Final Report **Project WFD61c**

A system for classifying the ecological potential of UK and Irish canals

March 2012

SNIFFER
SCOTLAND & NORTHERN IRELAND FORUM FOR ENVIRONMENTAL RESEARCH

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Research contractor
This document was produced by:

Dr Nigel Willby
Centre for River Ecosystem Science
Biological & Environmental Science
University of Stirling
Stirling
FK9 4LA
UK

SNIFFER's project manager
SNIFFER's project manager for this contract is:

Kirsty Irving, SNIFFER

SNIFFER's technical advisory group is:

Phil Green, Environment Agency – *Principal technical advisor*
Lucy Bolton, Environment Agency
Joe Caffrey, Central Fisheries Board
Dave Colvill, Scottish Environment Protection Agency
Tara Gallagher, Central Fisheries Board
Kirsty Irving, SNIFFER
Olivia Lassiere, British Waterways
Cormac McCarthy, Waterways Ireland
Patricia McCreesh, Environment Protection Agency
Graham Newman, British Waterways
Paula Treacy, Waterways Ireland

SNIFFER
First Floor, Greenside House
25 Greenside Place
EDINBURGH EH1 3AA
Scotland
UK
www.sniffer.org.uk

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EXECUTIVE SUMMARY

WFD61c: Canal Classification Tool (March, 2012)

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Background to research

Under the European Union's Water Framework Directive (WFD) member states must develop methods to classify the ecological status of their surface water bodies. Such classifications will be used to support operational and surveillance monitoring and to select appropriate programmes of measures for incorporation into River Basin Management Plans. In the case of artificial or heavily modified water bodies the WFD develops the concept of 'ecological potential' as a parallel to ecological status in natural water bodies. Navigation is recognised within the WFD as one reason why artificial water bodies may be created, or why the hydromorphological characteristics of natural water bodies may be modified. This report presents the final phase in the development of a system for classifying the ecological potential of canals (i.e. artificial cuts) in Britain and Ireland.

Objectives of research

(a) To devise a canal classification tool that can be used to classify canal waterbodies in Great Britain and Ireland by comparing their ecological condition to their expected ecological potential taking account of current levels of use for navigation.

(b) To provide sufficient information to determine the ecological potential of a canal water body based on the relevant biological, hydromorphological and physico-chemical quality elements as defined by the WFD.

Key findings and recommendations

1. Maximum Ecological Potential (MEP), as applied to canals, considers boat traffic to be an intrinsic characteristic of the canal system and therefore a source of permitted environmental variation. Thus, MEP is determined by the best available biology for any specified level of boat traffic, in combination with the physico-chemistry and hydro-morphology that occur in the absence of all other impacts.
2. Sufficient data and understanding only exists to develop biological classification tools for canals based on macroinvertebrates and macrophytes. Historic data has been collected using robust and tested approaches which are suitable for adoption in monitoring. The biological data available for the final phase of this project provided comprehensive coverage of canals in all parts of Britain and Ireland.
3. Several approaches to reference site screening were developed and tested due to the difficulty in linking biological data to pressure data. For macroinvertebrates screening initially by pressure data and supplemented by screening of biology at filtered sites was possible. For macrophytes it was necessary to identify sites with the best combined biology for a given level of

boat traffic followed by checking of filtered sites against the limited matched physico-chemical data that was available.

