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The use of the River Invertebrate Classification Tool (RICT) in the Foyle and Carlingford catchments, its application in management and the link between biotic index scores and fish density grades

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Non-technical Summary

THE USE OF THE RIVER INVERTEBRATE CLASSIFICATION TOOL (RICT) IN THE FOYLE AND CARLINGFORD CATCHMENTS, ITS APPLICATION IN MANAGEMENT AND THE LINK BETWEEN BIOTIC INDEX SCORES AND FISH DENSITY GRADES.

Application of RICT in river management and the link between ecological quality and salmonid density

BACKGROUND

The freshwater systems of Great Britain and Ireland represent a significant asset in terms of economic, social, cultural, scientific and biological value. However, increasing human activity has influenced and affected the natural functioning of these ecosystems, and efforts to reduce and remediate human influence is contingent upon our understanding of the links between different biological elements. The concept of 'good' health for a river system most commonly associates the biological traits of a river to that of a minimally impacted river. Definitions of health are usually applied with respect to the aspect of the river in question (e.g. fish species present, aquatic plants and invertebrates).

This report aimed to identify the link between two commonly used metrics of river biological health, namely salmonid fish (Atlantic salmon, *Salmo salar*, and trout, *Salmo trutta*) density and macroinvertebrate water quality grades. This information is central to improving our understanding of how to critically assess the biological conditions of rivers in the U.K. and is linked directly with Water Framework Directive targets of achieving 'good' ecological status in 2015.

MAIN FINDINGS

Macroinvertebrate data in the form of biotic index scores collected from the Foyle and Carlingford catchments of Northern Ireland from 2008 to 2010 was used in the application of the River Invertebrate Classification Tool to assess the ecological water quality of 52 sites. The River Invertebrate Classification Tool (RICT) is a fundamental component of water quality assessments used by UK agencies under the Water Framework Directive (WFD). The River Invertebrate Classification Tool produced site-specific ecological quality grades from 'bad' to 'high' which enabled the identification of the type of pollution or degradation present and to what extent sites are deviating from 'pristine' reference conditions. Sites where the deviation had reached critical conditions require further investigation.

However the quality grades produced from the macroinvertebrate N-TAXA biotic index scores which initially suggested that all sites were experiencing high exposure to toxic pollution and/or environmental degradation were deemed to be an under-representation of the 'true' quality of sites after consultation with ground biologists. It was concluded that the methodology adopted into routine sampling prior to the RICT analysis did not conform with standardised RICT methodology and was isolated as a contributing factor for the low quality grades.

Only one site across the three years presented with an ASPT ecological quality grade of poor, indicating exposure to levels of organic pollution capable of seriously degrading the macroinvertebrate community richness. The remaining sites from 2009 through to 2011 had ASPT quality grades of moderate or higher.

The macroinvertebrate biotic index scores were also used with juvenile salmon and trout density grades calculated from data collected through semi-quantitative electrofishing for the same sites. Correlations between biotic index scores and 0+ density grades were assessed using linear regression analyses which presented with results suggesting that there was a lack of correlation between macroinvertebrate index scores and juvenile fish density grades. This suggested that the macroinvertebrates and 0+ salmonids are responding differently to levels of pollution exposure.

This further emphasises the need for monitoring systems which cover all component members of river ecosystems and thus the adoption and application of the River Invertebrate Classification Tool

in routine surveillance would enable monitoring of the macroinvertebrate community and assess water quality in compliance to the WFD.

Recommendations for the future application of River Invertebrate Classification Tool

Changes to the biological sampling procedure currently adopted into routine sampling to adhere to the strict specifications of the RICT methodology.

- The addition of a 1-minute manual search into biological sampling.
- Sample sites within the months of spring and autumn to prevent a single season bias.
- Consideration of sampling sites locations to avoid the influence of anthropogenic structures on community structures and N-TAXA scores.

Ensure that the identification process is completed under laboratory conditions to coincide with standardised RICT identification requirements.

Modifications to the environmental data collection currently adopted into routine operations.

- A record taken of time variant environment variables from across the three predefined RICT seasons: spring, summer and autumn to produce annual readings.
- The adoption of measuring time invariant environmental variables with the use of GIS software to ease processing of a large number of sites.

The use of the River Invertebrate Classification Tool (RICT) in the Foyle and Carlingford Catchments, its application in management and the link between biotic index scores and fish density grades.

Courtesy of the Loughs Agency.

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ABSTRACT

Rivers are subject to many elements of stress which can detrimentally impact the natural functioning of the aquatic ecosystems. Recently, classification tools have been developed and adopted into the monitoring systems of the United Kingdom to adhere to the requirements asked by the Water Framework Directive. ASPT and N-TAXA scores derived from the macroinvertebrate data collected by the Loughs Agency from 2009-2011 was used with the River Invertebrate Classification Tool to assess the quality of river sites from across the Foyle and Carlingford catchments situated in Northern Ireland and the Republic of Ireland. The results indicated the ecological quality of each site and the type of anthropogenic disturbance deteriorating quality. The ASPT grades produced by RICT indicated that all the sites across both catchments did not greatly deviate from 'pristine' reference conditions although this deviation varied across all three years. The N-TAXA grades suggested a substantial deviation of all sites from reference conditions with regards to levels of toxic pollution, however the N-TAXA grades were deemed as under representations of the true quality. It was deduced that the disparity in the N-TAXA grades to those of the 'true' ground quality grades was linked to a different methodology used to those specified by RICT. The ASPT and N-TAXA scores were also paired with 0+ salmon and trout density grades gathered from semi-quantitative surveys, to assess whether changes in fish density grades correlated to changes in the biotic index scores. The linear regression analyses of the salmon and trout 0+ density grades did not significantly correlate with changes in the ASPT and N-TAXA scores (P= > 0.005) for 2009. For the N-TAXA scores and salmon density grade for 2010 there was a significant relationship between the two (P= 0.010), however they did not strongly correlate (r-sq= 0.1423). From this it was concluded that the juvenile fish density grades did not predictably change with ASPT and N-TAXA changes. As a result it was concluded that producing site-specific ecological quality grades derived from RICT would aid the identification of type-specific stressors and provide an alternative view to the functioning of a river's ecosystem.

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1 INTRODUCTION

The inland freshwater networks, transitional waters and sea loughs of Great Britain and Ireland are a significant natural asset, as with every country, in terms of their economic, social, cultural, scientific and biological value (Dudgeon et al, 2006; Schofield & Davies, 1996). The river basins of Ireland have a rich biodiversity of native and introduced aquatic species, which range from residential to anadromous species, with some species having a significant ecological and economic value to local industry such as the Atlantic salmon (*Salmo salar*) and trout (*Salmo trutta*) (Indecon International Economic Consultants, 2003; Environmental Protection Agency Biodiversity Team, 2010). However, an environments ability to host and support a variety of organisms and a naturally functioning ecosystem can be strongly influenced and reduced with increased pressure from human activity (Richter et al, 1996; Nel et al, 2007). This has been most prominent through abstraction for drinking water and industrial use as well as waste removal, dredging and the construction of weirs (Smith et al, 1999; Clarke et al, 2003). Elements of anthropogenic pollution such as, sewage, heavy metal pollution, nutrient enrichment and agricultural runoff can also result in varied responses of rivers ecosystems (Dewson et al, 2007; Allen & Hardy, 1980; Baylay & Williams, 1973; Graf, 2006). In view of the historic and current use of river systems, considering the associated organisms have restricted mobility, it is no wonder that these closed systems often exhibit unsatisfactory health (Lammert & Allan, 1999; Smith et al, 1999). While there is no universal definition for the 'health' of a river, there are various definitions provided (see Norris & Thoms, 1999). However most follow the concept of 'good' health as a river exhibiting similar biological traits to a minimally impacted river of the same calibre (Schofield & Davies, 1996). Although, definitions of health are usually applied with respect to the aspect of the river in question, hence the variety of definitions available (See Acreman & Dunbar, 2004, for various examples). The current European water legislation, the Water Framework Directive (WFD) specifies a definition of 'good' health as equating to values of the biological aspects of water which only slightly deviates from natural conditions with a minimal indication of human activity. It also defines the 'ecological condition' of a running water system as the functionality and structural content of aquatic ecosystems as expressed as quality bands (The European Parliament and of the Council, 2009). The use of the definitions provided by the WFD was deemed the most appropriate as this paper concerns itself with ecological assessments of sites in Northern Ireland using the River Invertebrate Classification Tool (RICT). This classification tool was developed to aid the classification of sites in compliance to the WFD (The European Parliament & Council, 2009).

The concept of assessing river health using biological elements has been around since Kolkwitz and Marsson's demonstration of the Saprobic system in 1902. Many government and independent bodies have since adopted methods of assessing 'health' and the 'ecological condition' into routine surveillance (Kolkwitz & Marsson, 1902; Bartram & Ballance, 1996). Furthermore, with the implementation of the WFD

in 2000 the importance of biological monitoring has attained recognition to a continental scale. (European Commission, 2000; Logan & Furse, 2002).

To establish the ecological condition of a river, the communities which form part of the ecosystem in question must be taken into account. That is, a method of measuring the responses of these communities to a variety of different stressors or pollutants should be applied. Many taxonomic groups are commonly used to assess the condition of surface waters in addition to chemical and physical assessments (Clarke et al, 2003; The European Environmental Agency, 2011). Macroinvertebrates have long proven their worth as a useful indicator of the ecological condition of bodies of water, possessing a great deal of features that make them ideal candidates (Wright et al, 1996). Clarke et al (2003) stresses these, drawing on key features such as their widespread occurrence in the majority of river and stream types, sampling does not require expensive techniques, and the identification of marcoinvertebrates to the taxonomic level required for quality assessments is relatively straightforward. Also, through extensive studies of pollution and disturbance tolerances of different families, their response is well understood (Walley & Fontama, 2000). The life cycle lengths of many invertebrate species correlates to the time required to identify prolonged environmental stresses (Clarke et al, 2003). Wright (1994) emphasised the benefits of this by stipulating that macroinvertebrates would respond to sporadic stressors, where chemical tests would only indicate the condition at the time of sampling. However Barbour et al (1999) noted that higher taxa, such as fish, can be measurably more sensitive to toxic substances than macroinvertebrates due to accumulation associated with prolonged exposure, although it is acknowledged that this will only reflect the presence of pollution some time after its introduction. Furthermore, macroinvertebrates as a whole have a rather limited mobility within their geographic range and have the highest abundance within their optimum environmental conditions (Johnson & Sandin, 2001). However, Wright et al (1996) highlights complications associated with the use of macroinvertebrates citing work conducted by Furse et al, (1993) that macroinvertebrate assemblages of headwater streams were significantly different to other areas of river systems and needed to be accounted for. Despite these complications, macroinvertebrates are still readily used due to their known responses to pollution and the wealth of knowledge surrounding them (European Environmental Agency, 2011). Within United Kingdom their distributions and community structures have been extensively studied and are well defined (Wright et al, 1996) with the use of macroinvertebrates in classification systems producing standardised data comparable at a national scale (Wright et al, 2000).

Macroinvertebrate biological indices have become a fundamental component of the biological aspect of water quality assessments in the UK and Europe (Metcalfe, 1989; Sandin & Hering, 2004). The most commonly used indices in the UK are formalised from the Biological Monitoring Working Party system, as the BMWP score (total score of all BMWP families present), N-Taxa (the number of BMWP scoring families) and ASPT (Average score per taxa) (Clarke et al, 2002). The use of component indices of the

BMWP system can provide information on the type of disturbance acting on a site. The ASPT index shows strong correlations with variables of organic pollution: dissolved oxygen (mg.l-1 and % saturation); Ammonia levels (mg.l-1) and biochemical oxygen demand (mg.l-1) (Clarke et al, 2011) where the N-TAXA index indicative of toxic pollution and environmental degradation (Anon, 2008; Dixon, 2010). The BMWP system along with previously used indices such as the Trent Biotic Index and the Chandler Biotic Index, are scoring systems based on the known sensitivity of specific families and species to disturbances or stressors (Bartram & Ballance, 1996). The BMWP system allocates a score between 1 and 10 for each BMWP family present, with high scores awarded to intolerant families (Leptophlebiidae) and low scores to tolerant families (Chironomidae) (Johnson et al, 1993; Clarke et al, 2002). Biological indices based on scoring systems allow for the allocation of a site-specific score in accordance to their perceived exposure to stress which has proven useful for policy implementation (Friedrich et al, 1996).

The current approach used to assess the ecological quality of rivers with the use of macroinvertebrates by UK government bodies follows on from the River InVertebrate Prediction And Classification System (RIVPACS) (Davy-Bowker et al, 2008). The RIVPACS approach is used to compare the observed macroinvertebrate fauna of a site to the expected fauna of the same site, if it were experiencing a minimum level of environmental stress (Clarke et al, 2002). This system used a catalogue of minimally impacted, high quality reference sites (Clarke & Murphy, 2006), from which the expected fauna is calculated (Centre for Ecology & Hydrology, 2010; Wright et al, 1998). These reference sites are made up from a broad variety of the best examples of different river and stream types, geographical ranges and altitudes throughout Great Britain and N. Ireland (Clarke & Murphy, 2006; Wright et al, 1998). The RIVPACS approach incorporates the use of the previously mentioned BMWP system and its associated indices to produce environmental quality indices (EQI) from the observed/expected ratio (O/E ratio) (Clarke et al, 2002; Johnson & Sandin, 2001). The EQI values could then be classified by their lower limits to produce a quality grade for each site. Consequently, RIVPACS became a prominent predictive system used by the regulatory bodies of United Kingdom, (SEPA, EA and NIEA) to assess water quality as part of management schemes (Davy-Bowker et al, 2008). It was the successful development and implementation of RIVPACS within the field that lead to the WFD instigating the use of observed/expected ratios and reference conditions as a method of assessing quality (Centre for Ecology & Hydrology, 2010). However, as the development of RIVPACS III+ predates the WFD, many of its functions did not comply with the quality level required and it could not support additional requirements asked of the WFD. This recognition lead to the development of the River Invertebrate Classification Tool (RICT). While many of the initial features of RIVPACS was retained with the RICT, the need for additional features and options was apparent. The RICT system offers increased functionality, with the ability to assess multiple years data within a single run, the choice from more taxonomic levels, a more extensive list of biological indices and includes predictions of the probabilities of abundance (Davy-Bowker et al, 2008). The RICT system also operates by creating ecological quality ratios

(EQR) of the observed metric values over the predicted metric values. This is a similar procedure to the creation of EQI values previously produced in RIVPACS assessments. However, sites are classified into five quality grades based on their EQR values to provide a common scale for quality assessments across regions (Van de Bund & Solimini, 2006). This is a method adopted for all the biological quality elements used as part of the overall ecological status classification of a water body under the WFD (Hatton-Ellis, 2008). With the RIVPACS III+ program the references sites were classified into 35 'end-groups' for Great Britain and seven groups for N.Ireland to account for distinct assemblages within individual regions and river types (Smith et al, 1999). This mode of classifying the references sites into a regional framework was also retained, however two new end-groups were developed after a reassessment of the quality of the reference sites in compliance to the standards set by the WFD (Davy-Bowker et al, 2008).

The operation system of the RICT was updated to adopt modern software and classifications based on two or more indices such as the MINTA system (Davy-Bowker et al, 2008). The MINTA system is used as part of the general quality assessments within the UK which classes a site based on the poorest grade as derived from the ASPT and N-TAXA EQR score, often referred to as the 'one out, all out' approach (Clarke, 2009). The new RICT program which is available to all, allows individuals, agencies and bodies across the UK to approach the assessment of ecological quality in a way that does not divert greatly from previously established methods, such as in the RIVPACS program. The results derived from RICT produces comparable data at local and regional scales which can be used in compliance to the assessment of water systems for the WFD (Davy-Bowker et al, 2008).

The Loughs Agency are a government body who are entrusted with the protection, management and conservation of the rivers, transitional waters and sea systems of the Carlingford and Foyle loughs catchments of the Republic and Northern Ireland. The Foyle catchment lies in the Northwest of Northern Ireland crossing the border into the Republic of Ireland, with the Carlingford catchment lying to the Southeast of Northern Ireland and Northeast of the Republic of Ireland. The Agencies priority lies in the current and future development of aquaculture, recreational and commercial fisheries with an emphasise in the native Atlantic salmon and trout stocks (Niven et al, 2010). As an effort to apply an appropriate management scheme, the agency conducts annual measurements and surveys of habitat grades, catchment land use, water chemistry, salmon and trout redd counts, adult fish numbers with electronic fish counters, smolt trapping, semi-quantitative and quantitative electrofishing of juvenile fish, and sets conservation limits and spawning targets for adult salmon (Niven et al, 2010). The agency's semi-quantitative electrofishing surveys throughout the two catchments acts as a means to monitor the juvenile salmonid population. This approach assesses the densities of 0+ salmon and trout fry at a vast number of sites. Where the Loughs Agencies prime efforts lie mainly in the monitoring of target fish stocks, the importance of maintaining and

protecting the natural biodiversity of fauna and water quality of each river basin is recognised and incorporated into their management scheme.
The Loughs Agency uses data on non-target species collected as a by-product of annual surveys and conducts tributary level chemical and biological water quality assessments each year (Niven et al, 2010). These surveys and assessments of non-target species can provide additional information on the functioning of river systems and may prove crucial for identifying the causes behind why fish populations are underproducing. The biological aspect of the water quality assessments involves the collection of macroinvertebrate samples from a number of sites, identifying each specimen to family level and applying the BMWP system for assessment.

As to aid the Loughs Agency with their biological assessment of water quality, the macroinverterbrate data collected from 2009 to 2011 will be used with the River Invertebrate Classification Tool. This will produce ASPT and N-TAXA ecological quality ratios (EQR) and quality grades for each site within the Foyle and Carlingford catchment. The production of grades will aid identification of sites under exposure to type-specific stressors. The appropriateness of the Loughs Agency current macroinvertebrate sampling procedure along with the potential of RICT within the Loughs Agencies current management scheme, its merits of use and the ability of moving on to assessments in compliance with the WFD will be discussed. The ASPT and N-TAXA scores will also be used with the juvenile 0+ salmon and 0+ trout data collected during the semi-quantitative electrofishing for the same sites and the same years as the macroinvertebrate sites. It will be assessed whether the ASPT and N-TAXA score for each site can be used to predict the quality grade of a semi-quantitative electrofishing for the same sites.

2 METHODOLOGY

The areas covered in this study are the Foyle and Carlingford catchments of Northern Ireland and the Republic of Ireland with the collection of the macroinvertebrate data and the electrofishing data conducted during the years 2009 to 2011.

2.1 Macroinvertebrate Collection

For the years 2009-2010 there were 54 point-sampling sites and for the year 2011 there were 28 point-sampling sites taken across the Foyle catchment, each represented by a ASPT and a N-TAXA grade. For the Carlingford catchment there were 5 point-sample sites in the year 2009 and 2 point-samples taken in 2010 with no sites represented for 2011.

