	1712.8427	0.032022322
Intertidal flats	7.3059	7.7486	0.060594862
Land principally occupied by agriculture, with significant areas of natural vegetation	60.8659	104.6006	0.718541909
Mineral extraction sites	256.0172	337.0339	0.316450223
Mixed forest	388.7411	500.5346	0.287578288
Moors and heathland	4492.3999	1278.0509	-0.715508208
Natural grasslands	1746.5042	1377.9388	-0.211030354
Non-irrigated arable land	9819.4055	8532.5647	-0.131050785
Pastures	7912.6355	8861.4711	0.119913978
Peat bogs	739.7961	2537.8654	2.430493078
Road and rail networks and associated land	113.0435	144.9999	0.282691176
Sparsely vegetated areas	0	60.9152	0
Sport and leisure facilities	912.1788	1154.9485	0.266142669
Transitional woodland-shrub	1834.6175	1865.5237	0.016846127
Water bodies	130.8262	129.9068	-0.007027644

Appendix 7

Macroinvertebrate family abundances that were found at the sites sampled with site location and BMWP score.

Taxon	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18
Ancylidae		1		2	6	1	3		2	3	1		5	2			7	2
Asellidae	5	1	18	36	36	7	2	5			4		1					
Baetidae	33	29	13	7	45	8	10	2	67	32	22	4	7		2	54	96	86
Brachycentridae										3								
Caenidae	1																	
Capniidae	1						4						10			8	9	
Chironomidae	213		538	465	74	241	3		4	9		5	2	1	2	12	68	25
Dipternan			7	4	3	9	1									1	7	1
Dryopidae													1	1				
Dytiscidae (Adult)			2	2		2	3	4		2	3	12				1		1
Dytiscidae		1																
Ecnomidae															1			
Elmidae (Adult)											3							
Elmidae									6	2		2	5				8	
Ephemerllidae											7						53	6
Erpobdellidae			18	4	34	3		1	1		1	35					1	
Gammaridae	19	11		37	13	2	19	22	3	2	6		45	1	2	12	24	2
Glossiphonidae					2			1										
Haliplidae (Adult)	2																	
Haliplidae				14		16				2								
Helodidae													1					
Heptageniidae	2				2				3	3	4	1				2	3	
Hydrobiidae				4				8			10	30						
Hydropsychidae	3		2	1	2		2		3								2	
Hygrobiidae (Adult)	2					8	11							3				
Lepidostomatidae			1			1												
Leuctridae	3						1		2				7		3	13	16	
Limnephilidae			1		4				1			1	40			2		
Lymnaeidae	4	7	140	151	13	65				1	1	7						
Nemouridae					1		1		2					1		5	1	
Neritidae			18		4							1					3	
Odontoceridae					2	1			16	4		4					4	
Oligochaetae	11	12	6	7	9	15	3	3	38	23	64	27	42	1	4		10	2
Perlodidae	5	9	1		2		5		19	4		1	15	2	4	27	49	
Philopotamidae	3	1										1						1
Phryganeidae	1																	
Physidae													7		1			
Planorbidae															1			
Polycentropodidae	1						2											1
Potamanthidae									10							1		
Psychomyidae	3		2						6	5								
Rhyacophilidae	38	23	28	8	23	1	17		10	1						20	3	5
Sialdidae		3										1		2				
Simulidae	12		82	3	85				33	2			1			6		
Sphaeridae													2					
Taeniopterigidae		1					1											
Tipulidae (Carnivore)	4	1	12		9	2			1				3	1		2	7	1
Tipulidae (Detrivore)	8	3	14	1	16	1	1		1	1			25		1	10	4	
Unionidae			5		2													
Veliidae, Mesoveliidae, Hebridae, Saldidae								1					1	4	2			

Reference sheet for sites in the invertebrate abundance table located above. HP refers to High Phosphorus input site and LP refers to Low Phosphorus input site.

Site Number	Site Name
1	Seafield#1 Riffle HP
2	Seafield#1 Pool HP
3	Seafield#2 Riffle HP
4	Seafield#2 Pool HP
5	Seafield#3 Riffle HP
6	Seafield#3 Pool HP
7	Seafield#4 Riffle HP
8	Seafield#4 Pool HP
9	Newbridge Riffle HP
10	Newbridge Pool HP
11	Kirkliston Riffle HP
12	Kirkliston Pool HP
13	Source LP
14	Harthill#1 LP
15	Harthill#2 LP
16	Polkemet LP
17	Almondell LP
18	Lin's Mill LP

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