4. Different metrics were developed for macroinvertebrates and macrophytes to reflect various aspects of the WFD normative definitions and the ecological effects of different pressures (e.g. boat traffic, channel modification and nutrient enrichment). Analysis of available data indicates that with increasing traffic the flora and fauna of canals changes from that typical of a series of well vegetated ponds to that of a large turbid lowland river.
5. Macroinvertebrates are assessed by standard sampling at 3-5 sites in a water body carried out in the spring. Metrics for macroinvertebrates comprise the number of families (N_FAM) recorded at a site, the Average Score per Taxon (ASPT), based on a published revision of BMWP scores, the number of Ephemeroptera, Plecoptera and Trichoptera (EPT) families recorded, the number of Coleoptera and Hemiptera Families (COL+BUG) and the number of Mollusc and Odonata families (MOLL+ODON).
6. Macrophytes are assessed based on a survey of a 150m section of channel at 3-5 sites in a water body. Surveys must be carried out in the summer. Metrics for macrophytes comprise an estimate of Aquatic Plant Biomass (APB) based on grapnel sampling, the number of aquatic plant taxa (N_AQUA), the Macrophyte Fertility Index (MFI) which is based on the composition of aquatic and emergent vegetation, the number of emergent plant species (N_EMERG), and the relative extent of emergent vegetation as a proportion of the local channel width (REL_EMERG).
7. The population of metric values associated with each biological quality element in reference sites was used as the basis for comparison with observed values at test sites in order to obtain an Ecological Quality Index (EQI). A computationally simple and flexible approach, based on multiple linear regression, was used to obtain a site-specific prediction of reference metric values. Boat traffic, alkalinity and bank type were the key environmental predictors of reference values for the various metrics. Class boundaries of Ecological Potential for individual metrics were established statistically based on the frequency distribution of EQI values.
8. After calculation of EQI values for each individual metric all values are standardised to a common scale and class width using a piecewise linear transformation. This permits different metrics to be combined into an overall classification for each Biological Quality element
9. An approach is presented for macrophytes and macroinvertebrates that uses information from multiple metrics to provide an overall measure of ecological potential for that quality element. The method of combination provides sensitivity to nutrient-related and hydromorphological pressures, although outputs should not be treated as diagnostic.
10. Derivation of environmental standards for supporting physico-chemical variables is problematic due to the very limited datasets of matched biology and physico-chemistry. Values for Biological Oxygen Demand and ammonia derived using invertebrates appear to be broadly in line with those standards proposed for rivers. For nutrients, standards for orthophosphate suggested by macrophytes are comparable to those for the nearest equivalent river type. The most detailed analyses are possible in relation to suspended solids. These suggest that suspended solid concentrations of <25-35mg/L will be required in heavily trafficked canals (>6000 passages/yr) in order to ensure a light regime where macrophytes will achieve Good Ecological Potential. Suitable management is therefore likely to be needed to reduce mobilisation of the bed or erosion of banks. In the case of phytoplankton, which have not been treated here as a discrete biological quality element due to lack of data,

standards are proposed for chlorophyll concentrations that might be considered compatible with Maximum or Good Ecological Potential.

11. Hydromorphological assessment occupies a critical role in the classification of Ecological Potential of Heavily Modified or Artificial Water Bodies because of its link to mitigation measures and the current use of the water body.
12. A standardized protocol has been developed to assess hydromorphology on canals. This first considers a set of exemptions that are expected to constrain the hydromorphology to a higher level of modification relative to passive meandering lowland rivers. These exemptions take account of access requirements, provision of alternative functions, heritage designations and whether a canal is a designated commercial waterway, carries very high boat traffic (>10000 passages per year), or is situated on an embankment, or in an urban or industrial area. A check list of hydromorphological attributes is then provided which carry different scores depending on the level of usage of a waterway and channel width. Attributes related to bank structure have the largest influence on assessments of canal hydromorphology since they most directly control lateral connectivity with riparian habitat. These attributes are scored, based on observations, either in the field, or remotely using a combination of GIS and aerial imagery resources to obtain an assessment of the hydromorphology at a site. Multiple assessments are required in a single water body and should be coincident with locations used for biological sampling.
13. Application of the hydromorphological assessment technique to canals in Ireland and Scotland indicates that all sites would be classified at Good or Maximum Ecological Potential.
14. The current guidance on integrating classifications using different quality elements has been followed to provide an overall classification of ecological potential of canal water bodies consistent with the ‘one-out, all-out rule. Invasive non-native species (INNS) are considered separately as a supporting element within this approach and influence classification of Ecological Potential in a similar way to hydromorphology. If taxa included on a country-specific short list of high risk INNS are present in a waterbody it is considered to fall below Maximum Ecological Potential.
15. Application of the macrophyte and macroinvetertebrate classification approaches generally produces expected results at well known sites, although the one-out, all-out approach to combining these classifications may be over sensitive. Confidence in the outputs based on physico-chemical standards is also lower and this should ideally be reflected in the approach used to combine information from across quality elements into a water body classification. In these respects a weight of evidence approach would be a superior option for water body classification.