The macroinvertebrate data was gathered by members of the Loughs Agency, during the recognised RICT seasons of summer (June-August) and autumn (September-November) for the years 2009-2011. The methods of collection and identification of the invertebrates described are based on the standard practices established by the Loughs Agency. The vast majority of sites were positioned in close proximity to bridges or roads as a by-product of gaining easy access to river stretches. Members of the Lough Agency conducted 3-minute kick-samples with standard pond-nets for each site, sampling each habitat type within the sample area in accordance to its presence which falls in line with the RICT kick-sampling procedure. Before each kick-sample is conducted, the RICT procedure requires a 1 minute manual search for invertebrates. This time should be divided between sampling the water surface and collecting invertebrates attached to submerged substrate or vegetation which may be missed during the kick-sampling. This part of invertebrate collection was not adopted into the Loughs Agency procedure and thus not completed. The implications of this will be discussed later. Throughout the kick-sampling the nets were periodically emptied into holding buckets if the sample size was restricting further collection. Each sample collected was then stored in a labelled container for sorting back at the laboratory for the years 2009-2010. After each site the nets were washed to ensure that there was no material or specimens left in the nets to prevent contamination of the next sites sample. The sample was then either stored and returned to the Lough Agency laboratory for identification, or in the case of the 2011 samples, the identification process was conducted at the side of the river. This approach adopted in 2011 goes against the RICT recommendation for all samples to be process in a laboratory.

The identification procedure carried out by the Loughs Agency for the years 2009 to 2010 was in compliance with the RICT procedure however as previously noted, the 2011 samples were sorted at the river side with the use of sorting trays and identification guides. The samples for the years 2009 and 2010 were processed and identified to family level, with the use of sorting trays, appropriate lighting and identification keys and guides in a laboratory. Of the families that were identified within each sample, a separate record was made of BMWP families present. The BMWP score for each sample was calculated by adding up the sensitivity scores of each BMWP family present within the sample; see Appendix I for the BMWP family sensitivity scores. The N-TAXA (number of taxa) score for each site was taken as the total number of BMWP families present within each sample. From the BMWP score and the N-TAXA score, the ASPT (average score per taxon) score can be calculated. The ASPT score was derived by dividing the BMWP score over the N-TAXA score, which gives the sample's average sensitivity score. The raw BMWP family lists were then entered into an excel sheet formatted for the use in the RICT.

2.2 Environmental variable measurement

Wherever possible, the methodologies and equipment required by RICT procedure for data collection ran parallel with the 'UK invertebrate sampling and analysis procedure for Star Project' manual by (Murray-

Bligh, 1997) provided with the RICT software. If recommended procedures could not be followed, alternative methods were used and will be subsequently noted. The data was collected by different surveyors, either working for the Loughs Agency (as part of routine sampling) or by surveyors contributing to this report.

Each invertebrate collection site was given a site ID and the coordinates were recorded with a handheld Trimble GPS device. The coordinates were then converted into the 6-figure Irish grid reference format required for RICT analysis. The Irish grid reference for each site was checked against Northern Irish and Irish ordnance survey discoverer series maps to ensure that these matched the tributary and river name for that site. The Foyle and Carlingford catchments lie in both Northern Ireland and the Republic of Ireland, thus sites in the data set are representative of both countries. After consultation with the Centre for Hydrology and Ecology, the group behind the development of RIVPACS and the developers of the new RICT program it was decided to include the sites lying in Ireland, in light of the requirement of the Lough Agency to assess these sites for quality regardless of their positioning/nationality. The following environmental measurements were collected as to comply with the RICT procedure, however the time variant environmental data: depth, width, current velocity, electrical conductivity and substrate cover, were not representative of annual conditions. The environmental data was collected in late January and early February of 2012 and substituted in for the appropriate sites for the years from 2009 to 2011.

The mean depth at each site was taken by three depth measurements at regular intervals along the same transect (at a quarter, half and three-quarter length of the transect) across the stream or river according to whether it was at high or low flow. The mean depth of the three readings was used as the final value in the RICT. The measurement of a rivers width was taken from the wetted edge of the river using a measuring tape held taut across the modal transect within the sampling area. The measurement of electrical conductivity was used in the absence of annual mean alkalinity readings. The electrical conductivity readings at each site was taken on a handheld device just upstream of other measurements as to avoid interference from water disturbance. The use of water current velocity categories was also used as a surrogate due to the lack of data of the annual discharges for each site. The current velocity was measured by marking out a set distance and recording the time taken for a floating object to flow past the marked distance. The current velocity readings were then categorised into one of the five predetermined RICT velocity categories shown in table 1.

Table 2.1 - The current velocity categories used by the River Invertebrate Classification Tool (Murray-Bligh et al, 1997)

Current Velocity Category	Current Velocity (cm.sec-1)
1	≤ 10
2	> 10 - 25
3	> 25 - 50
4	> 50 - 100
5	> 100

The substrate cover of each site was measured by estimating the percentage cover of the following four categories: silt/clay, sand, pebble/gravel and cobble/boulder with the sum of these adding up to 100%. The substrate cover estimation was subjective to the opinion of the individual collecting this information and thus the percentage cover estimation was conducted by the same individual.

The time invariant environmental measurements, altitude, slope and distance from source were taken from 1:50 000 scale Northern Ireland and Ireland ordnance survey maps. The altitude of each site was taken as the altitude of the nearest contour line to a sites national grid reference, this reading was checked twice. For each site, the distance to its furtherest source even if that tributary is of another name was measured three times with the use of a planimeter following the watercourse from the source to the site. The average of the three measurements was taken as the final value. The slope was also measured with the use of the planimeter, measuring the distance between the nearest contour line upstream and downstream crossing the watercourse of each site. The RICT procedures manual sets out alternative instructions for when the site is not positioned between contour lines and these were followed when necessary. The slope was measured in metres per kilometre and was calculated by subtracting the height in meters of the nearest contour downstream to the site from the height in meters of the nearest contour upstream divided by the distance in kilometres between the two contours. The distance between the two contour lines was checked twice and the mean distance was used. The tolerable standard error for each measurement of the distance to source and slope was checked against the parameters set out in the RICT procedures and it was ensured that each site included fell within these.

2.3 Prediction and classification

The prediction and classification option of the River Invertebrate Classification Tool was used to produce ASPT and N-TAXA EQR grades for each site. For the years 2009-2010, 54 sites across the Foyle catchment were assessed with 28 sites assessed in 2011. For the Carlingford catchment 5 sites were assessed in 2009 with 2 sites assessed in 2010. The appropriate data required to assess the 2011 Carlingford catchment was unavailable. The macroinvertebrate BMWP score, N-TAXA score and the ASPT score along with the environmental data for each site was uploaded on to the RICT program. The seasons from which the macroinvertebrate samples were collected was selected for each analysis run. For the 2009 sites the seasons option selected for analysis were summer and autumn. For 2010 and 2011, the season option selected for all sites was summer. The Northern Irish Prediction Environmental Variables (PEV) set was selected along with the New NI (11-group level) end group model for analysis. The taxonomic level selected for analysis was taxonomic level 1 (TL1), which predicts the ASPT and N-TAXA scores expected at each site if that site was in reference condition (Davy-Bowker *et al*, 2008). The RICT program takes the observed ASPT and N-TAXA scores and divides them by the expected values derived from the reference conditions to produce ecological quality ratios (EQR) for each index. The ASPT and N-TAXA EQR score produced determines the ASPT and N-TAXA grade given to each site with the borders for each grade displayed in *table 2*. The MINTA system graded each site based on the lowest grade presented by ASPT and N-TAXA grades. This approach will not be adopted in this study as it does not aid the identification of the type-specific stressor acting on each site.

Table 2.2 - The assigned status grades, description and symbol for each ecological quality ratio (EQR) for ASPT and N-TAXA values. (Anon, 2010).

EQR ASPT Score	EQR N-TAXA Score	Grade	Symbol
≥ 0.97	≥ 0.85	High	
0.86	0.71	Good	
0.75	0.57	Moderate	
0.63	0.47	Poor	
≤ 0.63	≤ 0. 47	Bad	

The ASPT and N-TAXA grade given to each site from the RICT analysis was then displayed on ordnance survey maps for the Foyle and Carlingford catchment, with a symbol grade on the location of each sites coordinates representing the grades given to that site. These maps were produced using the newest version of the Geographic Information System software (arcGIS).

2.4 Semi-quantitative electrofishing

The salmon and trout 0+ density data for each site was collected by the Loughs Agency using the semi-quantitative electrofishing methodologies described by Crozier and Kennedy (1994). This data forms a component of annual sub-catchment reports. The data used within this study only represents a subset of the total sites electrofished by the Loughs Agency as only electrofishing sites that had a corresponding macroinvertebrate sampling site were used. The months of collection were August to October for 2009 and July to October for 2010. The electrofishing data for the year 2011 was unavailable for analysis. Each electrofishing survey was conducted in a riffle mesohabitat at each site by two to three members of the Loughs Agency using mobile electrofishing backpack equipment and hand-nets. The equipment was used to fish in a downstream direction at each site concluding when the standard 5 minute time limit had been reached. Once the 5 minutes had been reached, the fish caught during the sweep were measured for fork length and then separated by species and by age group. From this information each site was graded based on the number of 0+ trout and salmon caught using the semi-quantitative electrofishing classification system in table 2. The semi-quantitative 0+ salmon and 0+ trout classification grades were then displayed on ordnance survey maps produced with arcGIS software.

Table 2.3 - The semi-quantitative electrofishing classification system used by the Lough Agency for both 0+ trout and 0+ salmon adopted from Crozier & Kennedy (1994).

Grade	Total N^{o} of 0+ Salmonids	Symbol
Excellent	25>	
Good	15-24	
Fair	5-14	
Poor	1-4	
Absent	0	

2.5 Invertebrate prediction of fish density grades

Finally, the original ASPT, N-TAXA scores and the semi-quantitative electrofishing data were used to assess whether ASPT and N-TAXA scores at a site could predict the juvenile salmon and trout electrofishing density grades at the same site. For the majority of the sites, the macroinvertebrate kick-sampling and electrofishing had been conducted at the same sampling area with matching GPS coordinates for each year. For these sites the ASPT and N-TAXA scores were paired with the salmon and trout density grades for assessment. Where a macroinvertebrate collection site did not have a corresponding electrofishing collection site, the closest electrofishing site downstream was used if it was within a defined distance of the macroinvertebrate site or if both sites had been identified as being in the same habitat category. The maximum acceptable distance between two sites was 70meters if habitat data was unavailable. This distance was selected as it matched the RICT recommended length of macroinvertebrate survey areas and observations during environmental collections suggested that within this distance the river stretch did not substantially vary. Habitat data collected by the Loughs Agency gathered in reference to the lifecycle units required for successful salmonid reproduction was used to pair macroinvertebrate sampling sites and electrofishing sites. River stretches were graded in accordance to the type of lifecycle habitat present and the quality of that habitat. The parameters of the mesohabitat type, river gradient, depth and substrate cover were used to categorise and grade each site. Any macroinvertebrate sites which did not have an electrofishing site paired with it was excluded from this part of the study. A linear regression analysis was used to assess the relationship between the ASPT and N-TAXA scores and fish density grades. A representative value was assigned to each density grade, as to pair this with the ASPT and N-TAXA scores. The values used to represent each fish grade in the linear regression analysis were as follows: 0 (absent); 2.5 (poor); 9.5 (fair); 19.5 (good) and 25 (excellent). The linear regression analysis was completed with the use of the statistical program, Minitab.

3 RESULTS

3.1 RICT EQR grades

There are considerable discrepancies between the EQR ASPT and the N-TAXA grades at each site. That said, the two biological indices are sensitive to different aspects of pollution, the ASPT index is indicative of organic pollution whereas the N-TAXA index is indicative of habitat degradation and/or toxic pollution (Bartram & Ballance, 1996). However with prior knowledge of the methodology and after consultation with several field biologists it was concluded with respect to the N-TAXA score, all sites were under-represented. All the sites had a N-TAXA score below 13 with a mean N-TAXA score of 9 for 2009 and a mean N-TAXA score of 8 for 2010. Taking this into account, it was deduced that the N-TAXA EQR grades are not a true

representative of the ecological quality at each site and as such it is imperative that the N-TAXA grades derived from this study are interpreted with caution.

The ASPT EQR grades for the Foyle catchment, figure 3.1, shows that in 2009, 35 sites presented with a high grade, 11 sites presented with grade of good with 8 sites indicating a moderate grade. With regards to the high APST EQR grades, this indicates that these sites did not greatly deviate from reference conditions suggesting a minimal level of exposure to organic pollution. However as 11 sites presented with a grade of good and 9 sites with a grade of moderate implies that these sites were exposed to levels of organic pollution was adversely deviating the macroinvertebrate assemblage further from the assemblage expected if that site were in 'pristine' condition.

The 2009 N-TAXA EQR grades for the same sites of the Foyle are displayed in figure 3.2. Taken independently from the ASPT EQR grades, the results from the N-TAXA EQR grades would under normal standardised procedures would indicate that all 54 sites were experiencing moderate to heavy exposure to some form of degradation or toxic pollution.

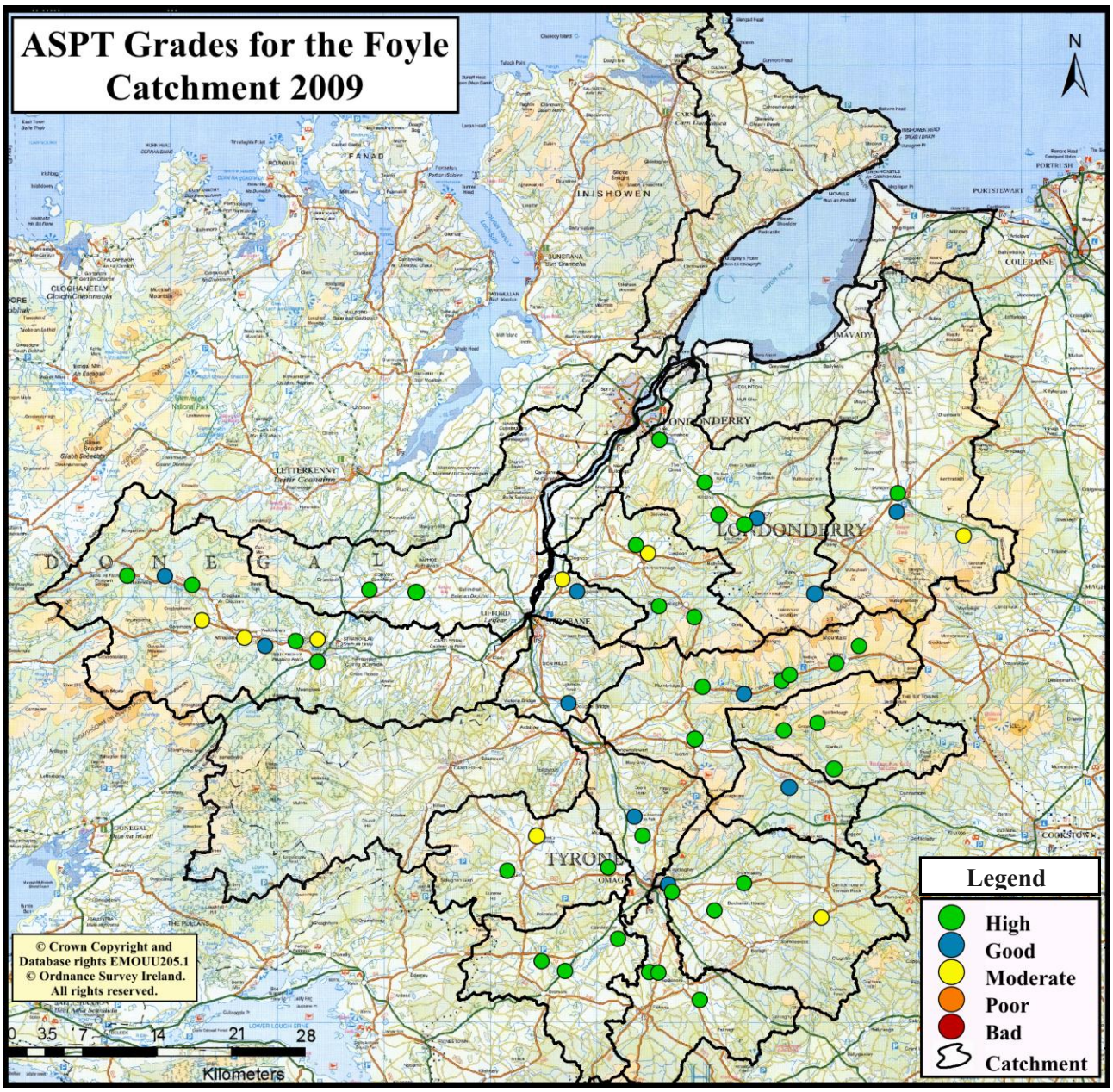

Figure 3.1 - The ASPT EQR grades for 54 sites (n=54) in the Foyle catchment, 2009. (Grades: moderate - 8 sites; good - 11 sites; high - 35 sites).

From this, these N-TAXA grades would typically suggest that the community richness of the 26 sites that presented with the lowest ecological quality grade had been severely depleted. The 7 sites with a N-TAXA EQR grade of poor would similarly indicate that these sites were exposed to levels of disturbance and/or pollution capable of substantially degrading the ecological quality. The remaining sites were graded as to having a moderate ecological grade.