Key words: Ecological Potential, navigation, ecological assessment, multi-metric, hydro morphology

TABLE OF CONTENTS

EXECUTIVE SUMMARY

1 INTRODUCTION 1

1.1 BACKGROUND TO THE PROJECT AND ORGANISATION OF THIS REPORT 1
1.2 CANALS IN BRITAIN AND IRELAND 2
1.3 INTERPRETING ECOLOGICAL POTENTIAL 2

2 INVERTEBRATE-BASED CLASSIFICATION OF CANALS 4

2.1 SAMPLING PROTOCOL 4
2.1.1 Sample collection 4
2.1.2 Timing and frequency of sampling 4
2.1.3 Where and how many sites to sample 4
2.1.4 Sample processing 5
2.1.5 Collection of environmental data 5
2.1.6 Data storage 6
2.2 CALCULATING METRICS USED IN CLASSIFICATION 7
2.2.1 Screening of metrics 7
2.2.2 Number of families (N_FAM) 8
2.2.3 Average Score Per Taxon (ASPT) 10
2.2.4 Number of Ephemeroptera, Plecoptera and Trichoptera families (N_EPT) 10
2.2.5 Number of Mollusc and Odonata families (N_MOLL+ODON) 10
2.2.6 Number of Coleoptera and Hemiptera families (N_COL+BUGS) 10
2.2.7 CPET 11
2.2.8 Data quality and sampling efficiency 11
2.3 IDENTIFYING REFERENCE SITES 12
2.3.1 Data availability 12
2.3.2 Environmental data availability GIS etc 13
2.3.3 Data screening 13
2.4 CALCULATING EXPECTED VALUES FOR INVERTEBRATE METRICS 15
2.4.1 Data requirements 15
2.4.2 Expected N_FAM 17
2.4.3 Expected ASPT 18
2.4.4 Expected N_EPT 20
2.4.5 Expected N_COL+BUGS 21
2.4.6 Expected N_MOLL+ODON 21
2.5 CALCULATING AN ECOLOGICAL QUALITY INDEX (EQI) FOR INVERTEBRATE METRICS 22
2.5.1 N_FAM 22
2.5.2 ASPT 22
2.5.3 N_EPT 22
2.5.4 N_COL_BUGS 22
2.5.5 N_MOLL_ODON 23
2.6 COMBINING EQIS TO ACHIEVE AN INVERTEBRATE-BASED CLASSIFICATION 23
2.6.1 Establishing class boundaries 23
2.6.2 Rescaling EQI values to an EQR 25
2.6.3 Rules for combining EQR values 26
2.6.4 Implications for national classifications 27
2.7 INTEGRATING CPET-BASED CLASSIFICATIONS 30
2.8 DETERMINING THE EQR FOR A WATER BODY 30
2.9 WORKED EXAMPLE 31
2.9.1 Raw data 31
2.9.2 Calculating observed metric values 31
2.9.3 Calculating Expected metric values 33
2.9.4 Calculating an EQI for biological metrics 34
2.9.5 Converting the EQI to an EQR 34

2.9.6 Combining EQRs into a value for the site 35

3 MACROPHYTE-BASED CLASSIFICATION OF CANALS 36

3.1 SURVEY PROTOCOL 36
3.1.1 Survey methodology 36
3.1.2 Timing and frequency of surveying 36
3.1.3 Where and how many sites to survey 36
3.1.4 Collection of environmental data 37
3.1.5 Data storage 38
3.2 CALCULATING METRICS USED IN CLASSIFICATION 38
3.2.1 Screening of metrics 38
3.2.2 Aquatic Plant Biomass (APB) 38
3.2.3 Number of scoring aquatic plant taxa (N_AQUA) 39
3.2.4 Macrophyte Fertility Index (MFI) 42
3.2.5 Number of scoring emergent taxa (N_EMERG) 44
3.2.6 Relative extent of emergent vegetation (REL_EMERG) 45
3.2.7 Data quality and sampling efficiency 46
3.3 IDENTIFYING REFERENCE SITES 46
3.3.1 Data availability 46
3.3.2 Environmental data availability 46
3.3.3 Data screening 47
3.4 CALCULATING EXPECTED VALUES FOR MACROPHYTE METRICS 49
3.4.1 Data requirements 49
3.4.2 Expected APB 50
3.4.3 Expected N_AQUA 50
3.4.4 Expected MFI 53
3.4.5 Expected N_EMERG 53
3.4.6 Expected REL_EMERG 55
3.5 CALCULATING AN ECOLOGICAL QUALITY INDEX (EQI) FOR MACROPHYTE METRICS 55
3.5.1 APB 55
3.5.2 N_AQUA 56
3.5.3 MFI 56
3.5.4 N_EMERG 56
3.5.5 REL_EMERG 56
3.6 COMBINING EQIS TO ACHIEVE A MACROPHYTE-BASED CLASSIFICATION 56
3.6.1 Establishing class boundaries 56
3.6.2 Rescaling EQI values to an EQR 59
3.6.3 Rules for combining EQR values 59
3.6.4 Implications for national classifications 60
3.6.5 Comparing classifications of ecological potential based on macrophytes and invertebrates 63
3.7 DETERMINING THE EQR FOR A WATER BODY 64
3.8 WORKED EXAMPLE 65
3.8.1 Raw data 65
3.8.2 Calculating observed metric values 65
3.8.3 Calculating Expected metric values 66
3.8.4 Calculating an EQI for biological metrics 67
3.8.5 Converting the EQI to an EQR 68
3.8.6 Combining EQRs into a value for the site 68

4 HYDROMORPHOLOGICAL ASSESSMENT 70

4.1 A HYDROMORPHOLOGICAL PERSPECTIVE ON ECOLOGICAL POTENTIAL 70
4.2 HYDROMORPHOLOGY AND MITIGATION 71
4.3 HYDROMORPHOLOGICAL ASSESSMENT 72
4.3.1 A two step approach to assessing hydromorphology on canals 72
4.3.2 Undertaking assessment in practice 78
4.3.3 Classifying ecological potential using HM assessments 78

4.4 EXAMPLE OF HYDROMORPHOLOGICAL ASSESSMENT 79

5 CLASSIFICATION OF ECOLOGICAL POTENTIAL BASED ON PHYSICO-CHEMICAL VARIABLES 82

5.1 INTRODUCTION 82
5.2 WATER QUALITY ASSOCIATED WITH REFERENCE/MEP CONDITIONS AND GOOD/MODERATE BOUNDARY AS INFERRED FROM INVERTEBRATES 82
5.2.1 Ammonia (as N) 83
5.2.2 Biochemical Oxygen Demand 83
5.2.3 Chloride 84
5.3 WATER QUALITY ASSOCIATED WITH REFERENCE/MEP CONDITIONS AND GOOD/MODERATE BOUNDARY AS INFERRED FROM MACROPHYTES 85
5.3.1 Soluble Reactive Phosphorus 85
5.3.2 Total Oxidised Nitrogen (TON) 87
5.3.3 Turbidity and TSS 87
5.3.4 Chlorophyll 92
5.4 SPECIFIC POLLUTANTS 93
5.5 SUMMARY OF THRESHOLDS FOR SELECTED PHYSICO-CHEMICAL PARAMETERS AT HIGH/GOOD AND GOOD/MODERATE BOUNDARIES 94

6 COMBINING CLASSIFICATIONS INTO AN OVERALL ASSESSMENT OF ECOLOGICAL POTENTIAL 95

6.1 INTRODUCTION 95
6.2 INVASIVE SPECIES 96
6.3 DATA REQUIRED FOR FULL CLASSIFICATION OF ECOLOGICAL POTENTIAL 100
6.4 WORKED EXAMPLE OF CLASSIFICATION OF ECOLOGICAL POTENTIAL 100