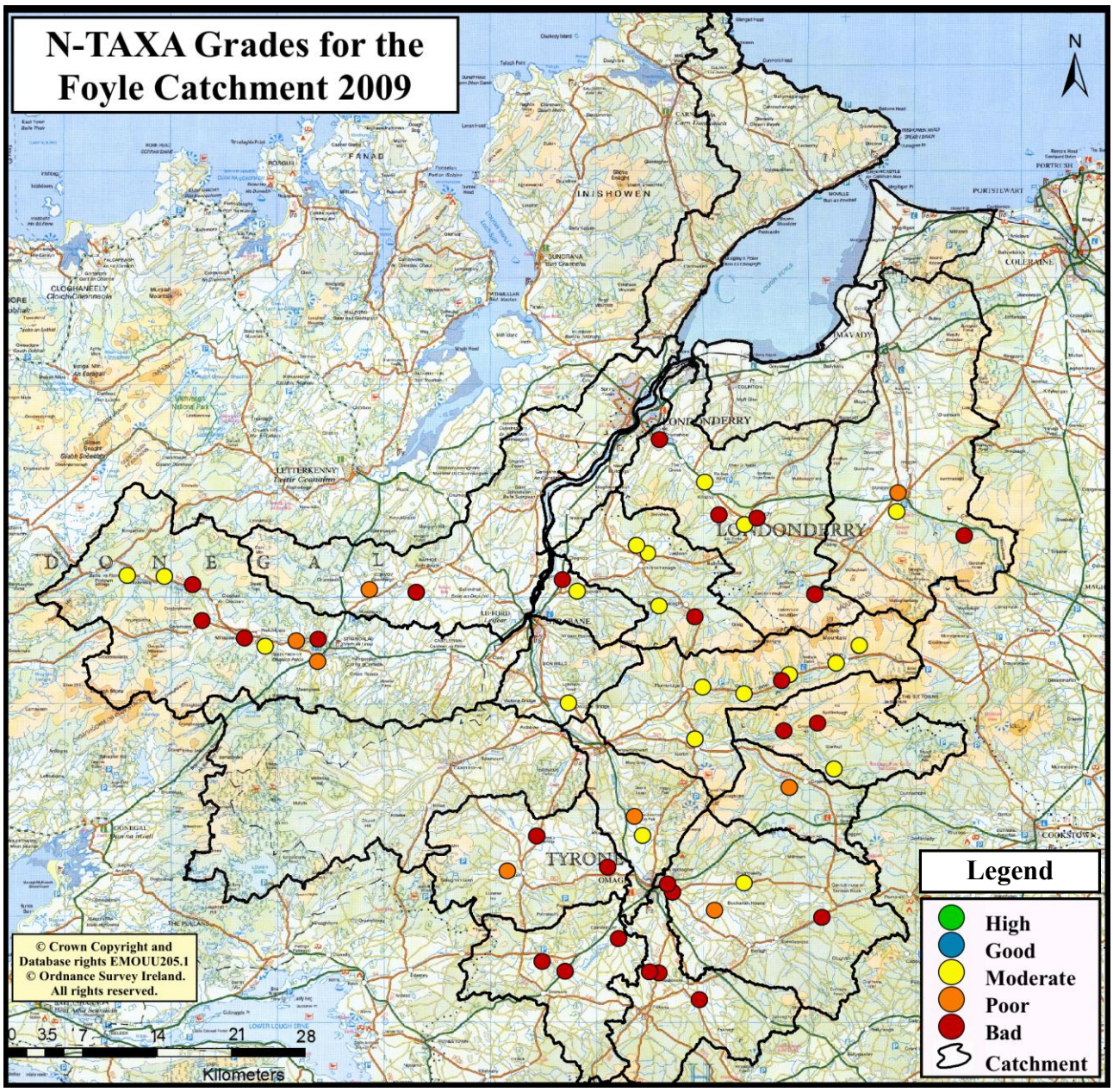

Figure 3.2 - The N-TAXA EQR grades for 54 sites (n=54) in the Foyle catchment, 2009 (Grades: bad - 26 sites; poor - 8 sites; moderate - 20 sites).

Application of the RICT methodologies these results would usually indicate that the ecological quality for all the sites across the Foyle catchment for 2009 had been adversely impacted on by exposure to toxic pollution or that each site was in the vicinity of environmental degradation. The ASPT EQR grades for the Carlingford catchment in 2009 presented with 2 sites with grade of good and 3 sites with grade of high (figure 3.3). From this, it can be concluded that the sites with a high grade were experiencing a minimal level of exposure to organic pollution. The 2009 N-TAXA EQR grades for the same sites in the Carlingford catchment are displayed in figure 3.4.

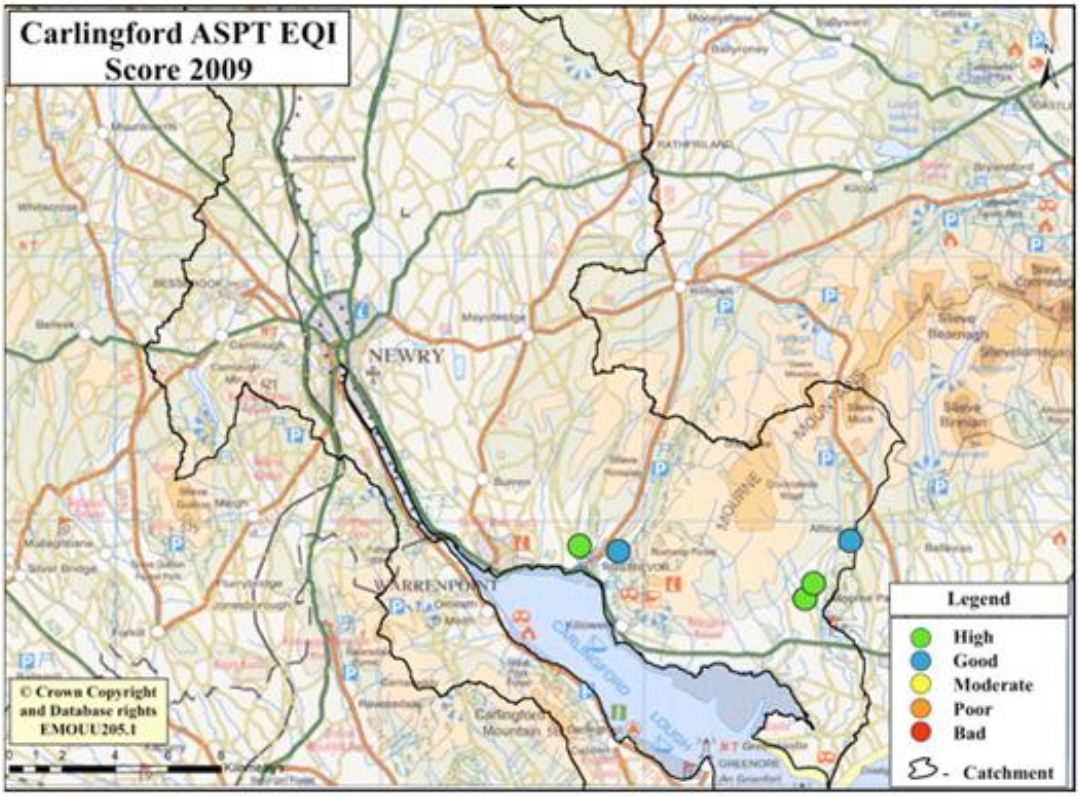

Figure 3.3 - The ASPT EQR scores for 5 sites (n=5) in the Carlingford catchment, 2009 (Grades: good - 2 sites; high - 3 sites

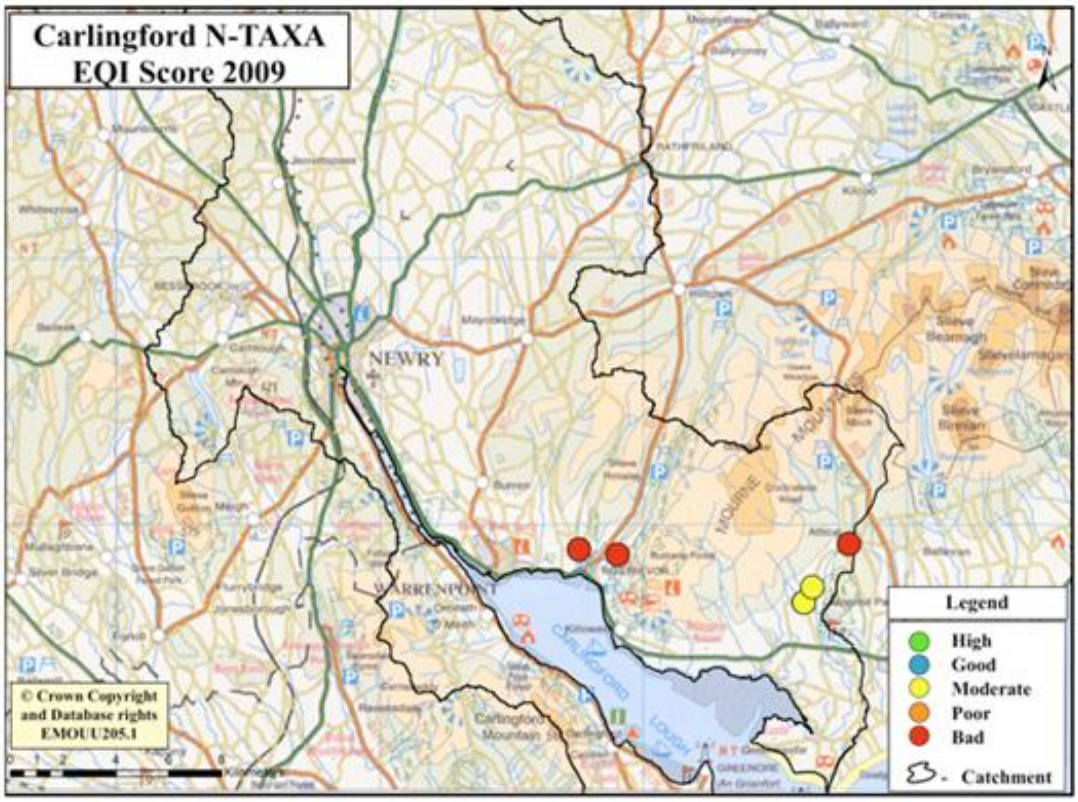

Figure 3.4 - The N-TAXA EQR for 5 sites (n=5) in the Carlingford catchment, 2009 (Grades: bad - 3 sites; moderate - 2 sites).

Of these sites, 3 sites presented with a N-TAXA grade of bad, which would indicate that these sites were positioned in an area experiencing significant environmental degradation or toxic pollution if the change in the sampling procedure had been not an influential factor. The remaining 2 sites had a N-TAXA EQR grade of moderate, would suggest that these two sites were also experiencing disturbance and/or pollution exposure however not to the same extent again if the N-TAXA EQR grades were representing the true ecological quality.

The ASPT grades for the Foyle in 2010 did not present with any sites graded as bad, with only 1 site graded as poor(figure 3.5). Of the remaining sites, 10 sites were graded as good, with 34 sites graded as high. The sites that were graded as good or high indicates that these sites were experiencing a minimal level of organic pollution however the sites with a grade of good had slightly deviated from reference conditions. The N-TAXA EQR grades for the Foyle catchment in 2010 are displayed in figure 3.6. Of the 54 sites sampled, 34 sites presented with the lowest quality grade, a N-TAXA EQR grade of bad. The remaining sites were graded as poor and moderate, with 8 sites as the former and 11 sites as the latter. Likewise to 2009, there were no sites graded higher than a moderate N-TAXA grade. These results would under normal circumstances indicate that across both years these sites were experiencing some form of exposure to habitat degradation or/and toxic pollution that the BMWP families were particularly responsive to. An overview of the ASPT EQR grades of the 54 sites in the Foyle catchment for the years 2009 (figure 3.1) and 2010 (figure 3.5) indicates that the quality grades between the two years did not remain static. The number of sites that were graded as high differed between the years, with 35 in 2009 to 34 sites in 2010. Sites graded as good dropped from 2009, with 11 sites to 10 sites in 2010. The number of sites with a ASPT EQR grade of moderate also shifted, with 8 sites in 2009 and 9 in 2010 with one site dropping to a grade of poor in 2010. This only illustrates the overall number of each grade for the two years, where the grades of only 34 sites remained constant between the two years. With the remaining sites either dropping or increasing in their ASPT quality grade from 2009 to 2010.

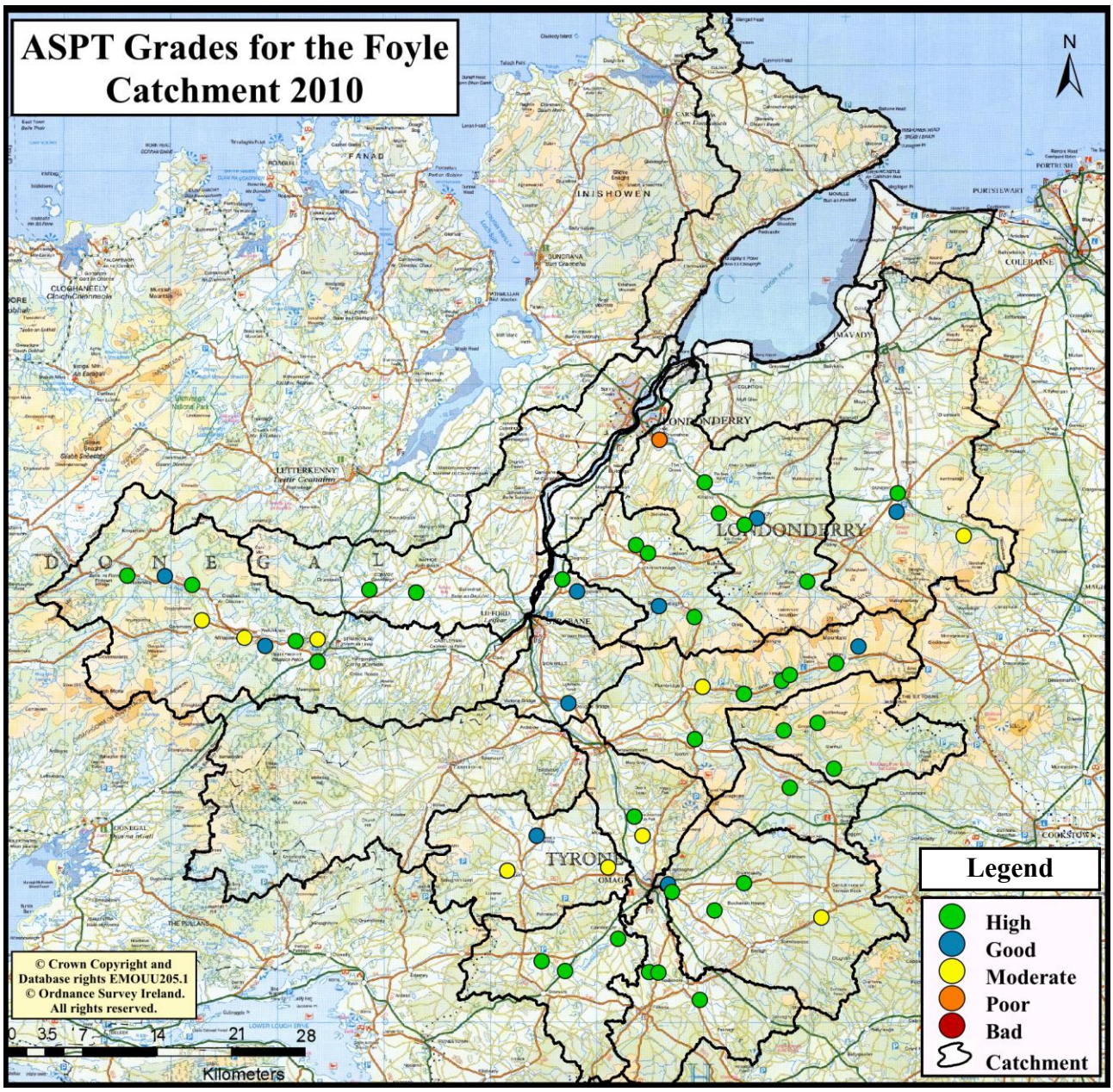

Figure 3.5 - The ASPT EQR grades for 54 sites (n=54) in the Foyle catchment, 2010 (Grades: bad - 0 sites; poor - 1 site; moderate - 9 sites; good - 10 sites; high - 34 sites).

The N-TAXA EQR grades also presented with changes between 2009 and 2010, however as previously mentioned the results here should be used with caution and will only be used for demonstration purposes. In 2009 there were 26 sites which were graded as bad whereas in 2010, the number of sites graded as bad increased to 35. The number of sites that were graded as poor remained constant however the number of sites with a moderate grade dropped in 2010 to 11 from a previously higher number, with 20 sites in 2009.

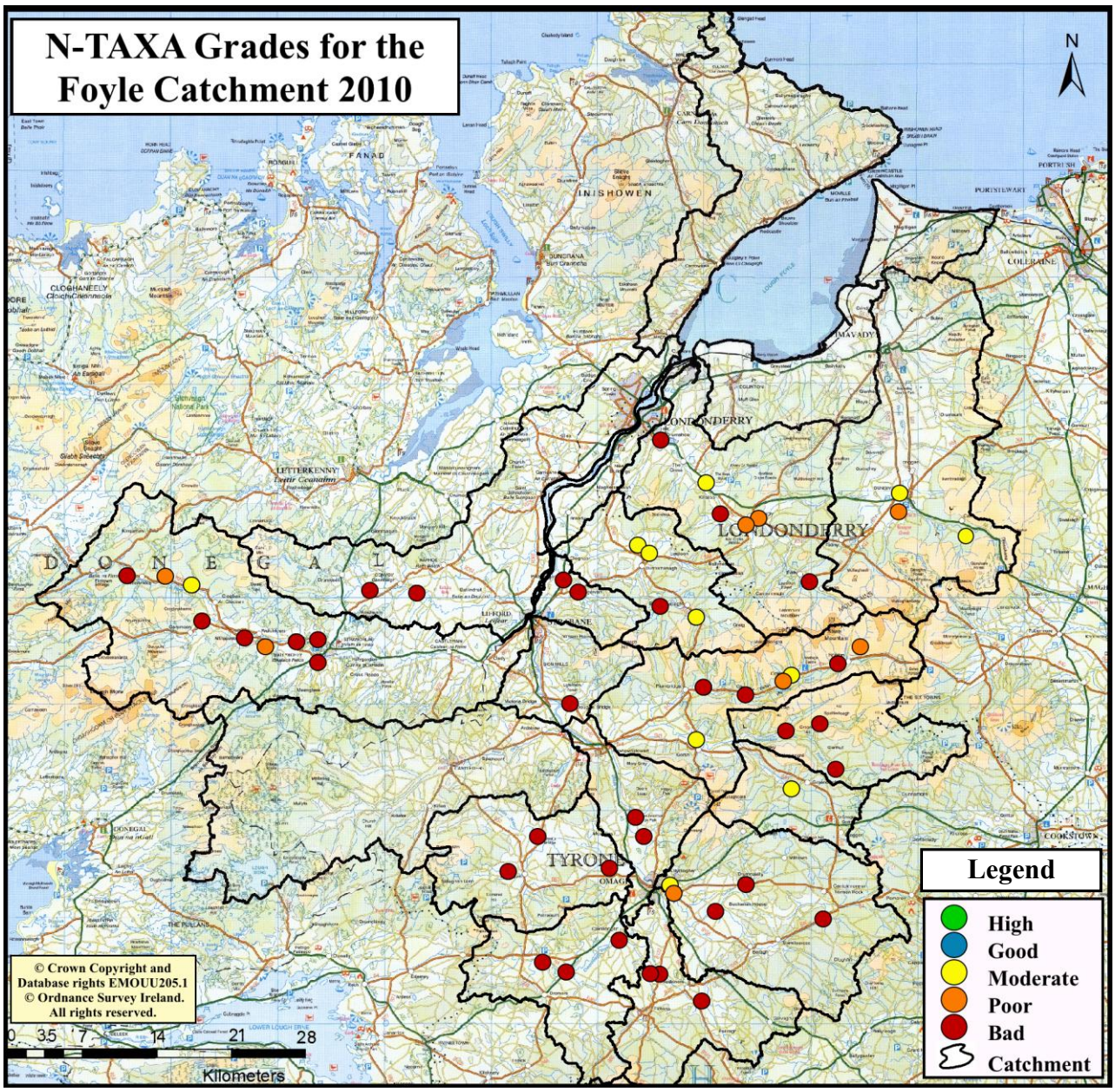

Figure 3.6 - The N-TAXA EQR grades for 54 sites (n=54) in the Foyle catchment, 2010 (Grades: bad - 35 sites; poor - 8 sites; moderate - 11 sites; good - 0 sites; high - 0 sites).