7 REFERENCES 103

FIGURES

FIGURE 2.1. ORDINATION BASED ON DETRENDED CORRESPONDENCE ANALYSIS (DCA) OF FAMILY LEVEL INVERTEBRATE DATA SHOWING DISTRIBUTION OF SAMPLES IN FULL DATABASE ACCORDING TO SEASON OF COLLECTION 12
FIGURE 2.2. ORDINATION BASED ON DETRENDED CORRESPONDENCE ANALYSIS (DCA) OF FAMILY LEVEL INVERTEBRATE DATA SHOWING DISTRIBUTION OF INVERTEBRATE REFERENCE (REF) AND NON-REFERENCE (NON-REF) SITES AFTER SCREENING BY BIOLOGY AND PRESSURE DATA..... 14
FIGURE 2.3. FLOWCHART FOR DETERMINING EXPECTED CHANNEL TYPE 16
FIGURE 2.4. RELATIONSHIP BETWEEN THE NUMBER OF SCORING TAXA AND BOAT TRAFFIC IN THREE DIFFERENT CHANNEL TYPES 18
FIGURE 2.5. RELATIONSHIP BETWEEN ASPT AND BOAT TRAFFIC IN THREE DIFFERENT CHANNEL TYPES 20
FIGURE 2.6. RELATIONSHIP BETWEEN THE N_COL+BUGS AND BOAT TRAFFIC IN THREE DIFFERENT CHANNEL TYPES 21
FIGURE 2.7. HISTOGRAMS SHOWING FREQUENCY DISTRIBUTION OF EQI VALUES FOR DIFFERENT METRICS IN A POPULATION OF 165 REFERENCE SITES..... 24
FIGURE 2.8. CLASSIFICATION OF SITES ON IRISH CANALS BASED ON MACROINVERTEBRATES. UPPER PANEL GRAND CANAL, INCLUDING BARROW LINE (BL), LOWER PANEL ROYAL CANAL. RED DASHED LINE IS THE GLOBAL MEAN VALUE ACROSS ALL SITES ASSESSED. CLASSIFICATIONS BASED ON SAMPLES COLLECTED SPRING 2006 BY CENTRAL FISHERIES BOARD. 29
FIGURE 2.9. MAP DETAILS AND IMAGE FOR SITE USED IN WORKED EXAMPLE, LAMBHILL, GLASGOW, FORTH & CLYDE CANAL. 31
FIGURE 3.1. RELATIONSHIP BETWEEN APB AND BOAT TRAFFIC IN ALL SURVEYS WITH RESPECT TO CHANNEL TYPE. 40
FIGURE 3.2. RELATIONSHIP BETWEEN N_AQUA AND BOAT TRAFFIC IN ALL SURVEYS WITH RESPECT TO COUNTRY AND CHANNEL TYPE. 43
FIGURE 3.3. RELATIONSHIP BETWEEN REL_EMERG AND BOAT TRAFFIC IN ALL SURVEYS WITH RESPECT TO CHANNEL TYPE. 44
FIGURE 3.4. RELATIONSHIP BETWEEN REL_EMERG AND BOAT TRAFFIC IN ALL SURVEYS WITH RESPECT TO CHANNEL TYPE. 45
FIGURE 3.5. ORDINATION BY DETRENDED CORRESPONDENCE ANALYSIS (DCA) OF MACROPHYTE SURVEY DATA SHOWING DIFFERENCES IN COMPOSITION AT NATIONAL OR REGIONAL LEVELS. 47
FIGURE 3.6. ORDINATION BASED ON DETRENDED CORRESPONDENCE ANALYSIS (DCA) OF MACROPHYTE SURVEY SHOWING DISTRIBUTION OF MACROPHYTE REFERENCE (REF) SITES AND NON-REFERENCE (NON-REF) AFTER SCREENING BY BIOLOGY AND PRESSURE DATA..... 49
FIGURE 3.7. RELATIONSHIP BETWEEN AQUATIC PLANT BIOMASS (APB, Y AXIS) AND BOAT TRAFFIC IN REFERENCE SITES IN SCOTLAND (LEFT) AND ENGLAND & WALES AND IRELAND (RIGHT). 50
FIGURE 3.8. RELATIONSHIP BETWEEN AQUATIC PLANT SPECIES RICHNESS (NAQUA, Y AXIS) AND BOAT TRAFFIC IN REFERENCE SITES IN SOFT (LEFT) AND HARD ENGINEERED CHANNELS (RIGHT). 51
FIGURE 3.9. RELATIONSHIP BETWEEN ALKALINITY AND MFI AT CANAL REFERENCE SITES ILLUSTRATING DISTRIBUTION OF REGIONAL SURVEYS..... 53
FIGURE 3.10. RELATIONSHIP BETWEEN N_EMERG AND CHANNEL TYPE AT CANAL REFERENCE SITES ILLUSTRATING REGIONAL DIFFERENCES, NOTABLY IN RELATION TO TRAPEZOIDAL CHANNELS. 54
FIGURE 3.11. RELATIONSHIP BOAT TRAFFIC AND REL_EMERG AT CANAL REFERENCE SITES ILLUSTRATING DEPENDENCY ON CHANNEL TYPE..... 55
FIGURE 3.12. HISTOGRAMS SHOWING FREQUENCY DISTRIBUTION OF EQI VALUES FOR DIFFERENT MACROPHYTE METRICS IN A POPULATION OF 127 REFERENCE SITES..... 58
FIGURE 3.13. CLASSIFICATION OF SITES ON IRISH CANALS BASED ON MACROPHYTES. UPPER PANEL GRAND CANAL, INCLUDING BARROW LINE, LOWER PANEL ROYAL CANAL. RED DASHED LINE IS THE GLOBAL MEAN VALUE ACROSS ALL SITES ASSESSED. CLASSIFICATIONS BASED ON SAMPLES COLLECTED SUMMER 2006 BY CENTRAL FISHERIES BOARD. 62
FIGURE 3.14. PERCENTAGE OF OCCASIONS DIFFERENT GROUPS OF METRICS DICTATED THE CLASSIFICATION OF A SUBSET OF SITES FROM GB AND IRELAND..... 63
FIGURE 4.1. AERIAL IMAGE (GOOGLE EARTH, 2009) AND GROUND-BASED PHOTOGRAPH FOR SITE USED IN WORKED EXAMPLE, LAMBHILL, GLASGOW, FORTH & CLYDE CANAL. 79
FIGURE 5.1. GLOBAL RELATIONSHIP BETWEEN ANNUAL MEAN SOLUBLE REACTIVE PHOSPHORUS (SRP) CONCENTRATIONS AND MACROPHYTE FERTILITY INDEX (MFI) COMPARING RIVERS AND CANALS. ... 86
FIGURE 5.2. GLOBAL RELATIONSHIP BETWEEN ANNUAL MEAN TOTAL OXIDISED NITROGEN (TON) CONCENTRATIONS AND MACROPHYTE FERTILITY INDEX (MFI) COMPARING RIVERS AND CANALS. ... 87
FIGURE 5.3. RELATIONSHIPS BETWEEN BOAT TRAFFIC AND SPOT MEASURED TSS OR TUBIDITY (AS NTU) AT HEIGHT OF ANNUAL BOAT ACTIVITY IN CANALS (DATA: N.WILLBY). RELATIONSHIPS ARE MODELLED USING LOCALLY WEIGHTED SCATTER PLOT SMOOTHING (LOWESS). 88