Again this only provides an overview of the total numbers and does not illustrate how individual sites change from year to year. Of the 54 sites, the grades of only 24 sites remained constant from 2009 to 2010 with the remaining either dropping or increasing the quality grade. However there was a tendency for the quality grade to drop from 2009 to 2010. For the Carlingford catchment in 2010, only two sites had the environmental and biological data required for RICT analysis (figure 3.7). These two sites differed in their ASPT EQR grades with one site graded as high and the other site graded as good.

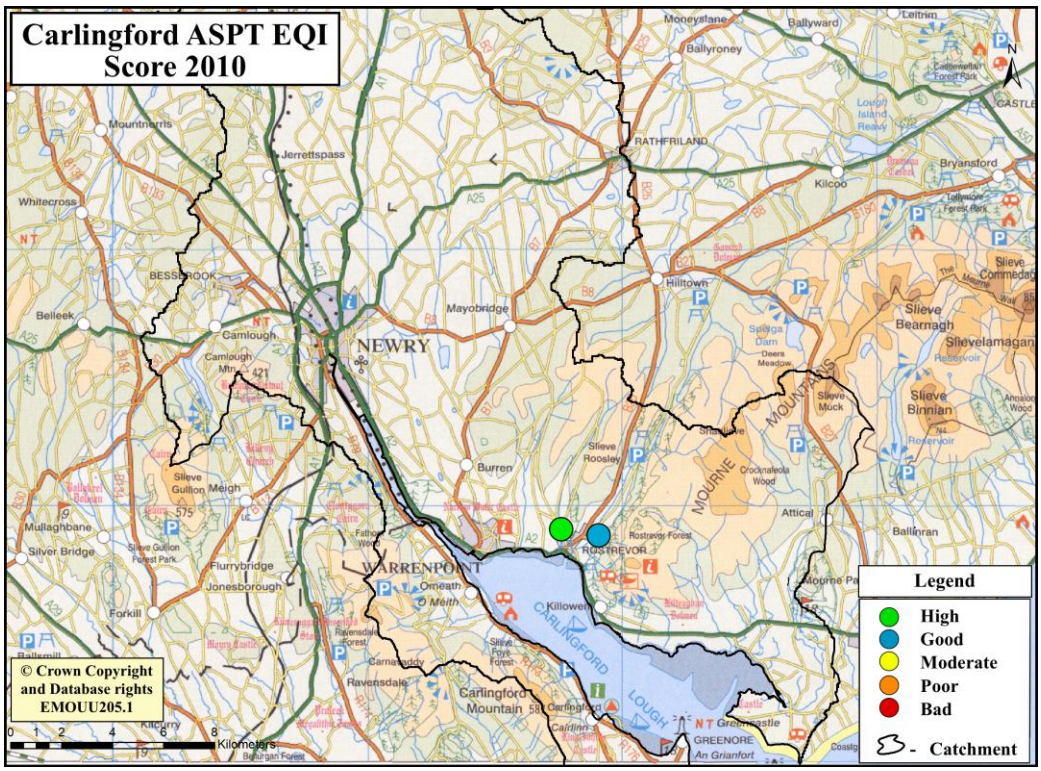

Figure 3.7 - The ASPT EQR grades for 2 sites (n=2) in the Carlingford catchment, 2010. (Grades: good - 1 site; high - 1 site).

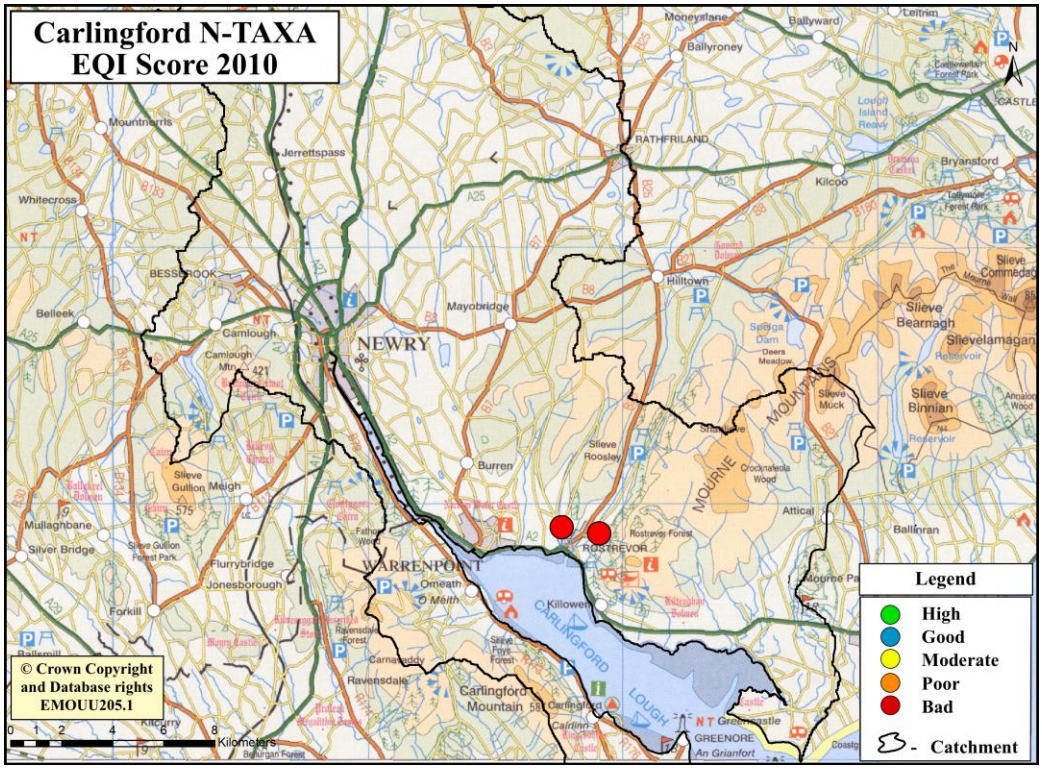

Figure 3.8 - The N-TAXA EQR grades for 2 sites (n=2) in the Carlingford catchment, 2010, (Grades: bad - 2 sites).

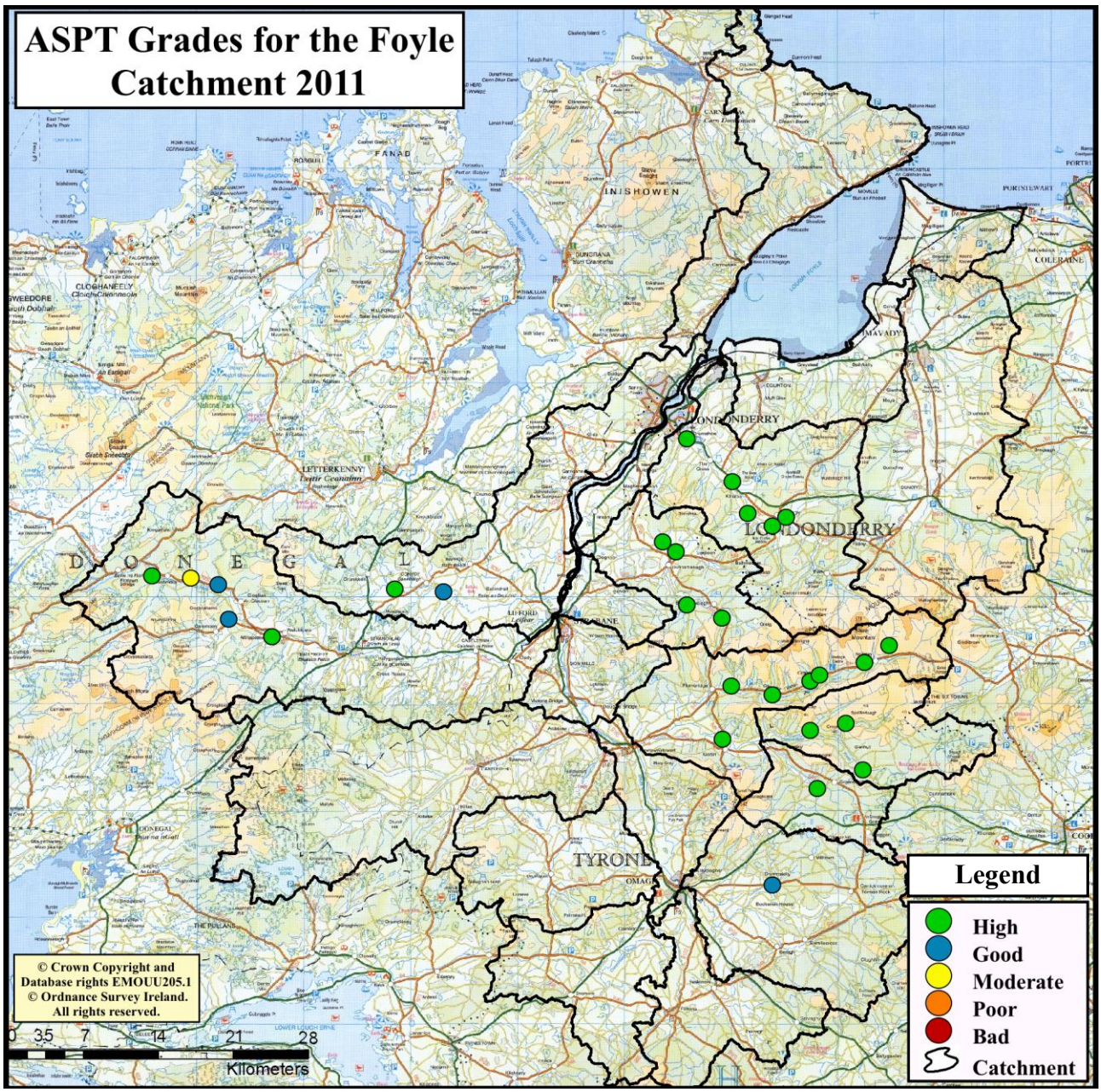

Figure 3.9 - The ASPT EQR grades for 28 sites (n=28) in the Foyle catchment, 2011.
(Grades: moderate - 1 site; good - 4 sites; high - 23 sites).

These results would indicate that the sites were experiencing very little to no organic pollution. The site with the ASPT EQR grade of good as previously stipulated, this site had likely been exposed to a small level of organic pollution, although not to the extent that would severely degrade a site. The 2010 N-TAXA EQR grades for the same sites in the Carlingford sites are displayed in figure 3.8. Of the two sites, both were graded as bad, which if the results were true representatives of quality would indicate that these sites were presenting with the lowest quality grade.

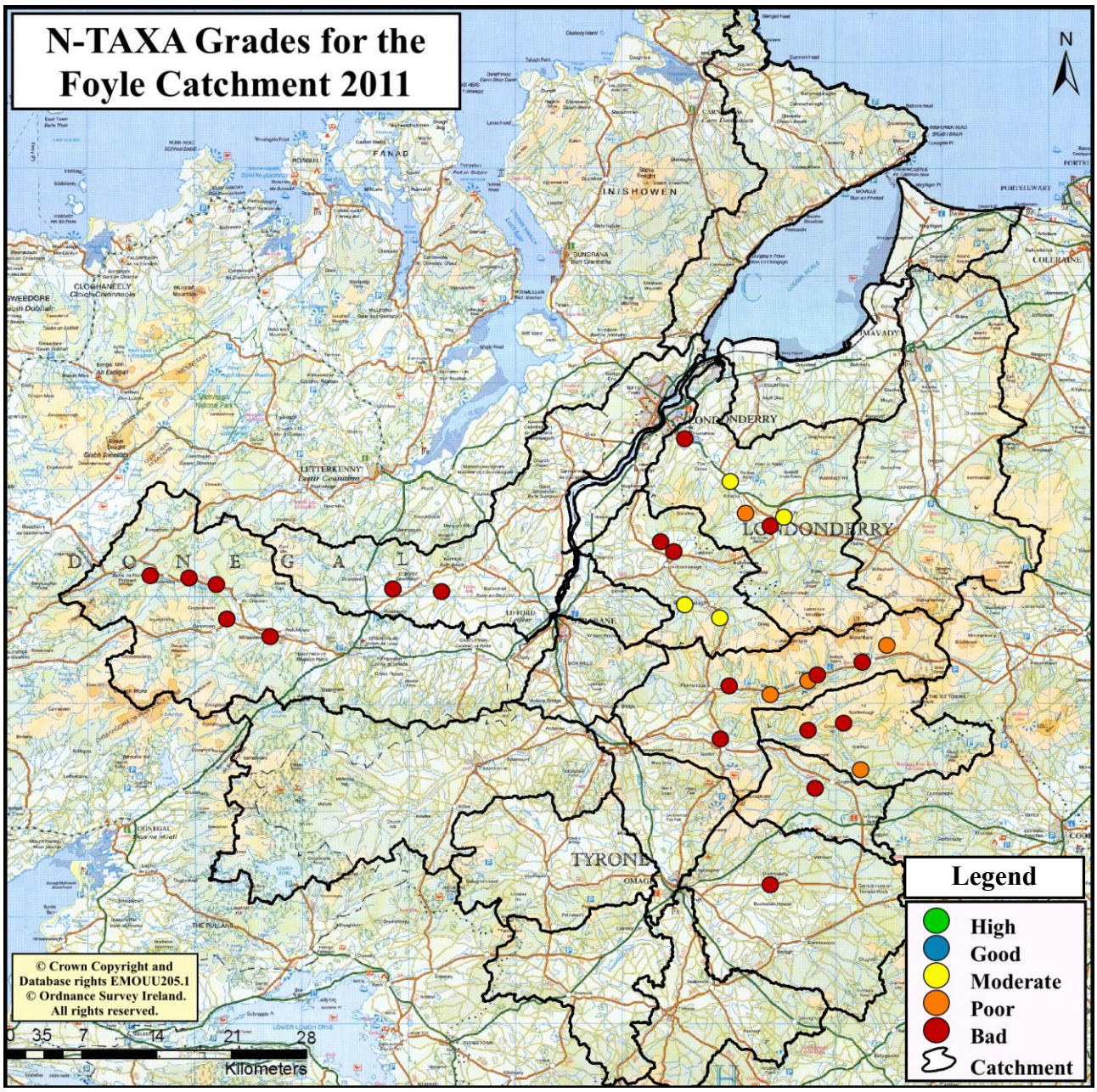

Figure 3.10 - The N-TAXA EQR grades for 28 sites (n=28) in the Foyle catchment, 2011. (Grades: bad - 19 sites; poor - 5 sites; moderate - 4 sites).

This would of suggested that the sites were exposed to a substantial amount of environmental disturbance or toxic pollution. From 2009 to 2010 the number of sites used in the Carlingford catchment with the RICT approach dropped, 5 sites were assessed in 2009 with only 2 in 2010. However, the 2 sites in 2010 were the same sites from 2009, where both the ASPT and N-TAXA EQR grades remained the same between the two years.

Finally in 2011, 23 sites were graded as a ASPT EQR grade of high, indicating a minimal level of exposure to organic pollution (figure 3.9). Of the remaining 5 sites analysed, 4 were graded as good with only 1 site

presenting with a low grade of moderate. This indicates that this one site graded as moderate had been exposed to some form of organic pollution. From the 28 sites sampled, the 19 sites presented with a N-TAXA EQR grade of bad (figure 3.10), 5 sites were graded as poor with the remaining 4 sites graded as moderate. Similar to the 2009 and 2010 datasets, the 2011 sites for the Foyle were poorly represented as indicated by the N-TAXA EQR grades. It may be concluded that the sites of the Foyle and Carlingford catchments were experiencing high levels of toxic pollution exposure and/or environmental degradation while experiencing minimal levels of organic pollution. The results from the RICT assessment would suggest there were no sites of Foyle catchment from 2009-2011 that achieved any higher than EQR grade of moderate, this would imply that every site had been substantially exposed to organic pollution and toxic pollution and/or degradation. As the N-TAXA grades presented with the lower grade at each site, it is likely that toxic pollution and/or environmental degradation was the prominent factor which was adversely affecting the quality of sites.

3.2 Semi-quantitative electro-fishing results

The following figures display the results from each year's semi-quantitative electrofishing surveys for the years 2009 to 2010, within the sub-catchments of the Foyle and Carlingford catchment area. The purpose of these results is to provide complimentary results to the RICT results for the same site with the salmon and trout grades also used in the linear regression analyses.

The 0+ salmon density for each site in the Foyle catchment for 2009 are shown in figure 3.11. As displayed, the majority of the sites differ within each sub-catchment of the Foyle except in the case of Faughan, Glenmoran, East Owenreagh and the Mourne sub-catchments.

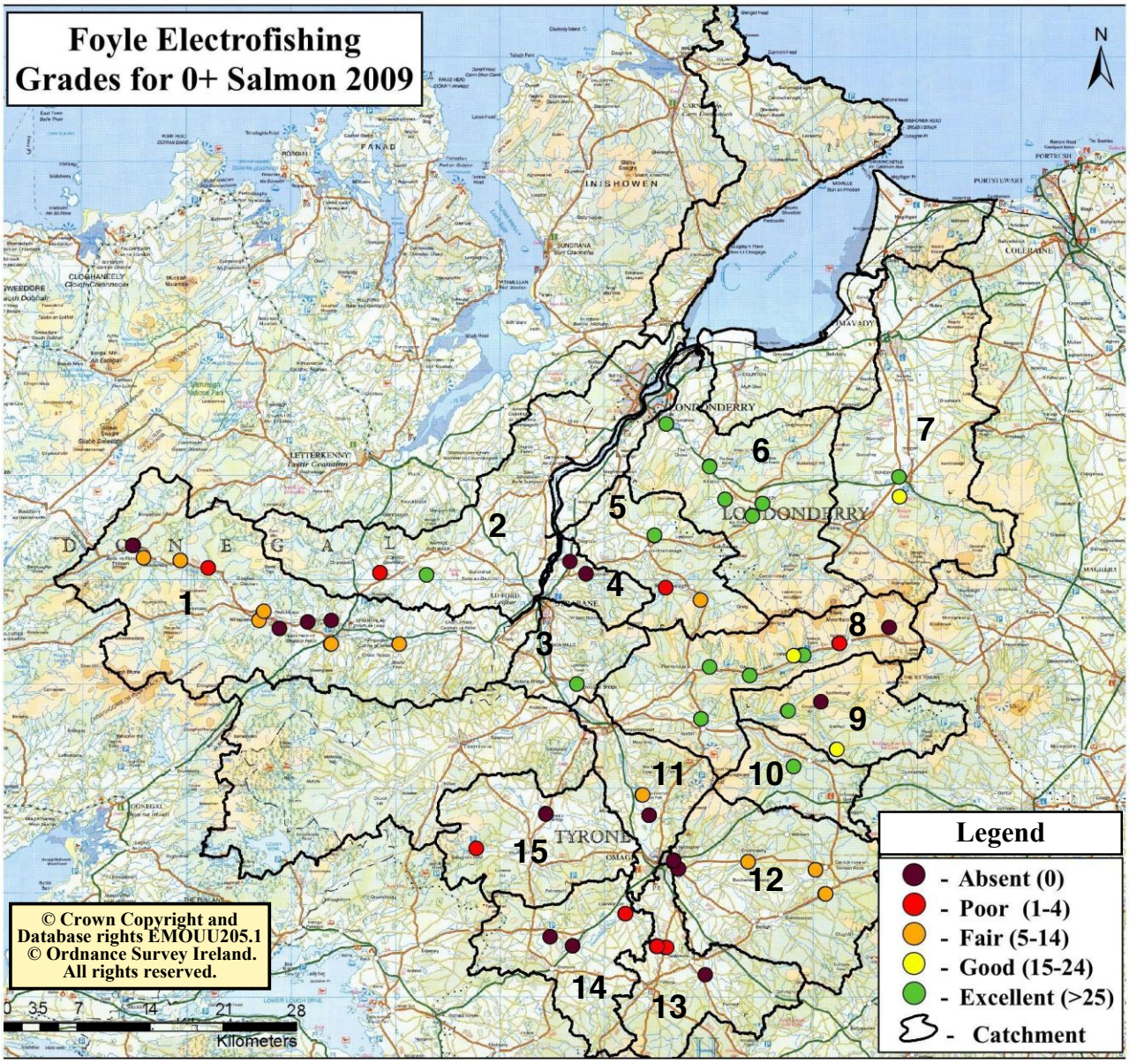

Figure 3.11 - The 0+ salmon semi-quantitative grades for 52 sites (n=52) in the Foyle catchment, 2009. (Grades: absent - 15 sites; poor - 8 sites; fair - 11 sites; good - 3 sites; excellent - 15 sites).
1) Finn. 2) Deele. 3) Mourne. 4) Glenmornan. 5) Burn Dennet. 6) Faughan. 7) Roe. 8) Glenelly. 9) Owenkillew 10) Owenreagh East. 11) Strule. 12) Camowen. 13) Drumragh. 14) Owenreagh South. 15) Fairywater.

The combination of density grades for sub-catchment differed from the next. Of the 52 sites within the Foyle catchment 15 sites were given a salmon density grade of excellent, of which 5 were found in the Faughan catchment. Only 3 sites presented with a density grade of good which were sporadically positioned across the Foyle. The remaining sites of the Foyle displayed with 11 sites given a density grade of fair, 8 sites with a density grade of poor and 15 sites indicating the absence of any juvenile salmon.

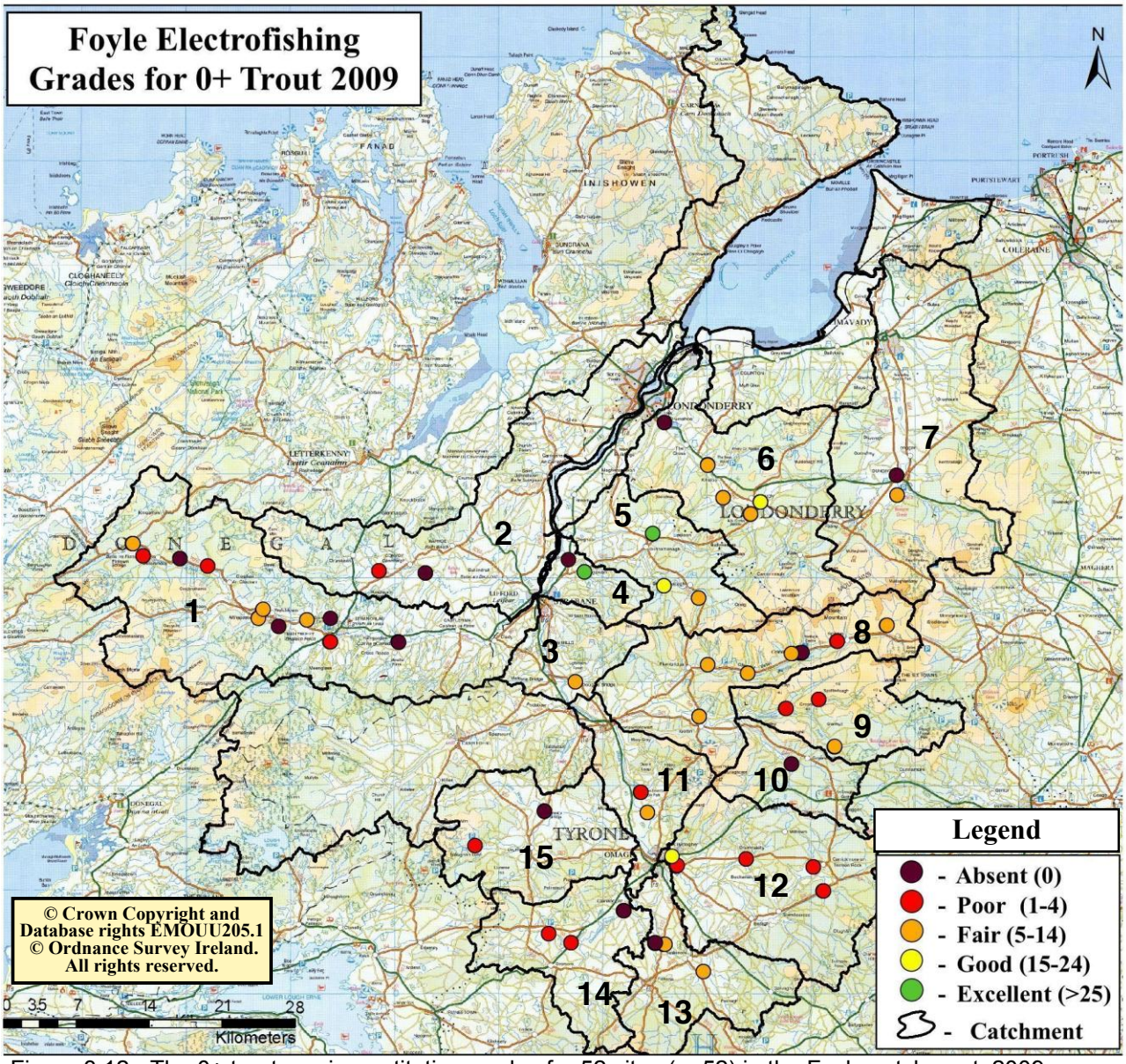

Figure 3.12 - The 0+ trout semi-quantitative grades for 52 sites (n=52) in the Foyle catchment, 2009 (Grades: absent - 13 sites; poor - 15 sites; fair - 19 sites; good - 3 sites; excellent - 2 sites).
1) Finn. 2) Deele. 3) Mourne. 4) Glenmornan. 5) Burn Dennet. 6) Faughan. 7) Roe. 8) Glenelly. 9) Owenkillew. 10) Owenreagh East. 11) Strule. 12) Camowen. 13) Drumragh. 14) Owenreagh South. 15) Fairywater.

The sites belonging to the cluster of sub-catchments in the south of the Foyle catchment along with the Finn and Glenmornan sub-catchments presented with the poorest density readings. The 0+ trout density grades of the Foyle catchment for 2009 are shown in figure 3.12. Of these sites, only 2 sites were graded as excellent and 3 sites were graded as good again they were sporadically positioned across the Foyle. Of the remaining sites 19 were graded as fair, 15 sites were graded as poor and 13 sites were absent of any juvenile trout.

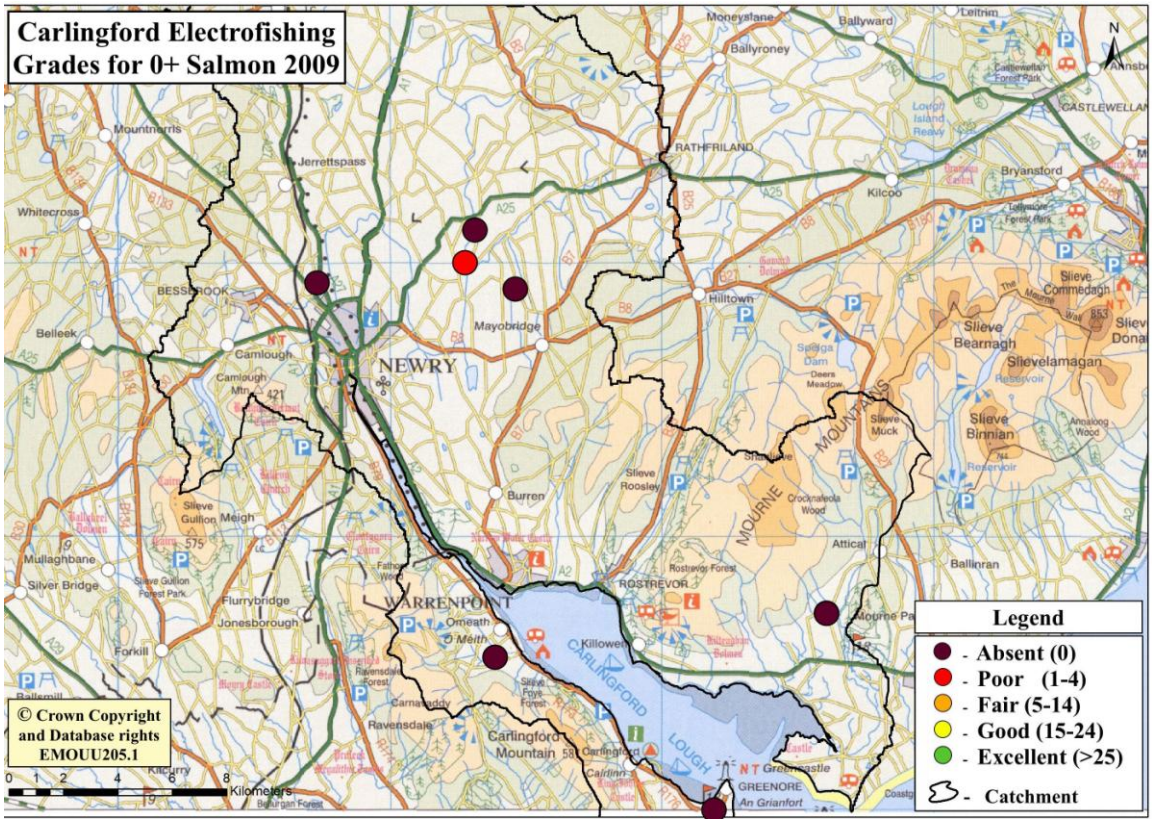

Figure 3.13 - The 0+ salmon semi-quantitative grades for 7 sites (n=7) in the Carlingford catchment, 2009. (Grades: absent - 6 sites; poor - 1 site).

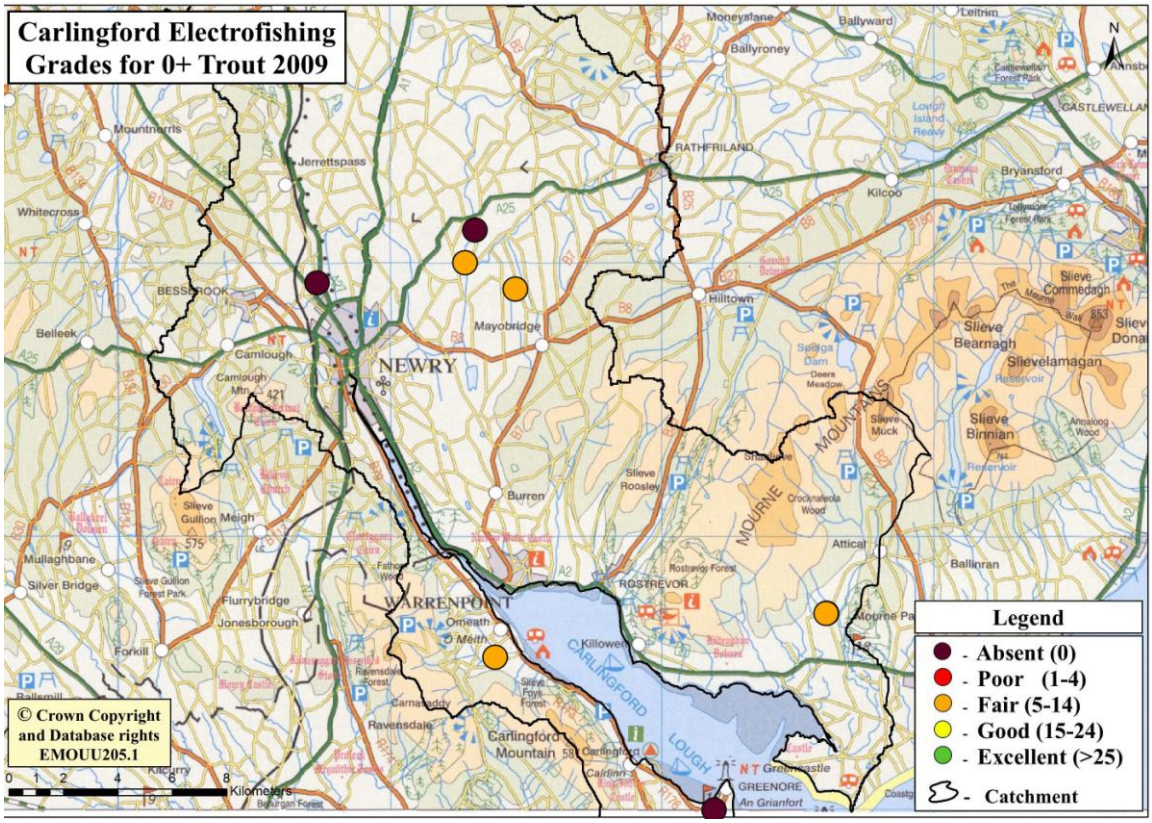

Figure 3.14 - The 0+ trout semi-quantitative grades for 7 sites (n=7) in the Carlingford catchment, 2009. (Grades: absent - 3 sites; fair - 4 sites).

The individual site in the Owenreagh East sub-catchment recorded no 0+ trout and was graded as absent. The Mourne sub-catchment also only had 1 site representing it, which was graded as fair.
These two sites along with the 45 other sites graded as fair, poor and absent indicates that in 2009 all these sites were underproducing with regards to juvenile trout.

The 2009 0+ salmon density grades of the Carlingford catchment are displayed in figure 3.13. Only 1 site out of the 7 sites recorded the presence of juvenile salmon however that site had a poor density grade and indicates that this site was seriously underproducing. The 2009 0+ trout density grades for the Carlingford catchment are displayed in figure 3.14. From these 7 sites none presented with a density grade of excellent or good, with 4 sites graded as fair and 3 sites having no record of trout capture. This indicates that of the 7 sites sampled, all were underproducing with respect to juvenile trout.

The 2010 0+ salmon grades for the Foyle are displayed in figure 3.15, for this year there were 50 sites which acted as indicators of the Foyle's quality of juvenile densities. From these, only 15 sites presented with high densities and a density grade of excellent. Of the sites graded as excellent, 7 of these sites were found within the sub-catchments of the Faughan, Roe and Owenreagh East. The rest of the sites, apart from the two sites graded as good, were thought to be underproducing. There were 10 sites were the presence of juvenile salmon was not recorded. The Foyle's 0+ salmon grades also saw changes between the two years, with only 28 sites maintaining the same quality grade from 2009 to 2010. All the sites sampled within the Faughan sub-catchment were consistent between the two years with all sites indicating excellent densities.

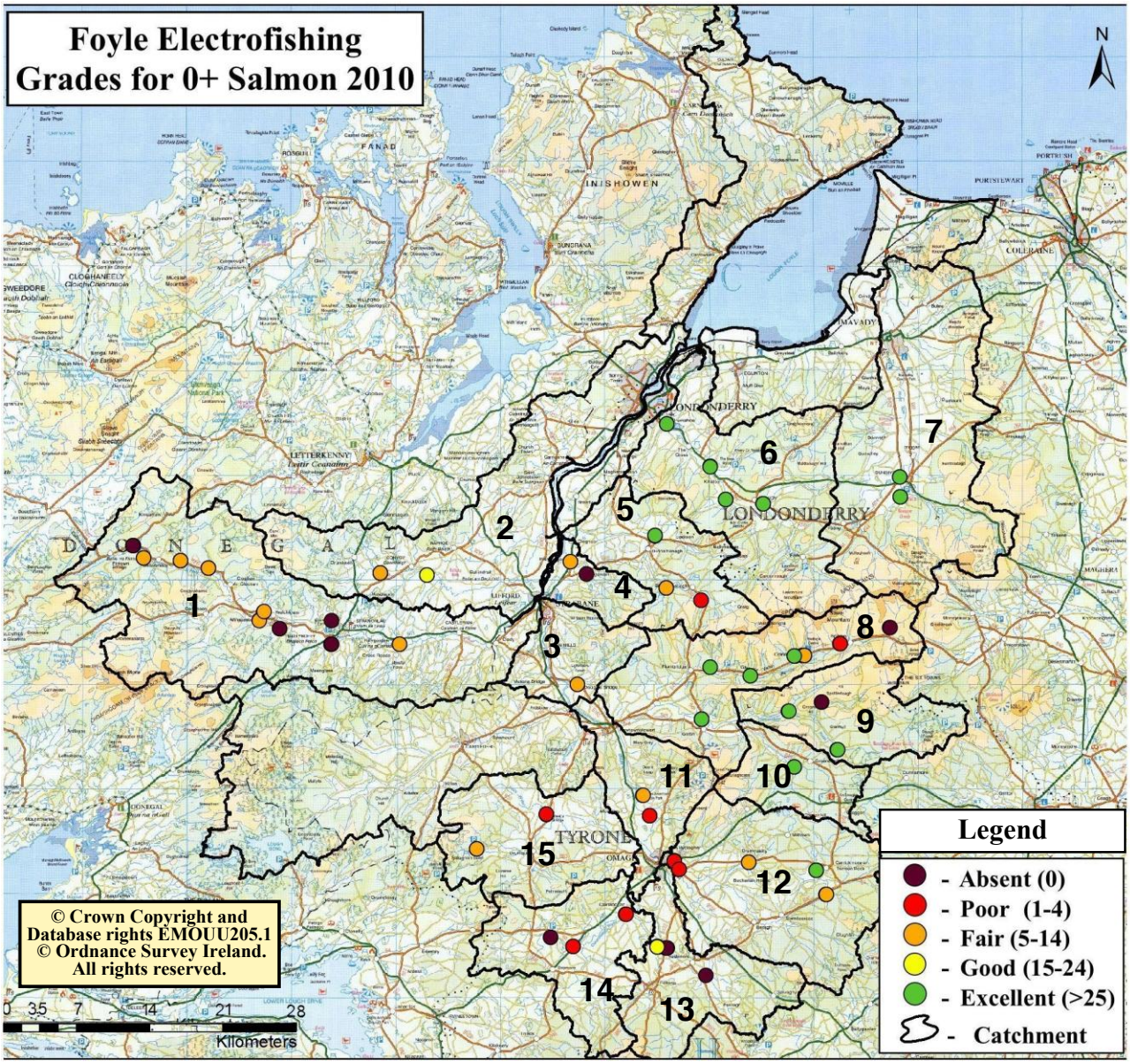

Figure 3.15 - The 0+ salmon semi-quantitative grades for 50 sites (n=50) in the Foyle catchment, 2010. (Grades: absent - 10 sites; poor - 8 sites; fair - 15 sites; good - 2 sites; excellent - 15 sites).
1) Finn. 2) Deele. 3) Mourne. 4) Glenmornan. 5) Burn Dennet. 6) Faughan. 7) Roe. 8) Glenelly. 9) Owenkillew. 10) Owenreagh East. 11) Strule. 12) Camowen. 13) Drumragh. 14) Owenreagh South. 15) Fairywater.