FIGURE 5.4. RELATIONSHIPS BETWEEN APB AND MEASURES OF TRANSPARENCY IN CANALS. THE RELATIONSHIP IS FITTED BY LOWESS. THE DASHED LINES ARE USED TO INDICATE THE TSS OR TURBIDITY ASSOCIATED WITH THE VALUES OF APB THAT INDICATE THE G/M THRESHOLDS AT FOUR CONTRASTING BOAT TRAFFIC DENSITIES GIVEN IN TABLE 5.6. 89
FIGURE 5.5 RELATIONSHIPS BETWEEN N_AQUA AND MEASURES OF TRANSPARENCY IN CANALS. THE RELATIONSHIP IS FITTED BY LOWESS. THE DASHED LINES ARE USED TO INDICATE THE TSS OR TURBIDITY ASSOCIATED WITH THE VALUES OF N_AQUA THAT INDICATE THE G/M THRESHOLDS AT FOUR CONTRASTING BOAT TRAFFIC DENSITIES GIVEN IN TABLE 5.6. 90
FIGURE 5.6. RELATIONSHIP BETWEEN TSS AND SECCHI DEPTH IN CANALS (PSYM DATASET)..... 92
FIGURE 6.1. THE BASIS FOR CLASSIFICATION OF SURFACE WATER STATUS OF WATER BODIES THROUGH THE ONE OUT ALL OUT APPROACH. TAKEN FROM THE DRAFT UK CLASSIFICATION SCHEME FOR SURFACE WATERS (UKTAG, 2007) 96