The 2010 0+ trout density data for the Foyle catchment are graded and displayed in figure 3.16. Of the 50 sites only 2 sites presented with optimal densities, a density grade of excellent, with 2 sites presenting with a grade of good. The remaining sites indicated underproducing juvenile densities with 8 sites absent of any juvenile trout. All the sites sampled in the sub-catchments Mourne, Glenmornan and Strule were presented with density grades of fair. However as each sub-catchment is only represented by a relatively few sites, it can only be stipulated that the specific river stretches sampled are underproducing and not the entire sub-catchment.

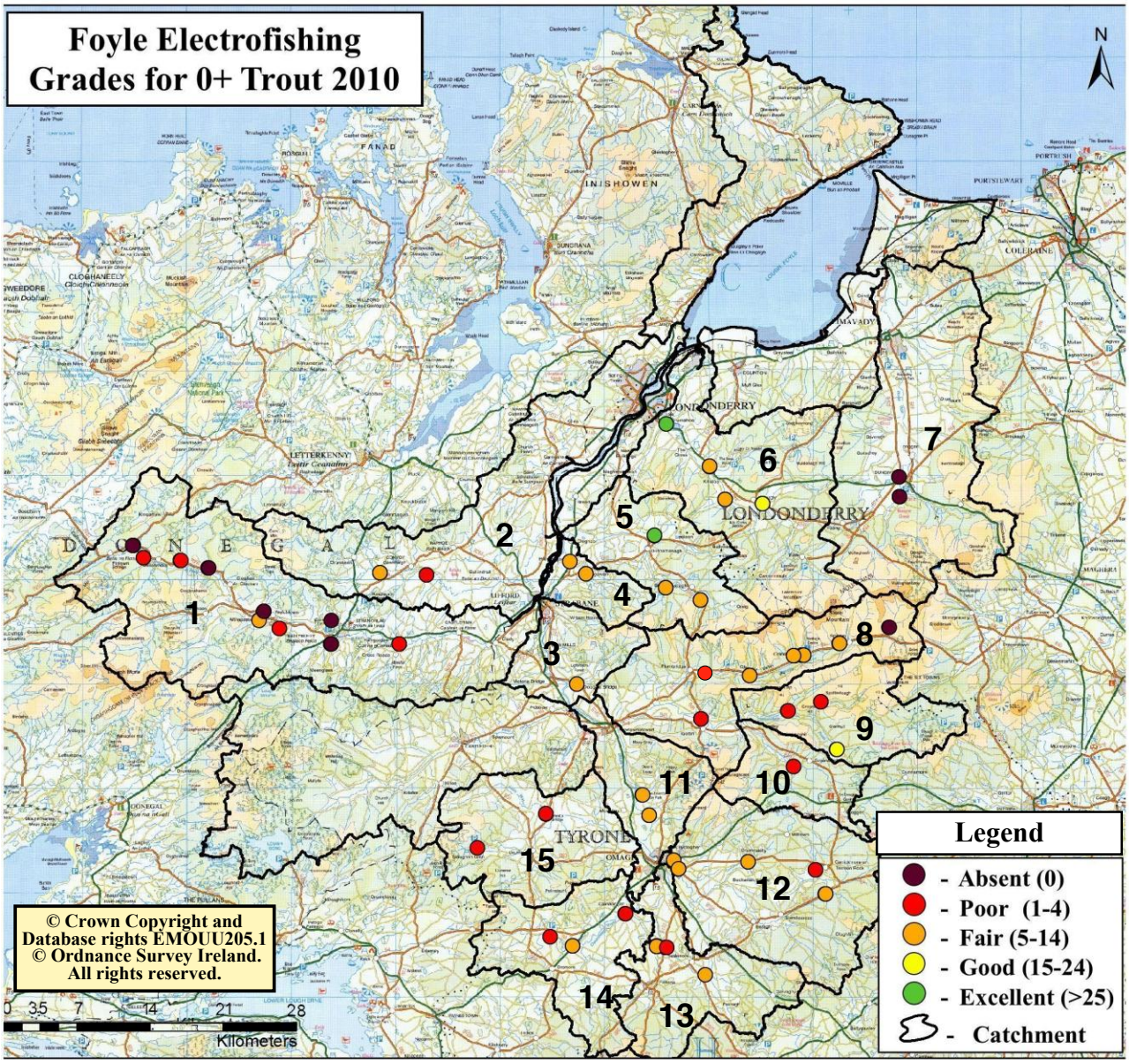

Figure 3.16 - The 0+ trout semi-quantitative grades for 50 sites (n=50) in the Foyle catchment, 2010. (Grades: absent - 8 sites; poor - 16 sites; fair - 22 sites; good - 2 sites; excellent - 2 sites). 1) Finn. 2) Deele. 3) Mourne. 4) Glenmornan. 5) Burn Dennet. 6) Faughan. 7) Roe. 8) Glenelly. 9) Owenkillew. 10) Owenreagh East. 11) Strule. 12) Camowen. 13) Drumragh. 14) Owenreagh South. 15) Fairywater.

For the Foyle catchment between 2009 and 2011 only 19 sites presented with the same trout density grades. However there was a tendency for density grades to improve from 2009 to 2010, with 23 sites exhibiting an improvement. That said, 8 sites did display with a drop in the density recorded. In 2010, the 6 sites of the Carlingford catchment are displayed on figure 3.17. Of the 6 sites, 2 were graded as fair, 1 was graded as poor with the remaining 3 sites indicating the absence of any juvenile salmon.

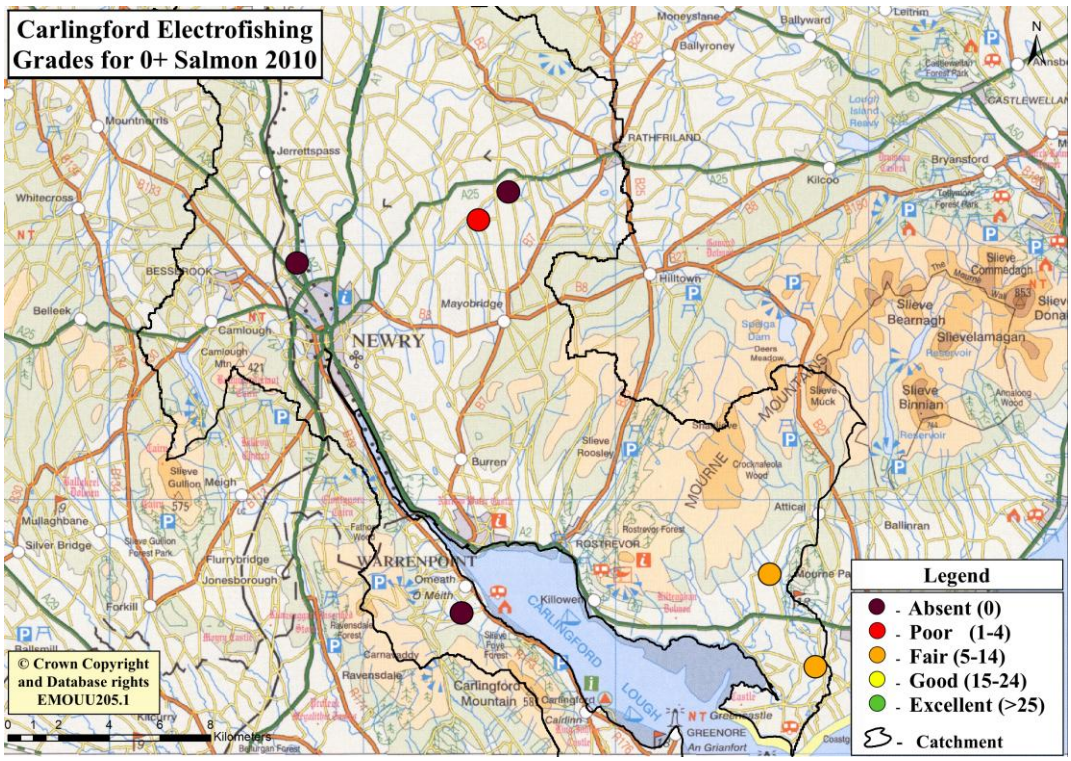

Figure 3.17 - The 0+ salmon semi-quantitative grades for 6 sites (n=6) in the Carlingford catchment, 2010. (Grades: absent - 3 sites; poor - 1 site; fair - 2 sites).

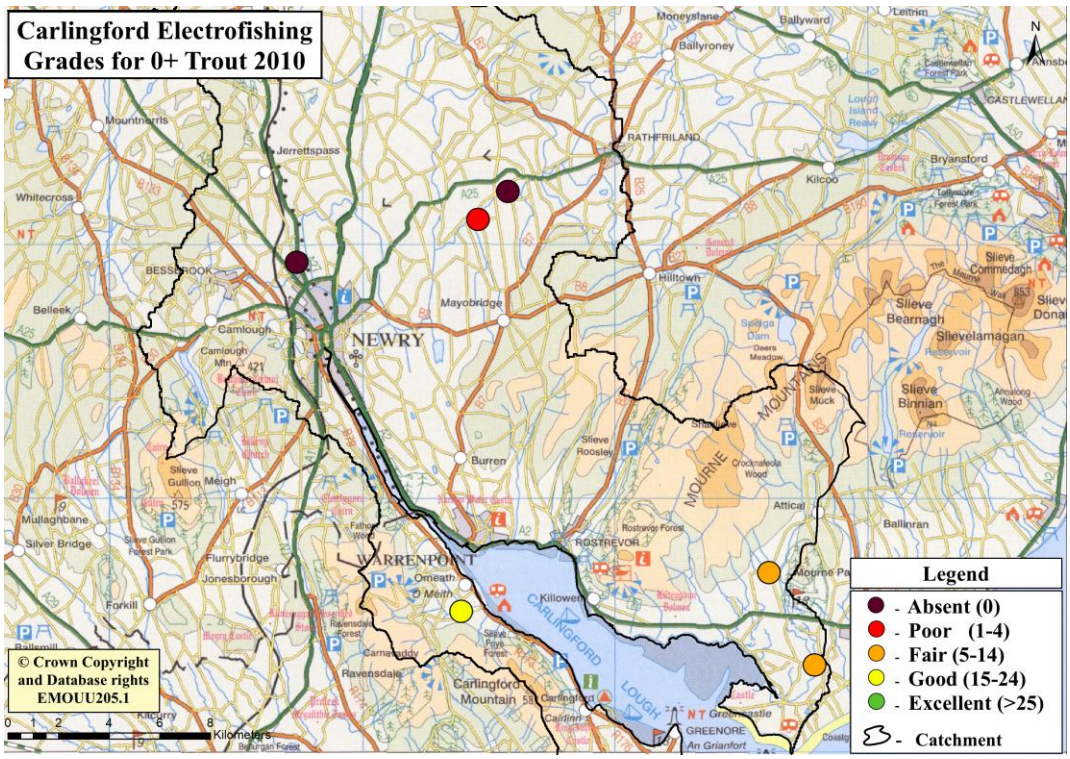

Figure 3.18 - The 0+trout semi-quantitative grades for 6 sites (n=6) in the Carlingford catchment, 2010. (Grades: absent - 2 sites; poor - 1 site; fair - 2 sites; good - 1 site).

There were 3 sites that overlapped between 2009 and 2010, the grades of 2 sites remained the same, while the other site changed from a grade indicating the absence of juvenile salmon to a grade of fair in 2010. This

indicates that all these sites are under some form of pressure. The 2010 trout sites for Carlingford catchment has 7 sites which range from a grade of absent to good. This is displayed in figure 3.18, where there was only one site graded as good with no sites graded as excellent. The remaining 6 sites presented with 2 sites absent of any 0+ trout and 4 sites indicating underproduction. From 2009 to 2010 only 3 sites were positioned in the same location within the Carlingford catchment. Only one improved in its quality grade, from a density grade of fair to good. The other 2 sites density grades remained the same for the years 2009 and 2010. It is clear from these results that many juvenile salmon and trout sites are not presenting with high densities and thus there must be some contributing factor that is acting on the river stretches and subsequently on the juvenile fish populations. This data gathered across both catchments, demonstrates that there was a substantial number of sites that indicated underproduction or a total absence of 0+ salmon and/or trout.

3.3 Linear regression analysis of biological indices against fish density grades

The linear regression analysis conducted of the N-TAXA score against trout grade for 2009, figure 3 19, revealed that the relationship between the two was not significant (P = 0.393), with only a very small proportion of the variation in the trout grade can be explained by changes in N-TAXA scores (*r-sq= 0.0159*), further indicating a lack of correlation between the two. For 2009, a change in the N-TAXA score did not correlate with a change in the trout semi-quantitative grade at the same site.

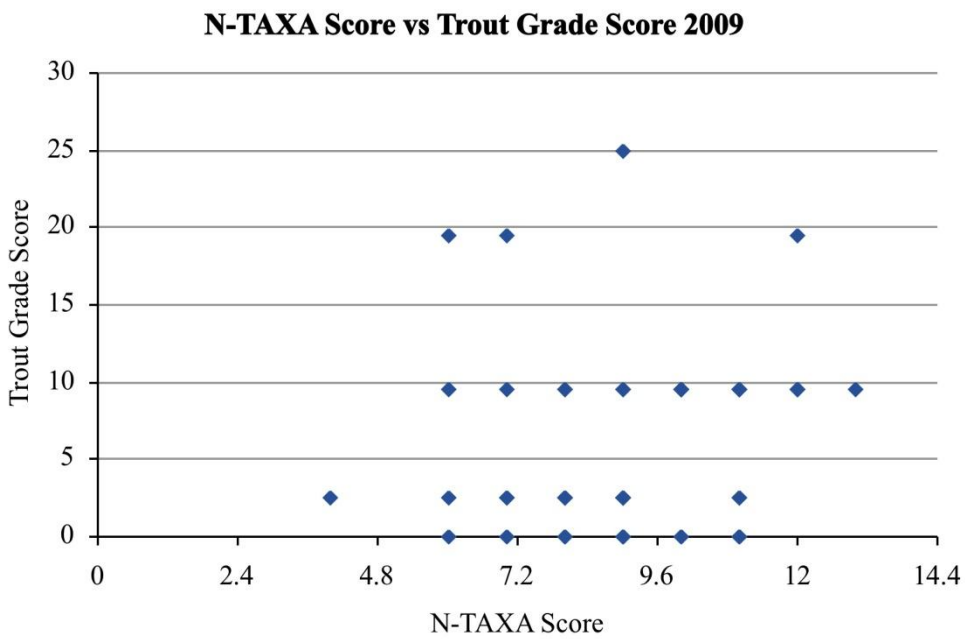

Figure 3.19 - Linear regression of trout density grade score plotted against N-TAXA score for 2009. (n=49; P-value = 0.393).

The regression analysis of the two (figure3. 20) revealed that the relationship between the ASPT scores and the trout density grades was not significant (*P=0.682*). It was also concluded that changes in the ASPT score did not correlate to changes in the trout density grades (*r-sq = 0.0037*). From both regression analyses it was found that neither the ASPT nor N-TAXA index scores showed a significant relationship with the 2009 trout grades.

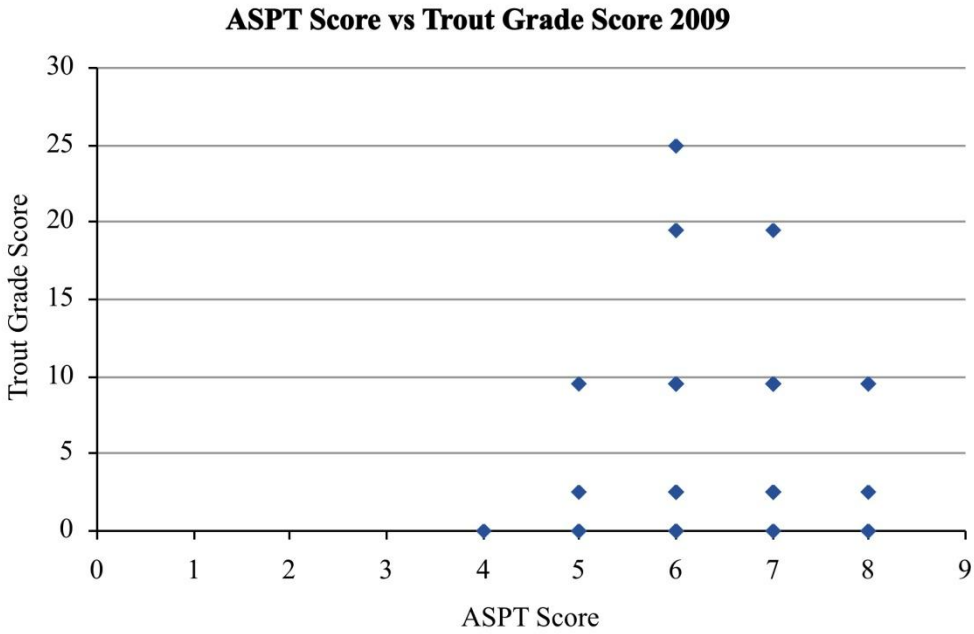

Figure 3.20 - Linear regression of trout density grade score plotted against ASPT score for 2009. (n=49; P-value = 0.682).

The biological index scores of 2009 were also assessed against the 2009 salmon grades. The linear regression of the N-TAXA scores against salmon grades (figure 3.21) indicated that the relationship between the two was not significant (*P=0.360*). Furthermore, the lack of variation in the salmon scores (*r-sq = 0.0179*) associated with changes in N-TAXA scores, indicated little correlation between the two. Similarly, the relationship between the 2009 ASPT scores and salmon density grades for the same year (figure 3.22), was found to be not significant (*P=0.325*). This was further emphasised by the lack of variation explained by changes in the ASPT score (*R-sq=0.0206*).

Again the lack of significance between the biological indices and the salmon grades gathered in 2009 indicates that there is no relationship between the two that could be measured using the current parameters. For 2010, the same approach was used to assess the relationships between the biological indices with salmon and trout grades. Using the linear regression analysis, it was concluded that there was no indication of a significant relationship between the N-TAXA scores and the trout density grades (*P=0.398*) as displayed in figure 3.23. The regression reveals that the trout density grade variation is not correlated to variations in the N-TAXA scores (*r-sq=0.0163*). The linear regression of the 2010 ASPT scores against trout grades

(figure 3.24), also failed to reveal a significant relationship (*P=0.341*). As with the 2009 analyses, the variation of the 2010 trout grades does not correlate with changes in the ASPT score (*r-sq=0.0206*).

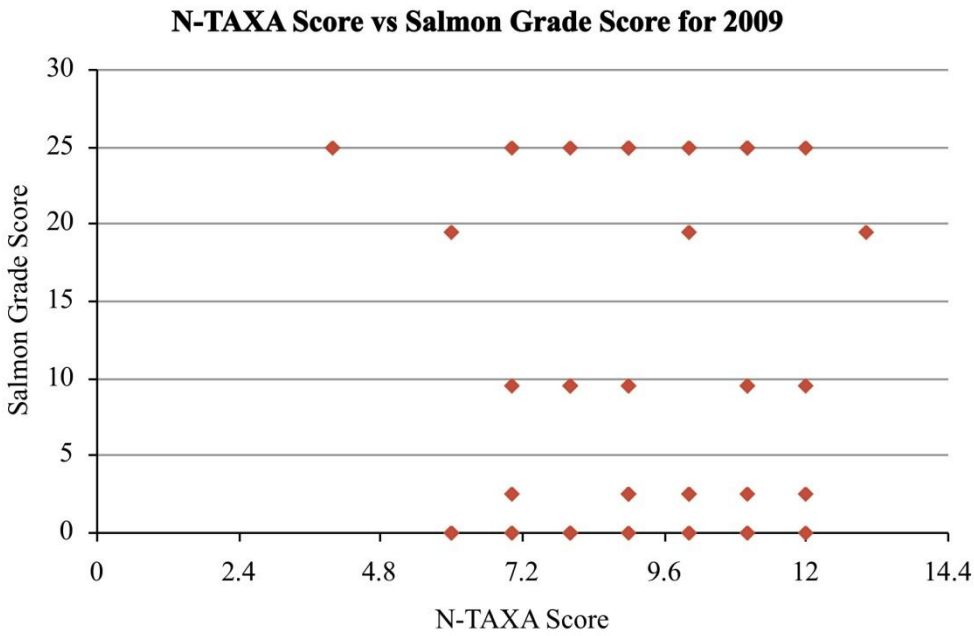

Figure 3.21 - Linear regression of salmon density grade score plotted against N-TAXA score for 2009. (n=49; P-value = 0.360).

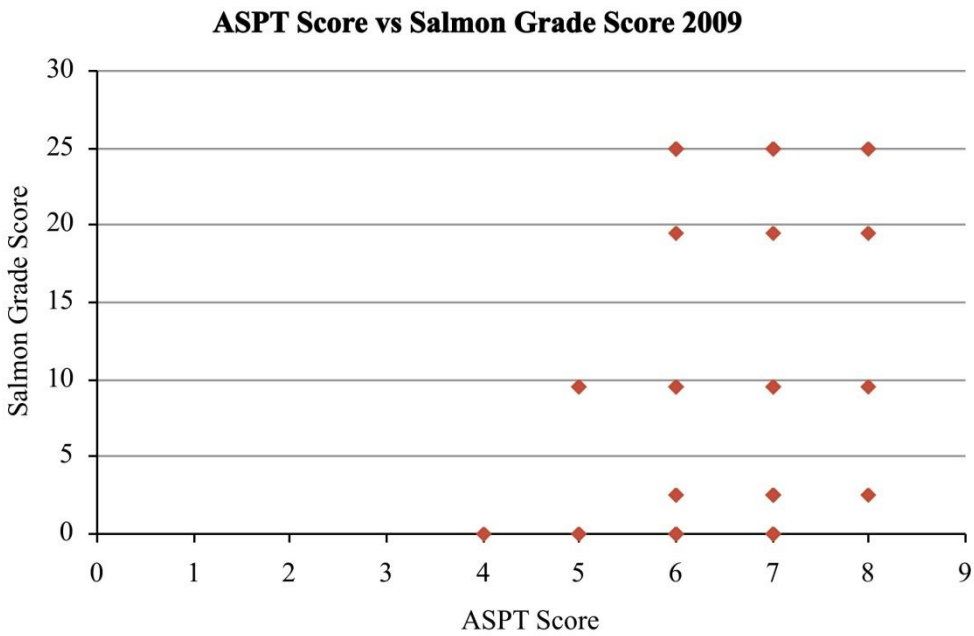

Figure 3.22 - Linear regression of salmon density grade score plotted against ASPT score for 2009. (n=49; P-value = 0.325).

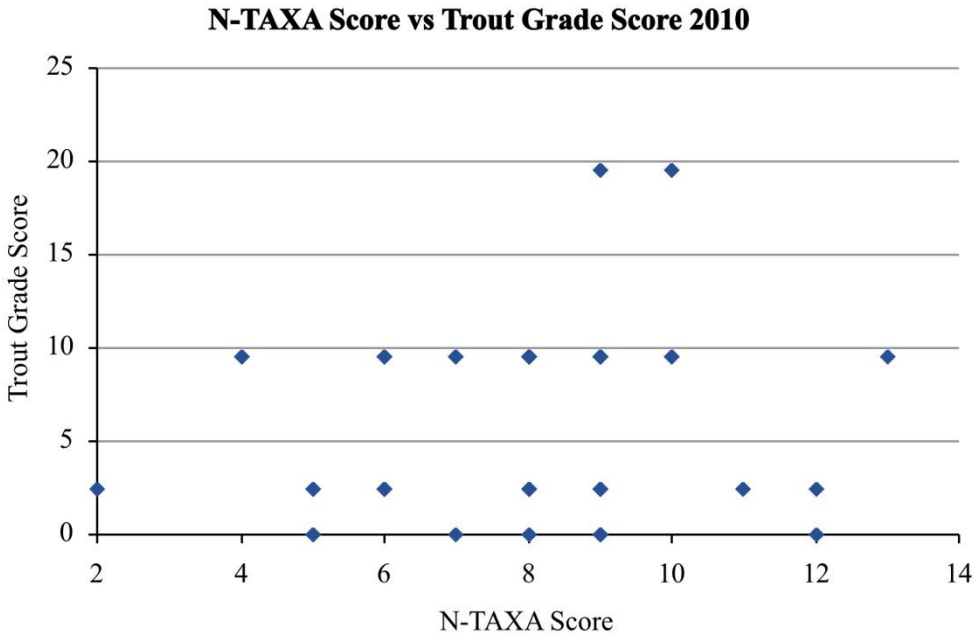

Figure 3.23 - Linear regression of trout density grade score plotted against N-TAXA score for 2010.(n=46; P-value = 0.398).

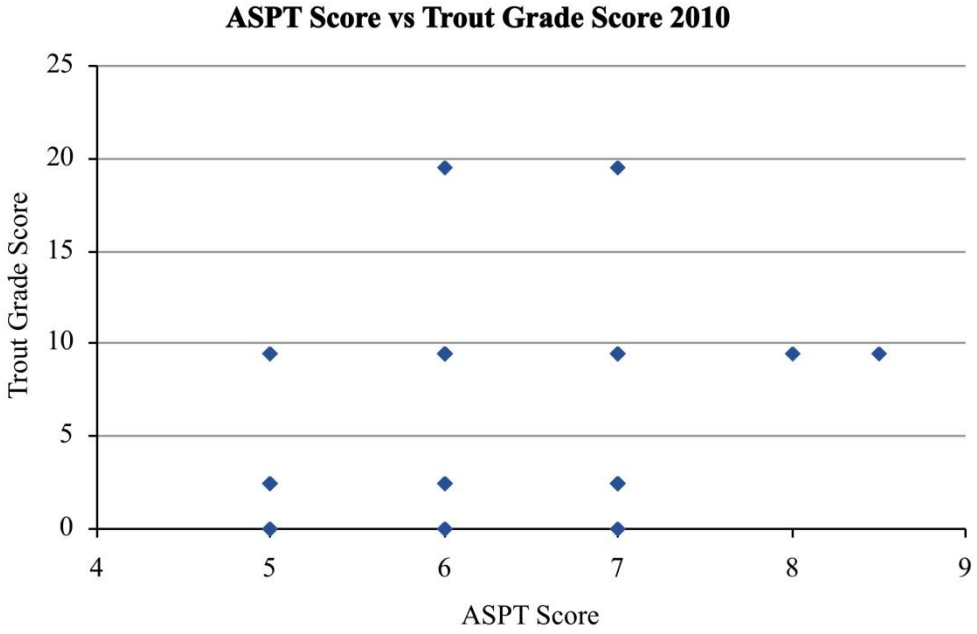

Figure 3.24 - Linear regression of trout density grade score plotted against ASPT score for 2010. (n=46; P-value = 0.341).

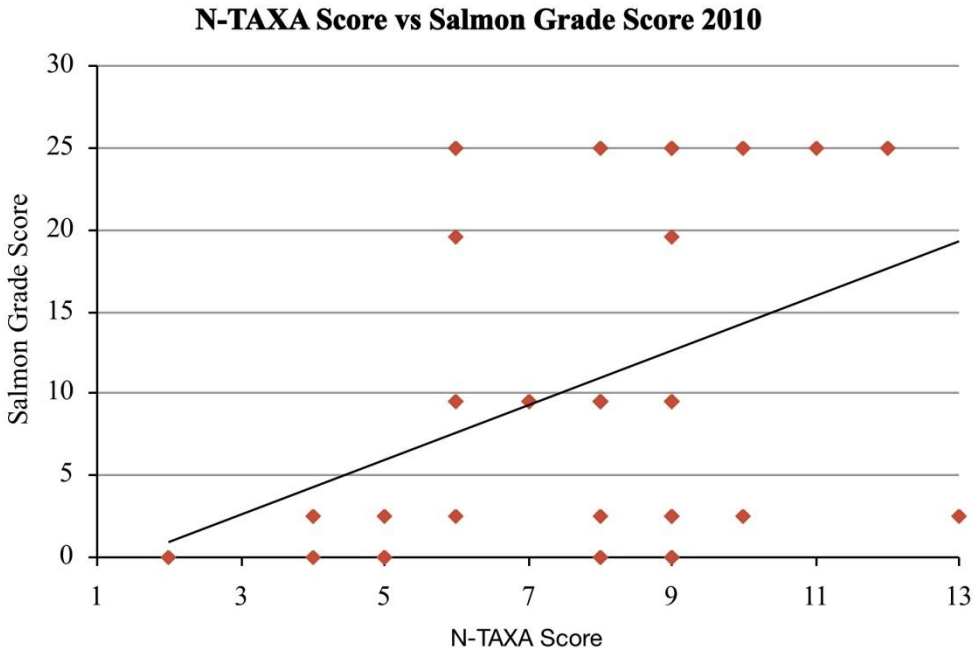

Figure 3.25 - Linear regression of salmon density grade score plotted against N-TAXA score for 2010. (n=46; P-value = 0.01).

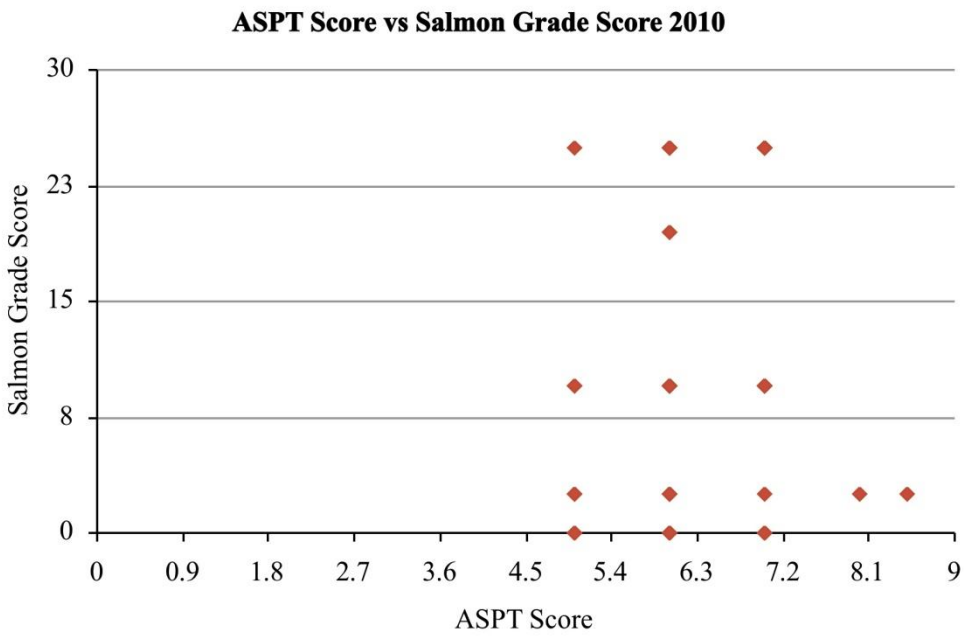

Figure 3.26 - Linear regression of salmon density grade score plotted against ASPT score for 2010. (n=46; P-value = 0.330).

Finally, the 2010 N-TAXA and ASPT scores were assessed against the 2010 0+ salmon grades. The linear regression of the N-TAXA scores against salmon grades revealed a significant positive relationship (*P=0.010*). From the linear regression of the N-TAXA score and the salmon grade (figure 3.25), 14% of the

salmon grade variability was correlated with changes in the N-TAXA score. However the regression trend-line does not fit to the data which illustrates the weak prediction power of the N-TAXA scores. Where the N-TAXA score showed a significant relationship with the 2010 salmon grades, the 2010 ASPT scores against the salmon grades did not reveal a significant relationship (*P=0.330)*, (figure 3.26). Furthermore the low r-squared value (*r-sq= 0.0216*) indicates that the ASPT score is not a strong prediction power. From the results of the linear regression analysis, excluding the significant relationship between the N-TAXA score and salmon grades for 2010, implies that with changes in the ASPT and N-TAXS scores does not correlate with predictable changes in the salmon and trout density grades.

4 DISCUSSION

This study, despite the implications associated with the methodology, illustrates that the RICT approach can produce quality grades from the macroinvertebrate samples collected across the Foyle and Carlingford catchments in 2009-2011. The grades of each site should however be taken with caution, as the N-TAXA grades are under-representing the 'true' ecological quality of each site. This may be contributed to the sampling procedure used to assess community richness. The ASPT which is a function of the N-TAXA grades and the total BMWP score, should also to a degree be regarded with caution.

The ASPT EQR quality grades produced for the Foyle catchment in 2009 would suggest that with respect to organic pollution the majority of sites are experiencing very little exposure. However, as 11 sites were graded as good and 8 sites graded as moderate it is clear that these sites have deviated from reference conditions due to exposure to organic pollution. The N-TAXA results for the Foyle catchment in 2009 if taken as representing the 'true' quality would provide an alternative view on the ecological quality of each site. Of all the sites sampled, 48% of the sites (n=26) presented with N-TAXA grade of bad. This profound deviation from reference conditions would suggest that with respect to toxic pollution and/or degradation these sites are being substantially impacted on. The grades of the remaining sites were divided between poor N-TAXA grades and moderate N-TAXA grades being the highest of this year. As both indices are indicative of type-specific stressors, the ASPT and N-TAXA grades for the Foyle in 2009 suggests that ecological quality of all the sites were primarily being degraded by exposure to environmental degradation or levels of toxic pollution if the N-TAXA grades were to be taken without consideration to the implications that the methodology has had on the results. However at some sites the quality was also being degraded by mild exposure to organic pollution.

The grades from the 2010 Foyle catchment exhibited similar results, where the ASPT grades indicates that 81% (n=44) of the sites sampled were experiencing minimal or no exposure to organic pollution. The remaining sites presented a moderate ASPT grade which suggests that the deviation of these sites from

reference conditions was due to mild exposure to organic pollution. As noted in the results, the ASPT grades of 34 sites remained static between 2009 and 2010 which suggests that the level of organic pollution exposure to these sites was constant between the two years.

The 2010 N-TAXA grades for the Foyle catchment, as in 2009 under normal circumstances reflect that sites are experiencing high exposure to toxic pollution and/or habitat degradation. Again there were no N-TAXA grades that were higher than moderate suggesting that all sites are experiencing substantial exposure to an individual or a combination of stressors. There were only 24 sites where the grade remained constant between the two years, similarly to the ASPT grades this may indicate that the exposure level between years remained constant. It is possible that the unchanged N-TAXA grades between years was due to the continual presence of environmental degradation within the vicinity of sites.

Finally 28 sites in the Foyle catchment were assessed for their ecological quality grades in 2011. Of these sites, 23 exhibited with an ASPT grade of high which is an improvement in the quality of these sites from 2009 and 2010. In the previous years, these 23 sites were between high and poor quality grades however in 2011 all were graded as high quality sites. This suggests that either the sources of the organic pollution has been removed or that in the previous years the deteriorated quality was caused by a sporadic pollution event and the sites had recovered in 2011. The N-TAXA grades for the 28 sites in 2011 as with the years prior would suggest that these sites are experiencing high exposure levels to toxic pollution or general degradation. Again the highest ecological grade awarded to any site within the Foyle catchment for 2011 was a grade of moderate. As before, taken without reflection on the changes to the N-TAXA grades due to the methodology this would indicate that these sites are experiencing substantial levels of toxic pollution and/or general degradation. Unlike the ASPT grades, the N-TAXA grades did not show a vast improvement in the quality of sites, with improvements in one site often met with a deterioration in another.

The results from 5 sites in the Carlingford catchment presented similarly with contrasting ecological quality derived from the ASPT grades and N-TAXA grades. The ASPT grades for each of the 5 sites sampled presented grades which indicated minimal levels of organic pollution exposure at all five sites. Whereas the N-TAXA scores taken as they are would indicate levels of toxic pollution and/or habitat degradation that severely impacted on the ecological quality at these sites. The results for 2010 in the Carlingford catchment again indicated sites that the two sites sampled were experiencing high exposure to toxic pollution and/or degradation with a minimal exposure to organic pollution. The grades of the two sites sampled in 2010 remained the same from 2009, which would imply that the level of pressure acting on these sites, predominantly from toxic pollution and/or environmental degradation were consistent over the two years.

Again these results indicate that all of the sites across the 3 years were suffering from exposure to environmental degradation and/or toxic pollution at levels that were adversely shifting macroinvertebrate communities if the N-TAXA represented what is thought to be the 'true' ground results. However, the sites with moderate ASPT grade suggests that organic pollution was also a contributing factor in the deterioration of the ecological quality at some sites. The sporadic positioning of sites of the same grade would suggest that the surrounding land-use, especially the land-use upstream of each site is a key determinant of the ecological quality. Thus the interpretation and use of the ASPT and N-TAXA grades should be made with respect to this. The sites that exhibited consistent N-TAXA or ASPT grades across the three years suggests that these sampling sites may be within close proximity to point-source pollution. However, where the grade of a site fluctuated from year to year but still strongly indicated the exposure to anthropogenic influence, it may be possible that the timing of macroinvertebrate collection coincided with seasonal land-use practices that were more influential during the summer.

As sources of organic pollution can reduce the quality of sites as indicated by the results, it is important to be able to identify areas subject to adverse conditions especially when this can have implications downstream (Davie, 2008). The local land-use as previously stated is a considerable source of organic pollution, Abel (1996) summarised run-off from agricultural practices such as manure along with sewage effluents acts as primary sources.

If the N-TAXA grades here were true representations of the true quality it would be clear that all sites had been subjected to exposure to toxic pollution. As with sources of organic pollution, toxic pollutants entering watercourses from farm land run-off includes pesticide use and sheep dips can have significantly adverse affects on watercourses and the associated aquatic life (Ongley, 1996; Shardlow, 2006). Other sources of toxic pollution include industrial effluents, oils and chemicals from road run-off, forestry run-off and leaching from waste dump sites (Abel, 1996). The RICT approach would help identify sites that have been exposed or are experiencing such influences. This is of vital importance as spillages or watercourse exposure to toxic and organic pollutants often goes unreported, hence the importance of being able to identify sites of detrimental quality and the associated stressor. With respect to environmental degradation its influence may act on sites continually for years which may lead to sites within the vicinity of such degradation to consistently exhibit adverse quality. This may could of been a contributing factor for some sites that consistently achieved low N-TAXA grades over the three years. The use of the N-TAXA grades may lead to a confounding diagnoses of the source behind the low quality grade. However as previously proposed, further investigation would be required to identify the specific source, assess how it affects other components of the ecosystem and whether it's adversely affecting areas downstream. It must be emphasised that interpreting this data without consideration to the sampling procedure used within this study will lead to misconceptions over the true quality grades of each site. As the N-TAXA is a measurement of the

community richness at a site, it is clear that missing out on the manual search may result in a lower taxon richness being recorded. As a result the grades displayed in this study may be lower and present a biased ecological quality at each site. Furthermore, it is usually the ASPT grades that predominantly drives the overall classification of a site using the 'one out, all out' MINTA system (Owen & Guthrie, n.d). This would suggest, as predicted the N-TAXA grades exhibited here are not true representatives of the quality at each site. However, the MINTA system does have its merits when sampling procedure is adhered to, it provides a means to produce an overall grade for a site based on the lowest grade achieved from both index EQR grades. This will allow managerial schemes to target their efforts towards the worst performing sites.

The comparisons made from across the three years and across the catchments, presented a picture of the overall health of each catchment year to year but also highlighted local variations. The ability to monitor sites over temporal scales will allow continuous surveillance but also indicate if work conducted to improve site conditions is reflected in the invertebrate community structure. Previous use of ASPT scores has illustrated their use as determinants of recovery of sites after the of removal of point-source pollution (Wenn, 2008). The use of ASPT scores highlights their sensitivity to sporadic pollution events even after improvement works, further illustrating their use as an early warning system. Statistical comparisons between sites over temporal scales or spatial scales, will not only provide an indication of whether sites have changed grade but it can indicate if there are differences of the EQR score within grade boundaries. While the benefits of comparisons to management schemes are quite obvious, this approach was not adopted within this study due to the uncertainties surrounding the methodology. However, these comparisons would be particularly useful to assess sites that appear to maintain the same grade seasonally or annually. This further emphasises the potential use of ASPT and N-TAXA grades for surveillance purposes especially as polluting events are unpredictable and monitoring the recovery of sites is crucial to management.

If the Loughs Agency wished to start classifying sites in compliance to the WFD, the use of the RICT approach would allow them to conduct assessments for the biological status component which feeds into overall ecological classification status of sites. Where assessments for the WFD can be conducted, RICT is capable of assessing sites using additional biotic indices out with the assignments of ASPT and N-TAXA EQR grades. The new RIVPAC IV models and biotic indices of RICT now allows invertebrate data to be assessed using different indices which are indicative of different stressors. While some of the biotic indices are indicative of general and organic stresses (ASPT, N-TAXA, WHPT) others prove useful in identifying specific-stress types, such as acidity (AWIC Index), siltation (PSI Index) and flow stress (LIFE Index) available against multiple season combinations (Clarke *et al*, 2011).

For the years 2009-2011 the ecological quality of the sites in the Foyle and Carlingford catchments were exposed to various levels of pressure, which differed between sites and between years. It was found in only

three sites that the level of exposure to both organic and toxic pollution/degradation remained the same between the three years. This implies that the local land-use and fluctuations in its use is a significant determinant to the overall grade of a site and that variation in grades between years is most likely due to improvement works and/or sporadic polluting events. Improvements exhibited by the ASPT grades over the three years sheds a positive light on the quality of sites but emphasises that where organic pollution is minimal, sites still may be detrimentally impacted from other stressors. The RICT procedure provides a standardised method of assessing the ecological quality of macroinvertebrate community richness and structure for each site, as such this tool has been formally adopted by the environmental government bodies of the UK (SEPA, NIEA and EA) (Davy-Bowker *et al*, 2008). The provision of data that has been produced with the RICT will allow comparisons to be made over local and regional scales with assessments conducted by the NIEA. This tool provides a means to monitor trends, identify sites with the worst ecological quality and identify the type of stress that each site is subjected to.

The density grades for the 0+ salmon and trout across both catchments, indicates considerable discrepancies between salmon and trout densities within and between sites. Juvenile density readings are used to determine the expected smolt production for each river (Gibson & Cutting, 1993), subsequently sites with an absence of trout and/or salmon would suggest that the smolt run from these sites in the following years would be limited. This was also the case for the juvenile trout densities for both years. As with the differences displayed in the ASPT and N-TAXA grades of macroinvertebrates, it is probable that the local surrounding land-use had a significant influence on juvenile densities especially since local practices such as intensive forestry can adversely affect survival rates (Aas *et al*, 2010). Reasons behind underproducing juvenile salmon and trout densities have often been attributed to hostile habitat conditions associated with drought, high flow, flooding and pollution events (Ottoway & Clark, 1981; Jonsson & Jonsson, 2011). Natural quality variations of spawning and nursery habitats along and between watercourse may also explain the density variance between sites (Godfrey, 2005). Furthermore, differences of density grades between sites may be a factor of the differences in the natural carrying capacity of rivers (Uusitalo *et al*, 2005). It is likely that the surrounding land-use and the habitat conditions at each semi-quantitative electrofishing site were significant drivers of each density grade exhibited here and can account for the variability expressed across both catchments. One other factor which should be also considered is the presence of fish barriers which would partially or fully prevent the migration of salmon and trout to upstream sites (Aas *et al*, 2010). Natural, artificial and temporary barriers can all prevent adult fish migrations upstream and where temporary barriers appear such as substantial wood and rubbish debris or construction work, sites which had previously exhibited higher density grades may present as underproducing (SEPA, 2009; however see Opperman *et al*, 2006). Where this could be a contributing factor, there was only three sites over the 2 years that were consistently absent of both trout and salmon. Furthermore, factors which specifically influence the success of egg and alevin survival will contribute to the density grades at each site. Records of the juvenile densities

forms an important aspect of fisheries management as this will facilitate the decisions made to managerial practices (Niven *et al*, 2010). However additional assessments should be conducted to investigate why sites are underproducing, especially if this persists over years or the density grade deteriorates.

Using the trout and salmon density data in parallel with other biological and chemical assessments will further the understanding of the functioning of watercourses. However working knowledge of how complimentary assessments correlate is crucial to understanding the usefulness of data and it's most appropriate application. The results from the linear regression analysis indicated that there was not a significant correlation between the two biotic index scores and the juvenile trout grades for both years and juvenile salmon grades for 2009. This indicates that salmon and trout 0+ densities at sites can not be predicted using the ASPT and N-TAXA scores. The lack of a significant relationship between the ASPT and N-TAXA biological index scores and the salmon and trout grades may be due to several factors. Responses to different stressors between the two taxonomic groups is acknowledged to naturally differ. Organic and toxic pollution can completely remove sensitive macroinvertebrate families from water bodies whereas fish may only display with changes to reproductive ability, growth rates or feeding capability. A study by Berkman *et al*, (1986) contributed this to the direct effects of disturbance on invertebrates whereas the disturbance had indirect implications on fish. It has been recorded that conditions associated with low levels of organic pollution can have positive effects on fish productivity and survival which may explain why there was a lack of correlation between ASPT and juvenile fish scores (Aas *et al*, 2011). However low levels of organic pollution would be reflected in the ASPT score as the most sensitive families would disappear. Lower ASPT and N-TAXA scores may be responding to early exposure of a polluting event, where low concentrations or level of degradation may be high enough to detrimentally affect sensitive macroinvertebrates. However, the scale and period of exposure may not have met the threshold needed to change salmon and trout densities (Aas *et al*, 2011). It was also expected that the ASPT scores and the salmonid density grades would of presented with a stronger correlation, as fish are often used as indicators of organic pollution since they are sensitive to depleted oxygen conditions. However, the ASPT scores were not so low as to indicate severely depleted oxygen levels which may explain why there was no corresponding response. Furthermore, it is acknowledged that fish are sensitive to oxygen levels but have shown to be more sensitive to water abstraction, morphological alteration and general disturbance (Environmental Agency, 2011). Jonsson & Jonsson (2011) also summarised that water level, current velocity, river width, riparian cover and substrate cover were contributing factors to density variation in salmonids which may not induce a strong response from macroinvertebrates. As certain macroinvertebrates families are primary prey items of juvenile salmonids it was expected that fish density grades would of corresponded to lower N-TAXA scores (Aas *et al*, 2011). However as only the N-TAXA score in 2010 significantly correlated with salmon density grades, this would suggest that this may not be a contributing factor to lower density grades. Terrestrial invertebrates from riparian cover which forms prominent prey items to juvenile salmon and trout might

compensate any loss of aquatic invertebrates. It is clear from these results that monitoring is required on multiple levels, especially as fish may only reflect a problem far past the initial exposure where other aquatic biota had severely suffered. The presence of temporary and permanent fish barriers as previously mentioned may contribute to fish densities having an absent grade whereas the macroinvertebrate scores indicate minimal organic, toxic pollution and/or degradation. Where this is a distinct possibility, it is unlikely that the Loughs Agency would of conducted density measurements above known barriers. This is further emphasised, as previously highlighted, only three sites consistently exhibited the absence of fish. This would suggest that rather than fish barriers restricting migration there is some sort of disturbance acting on each juvenile population, which the ASPT and N-TAXA scores are not responding to. The difference in responses to organic pollution, toxic pollution and/or degradation highlights the importance of monitoring rivers using more than one taxonomic group. This further illustrates the importance of the RICT approach as to aid in the identification of sites deviating from reference conditions.

Rivers and their associated ecosystems are significant resources which require protection. It is crucial to implement a monitoring system which can identify sites at risk for the adoption of appropriate management schemes. Macroinvertebrate data collected by the Loughs Agency previously has given an insight to the ecological quality of sites, through this study environmental data was collected with the intention of using both datasets with the River Invertebrate Classification Tool. From this it was possible to identify type-specific stressors which were adversely influencing the quality of sites and then to monitor annual variation. It was possible to use the macroinvertebrate data to produce quality grades for the Foyle and Carlingford catchments. Furthermore, additional considerations with regards to the methodology would benefit future studies. Although the use of macroinvertebrates as bioindicators has provided an insight to the ecological quality of sites, it shouldn't be solely relied on. Differences were found between macroinvertebrate scores and juvenile salmon and trout densities, which suggests no correlation between the two bioindicators in response to adverse conditions. This demonstrates the need for a comprehensive monitoring system to assess different ecosystem components. Managerial decisions can then be based on multiple indicators, particularly grades produced by RICT which would put the Loughs Agency in a better position to assess sites in compliance with the Water Framework Directive.

5 RECOMMENDATIONS

Every effort should be made to limit the level of uncertainty associated with each sampling and processing procedure. The results obtained from the RICT approach have been deemed as not representing the 'true' quality of each site. Although both EQR index grades were subject to variation attributed to the methodology, the N-TAXA grades especially should be taken with caution. There were several factors that contributed to the possible misclassification of sites which should be rectified if this approach is to be used

by the Loughs Agency in the future. This is of significant importance if this approach is to be incorporated into managerial decisions and annual reports.

Emphasise in future studies should be made to strictly adhere to the requirements of the RICT methodology. The 1-minute manual search is a crucial component of the macroinvertebrate collection as it offers a means of collecting macroinvertebrates that are missed through the kick-sampling procedure. The exclusion of this manual search will result in the misclassification of grades to sites as discovered in this study, thus the manual search must form a component of the overall sampling procedure adopted.

Leading on from this, onsite river bank identification as adopted in 2011 may be less effective compared to identifications made in a laboratory (Cao *et al*, 2003). However, while this may prove to be the most time and resource efficient approach, the identification procedure should adhere to the accepted identification methodology set out within the RICT procedure manual. All biological samples should be appropriately stored and transferred back to a laboratory for identification.

The location of future sampling sites within a river stretch should be made with consideration to their proximity to bridges, roads or artificially modified channels. As recommended by the RICT procedure and Beatty *et al*, (2006) sampling should not be conducted in areas that could be greatly influence by the presence of such modifications. Shading from bridges can detrimentally affect the productivity of macrophytes, the density and diversity of benthic invertebrates within close proximity and underneath bridges (Struck *et al*, 2004). Changes to the water velocity due to modified channel structure associated with bridges is known to adversely affect invertebrate communities and abundances (Blettler & Marchese, 2005). Furthermore, runoff from roads and bridges has been recorded to lower ASPT and N-TAXA scores downstream of a bridge compared to scores taken upstream of such structures (Sriyaraj & Shutes, 2001). A minimum sampling distance upstream and downstream from anthropogenic structures must be adopted to limit their influence on biotic scores if this is not the intended purpose of investigation. However, avoiding bridges and such structures may be unfeasible within built up areas, subsequently a record of the structure and its proximity to sampling sites should be taken and referred to when interpreting results.

Obtaining or measuring annual records of the environmental variables for each site is necessary to limit the use of reference conditions produced from seasonally biased conditions. The environmental time variant data is required to be collected from all three recognised seasons, Spring (March-May), Summer (June-August) and Autumn (September -November) or be a modal representation of the annual environmental conditions. This requirement is asked irrespective of whether the macroinvertebrate sampling was conducted in one season or a combination of seasons. As such every effort must be made to acquire annual readings to avoid the bias of a single season. It would also prove more time efficient to measure the time invariant

environmental variables using GIS software, opposed to manually measuring distances using a planimeter. This would be especially advantageous is a large number of sites wished to be assessed.

While RICT does accommodate assessments based on samples collected from single seasons, samples must be collected during the seasons of spring and autumn if sites are to be assessed in compliance to the standards established by the WFD. Thus the time of the macroinvertebrate collection procedure should meet those required of RICT.

Full compliance with the required methodology asked of RICT will allow for the Loughs Agency to incorporate an additional level of monitoring that can grade the quality of tributaries and rivers of interest based on their invertebrate assemblages. This mode of assessment as demonstrated can provide crucial information on the type of disturbance or pollution that a site may be experiencing and allow for further investigation to be directed towards poorer sites. The adoption of biological monitoring already plays a crucial component of the Loughs Agency who allocates time and resources towards assessing water quality from an alternative but complimentary approach. However, the use of the RICT approach would allow for site-specific grades factoring in natural variations of ASPT and N-TAXA scores associated with different rivers. The standardised approach will allow the Loughs Agency to make comparisons between sites across temporal and spatial scales and work towards assessing sites in accordance to the WFD if worthwhile and desirable.

A future strategy should then involve more appropriate environmental data, sampling seasons and invertebrate collection and identification methods which all fall directly inline with the River Invertebrate Classification Tool.

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7 ABBREVIATIONS AND ACRONYMS

ASPT - Average Score Per Taxon.

AWIC - Acid Waters Indicator Community index.

BMWP - Biological Monitoring Working Party.

EA - Environmental Agency.

EQI - Environmental Quality Index.

EQR - Ecological Quality Ratio.

LIFE - Lotic Invertebrate index for Flow Evaluation.

NIEA - Northern Ireland Environmental Agency.

N-TAXA - Number of taxa.

PSI - Proportion of Sediment-sensitive Invertebrates.

RICT - River Invertebrate Classification Tool.

RIVPACS - River Invertebrate Prediction and Classification System.

WFD - Water Framework Directive.

WHPT - Walley Hawkes Paisley Trigg index.

8 APPENDICES

Appendix I

BMWP Macroinvertebrate Taxa	Sensitivity Score
Aeshnidae	8
Ancylus group (Ancylidae, Acroloxidae, Ferrissia)	6
Aphelocheiridae	10
Asellidae	3
Astacidae	8
Baetidae	4
Beraeidae	10
Brachycentridae	10
Caenidae	7
Calopterygidae	8
Capniidae	10
Chironomidae	2
Chloroperlidae	10
Coenagrionidae	6
Cordulegastridae	8
Corduliidae	8
Corixidae	5
Corophiidae	6

BMWP Macroinvertebrate Taxa	Sensitivity Score
Dendrocoelidae	5
Dryopidae	5
Dytiscidae (incl. Noteridae)	5
Elmidae	5
Ephemerellidae	10
Ephemeridae	10
Erpobdellidae	3
Gammaridae (incl. Crangonyctidae & Niphargidae)	6
Gerridae	5
Glossiphoniidae	3
Goeridae	10
Gomphidae	8
Gyrinidae	5
Haliplidae	5
Heptageniidae	10
Hirudinidae	3
Hydrobiidae (incl. Bithyniidae)	3
Hydrometridae	5
Hydrophilidae (incl. Hydraenidae, Helophoridae, Georissidae & Hydrochidae)	5
Hydropsychidae	5
Hydroptilidae	6
Hygrobiidae	5

BMWP Macroinvertebrate Taxa	Sensitivity Score
Lepidostomatidae	10
Leptoceridae	10
Leptophlebiidae	10
Lestidae	8
Leuctridae	10
Libellulidae	8
Limnephilidae (incl. Apataniidae)	7
Lymnaeidae	3
Mesoveliidae	5
Molannidae	10
Naucoridae	5
Nemouridae	7
Nepidae	5
Neritidae	6
Notonectidae	5
Odontoceridae	10
Oligochaeta	1
Perlidae	10
Perlodidae	10
Philopotamidae	8
Phryganeidae	10
Physidae	3

BMWP Macroinvertebrate Taxa	Sensitivity Score
Piscicolidae	4
Planariidae (incl. Dugesiidae)	5
Planorbidae (excl. Ancylidae)	3
Platycnemididae	6
Pleidae	5
Polycentropodidae	7
Potamanthidae	10
Psychomyiidae (incl. Ecnomidae)	8
Rhyacophilidae (incl. Glossosomatidae)	7
Scirtidae	5
Sericostomatidae	10
Sialidae	4
Simuliidae	5
Siphlonuridae	10
Sphaeriidae	3
Taeniopterygidae	10
Tipulidae	5
Unionidae	6
Valvatidae	3
Viviparidae	6

- *This list was produced by The United Kingdom Advisory Group (WFD -UKTAG, 2008)*