TABLES

TABLE 2.1. RELATIONSHIPS BETWEEN INVERTEBRATE METRICS USED IN CLASSIFICATION AND A SUITE OF PRESSURE VARIABLES, OR INDICATORS OF PRESSURES, RELEVANT TO CANALS IN GB & IRELAND. .. 7
TABLE 2.2. INDICATIVE VALUES FOR INVERTEBRATE METRICS BASED ON THE MEAN (UPPER LINE) AND INTER QUARTILE RANGE (LOWER LINE) OF ALL AVAILABLE SPRING SAMPLES FROM CANALS IN ENGLAND & WALES, SCOTLAND AND IRELAND..... 8
TABLE 2.3. FULL LIST OF SCORING FAMILIES, REVISED BMWP SCORES FOR CALCULATING REVISED ASPT AND MEMBERSHIP OF EPT, COL+BUG AND MOLL+ODON GROUPS FOR CALCULATING ADDITIONAL RICHNESS METRICS..... 9
TABLE 2.4. ANNUAL MEAN VALUES OF SELECTED PRESSURE-INDICATING VARIABLES (MEDIAN AND LOWER OR UPPER PERCENTILES) IN REFERENCE SITES..... 15
TABLE 2.5. MEDIAN FAMILY RICHNESS IN REFERENCE SITES IN DIFFERENT COUNTRIES/ REGIONS, AND APPROPRIATE CORRECTION FACTORS FOR CALCULATING EXPECTED VALUES FOR THE METRIC N_FAM. 17
TABLE 2.6. FUNCTIONS AND COEFFICIENTS FOR MODELING EXPECTED VALUES OF MACROINVERTEBRATE METRICS. 19
TABLE 2.7. CLASS BOUNDARIES FOR DIFFERENT LEVELS OF ECOLOGICAL POTENTIAL IN FIVE MACROINVERTEBRATE METRIC EQIS..... 24
TABLE 2.8. ECOLOGICAL POTENTIAL BOUNDARIES, AFTER IMPLEMENTATION OF PIECEWISE LINEAR TRANSFORMATION TO STANDARDISE EQI VALUES TO AN EQR SCALING..... 25
TABLE 2.9. PROPORTION OF SAMPLES ASSIGNED TO DIFFERENT CLASSES OF ECOLOGICAL POTENTIAL BASED ON DIFFERENT RULES FOR COMBINING METRICS..... 26
TABLE 2.10. NUMBERS AND PROPORTIONS OF SITES IN DIFFERENT COUNTRIES/REGIONS CLASSIFIED AT DIFFERING LEVELS OF ECOLOGICAL POTENTIAL ACCORDING TO A COMBINATION RULE BASED ON THE MINIMUM OF EITHER 'WATER QUALITY' OR 'HYDRO-MORPHOLOGICAL' GROUPS OF METRICS. 27
TABLE 2.11. PROPORTION OF TIMES DIFFERENT GROUPS OF METRICS DICTATED THE CLASSIFICATION OF A SUBSET OF 106 SITES FROM GB, SHOWING THE EFFECT OF INTEGRATION OF CPET ON THE OVERALL ASSESSMENT. 28
TABLE 2.12. ENVIRONMENTAL DATA FOR CASE STUDY SITE AT LAMBHILL, GLASGOW, FORTH & CLYDE CANAL. 32
TABLE 2.13. RAW BIOLOGICAL DATA FOR CASE STUDY SITE AT LAMBHILL, GLASGOW, FORTH & CLYDE CANAL, AS SAMPLED BY SEPA IN SPRING 2006. 32
TABLE 2.14. SUMMARY OF OBSERVED AND EXPECTED METRIC VALUES, EQI AND EQR VALUES AND OVERALL ECOLOGICAL POTENTIAL FOR CASE STUDY SITE AT LAMBHILL, GLASGOW, FORTH & CLYDE CANAL. 33
TABLE 3.1. RELATIONSHIPS BETWEEN MACROPHYTE METRICS USED IN CLASSIFICATION AND A SUITE OF PRESSURE VARIABLES, OR INDICATORS OF PRESSURES, RELEVANT TO CANALS IN GB & IRELAND. 39
TABLE 3.2. FULL LIST OF SCORING MACROPHYTE TAXA USED IN CANAL ASSESSMENT. 41
TABLE 3.3. ANNUAL MEAN VALUES OF SELECTED PRESSURE INDICATING VARIABLES (MEDIAN AND LOWER OR UPPER PERCENTILES) IN MACROPHYTE REFERENCE SITES. 48
TABLE 3.4. FUNCTIONS AND COEFFICIENTS FOR MODELING EXPECTED VALUES OF MACROPHYTE METRICS. 52
TABLE 3.5. CLASS BOUNDARIES FOR DIFFERENT LEVELS OF ECOLOGICAL POTENTIAL IN FIVE MACROPHYTE METRIC EQIS. 59
TABLE 3.6. ECOLOGICAL POTENTIAL BOUNDARIES AFTER IMPLEMENTATION OF PIECEWISE LINEAR TRANSFORMATION TO STANDARDISE EQI VALUES TO AN EQR SCALING..... 59
TABLE 3.7. PROPORTION OF SURVEYS ASSIGNED TO DIFFERENT CLASSES OF ECOLOGICAL POTENTIAL BASED ON DIFFERENT RULES FOR COMBINING METRICS, BASED ON 762 SURVEYS FOR WHICH AT LEAST THREE METRICS WERE AVAILABLE..... 60
TABLE 3.8. NUMBERS AND PROPORTIONS OF SURVEYS IN DIFFERENT COUNTRIES/REGIONS CLASSIFIED AT DIFFERING LEVELS OF ECOLOGICAL POTENTIAL ACCORDING TO A COMBINATION RULE BASED ON THE AVERAGE OF THREE GROUPS OF METRICS..... 61
TABLE 3.9. COMPARISON OF CLASSIFICATIONS OF CANAL SITES IN IRELAND AND SCOTLAND SAMPLED IN 2006 AND ASSESSED USING INVERTEBRATES, MACROPHYTES OR BOTH QUALITY ELEMENTS. 64
TABLE 3.10. ENVIRONMENTAL DATA FOR CASE STUDY SITE AT LAMBHILL, GLASGOW, FORTH & CLYDE CANAL. 66
TABLE 3.11. RAW MACROPHYTE SURVEY DATA FOR CASE STUDY SITE AT LAMBHILL, GLASGOW, FORTH & CLYDE CANAL, AS SAMPLED BY SEPA IN SUMMER 2006..... 66
TABLE 3.12. SUMMARY OF OBSERVED AND EXPECTED METRIC VALUES, EQI AND EQR VALUES AND OVERALL ECOLOGICAL POTENTIAL FOR CASE STUDY SITE AT LAMBHILL, GLASGOW, FORTH & CLYDE CANAL. 67

TABLE 4.1. SPECIFIC EXEMPTIONS TO BE CONSIDERED IN ASSESSMENTS OF CANAL HYDROMORPHOLOGY (I.E. DEFAULT NO LONGER APPLIES AND CAN BE JUSTIFIED). 73
TABLE 4.2. MATRIX OF BOAT TRAFFIC AND CHANNEL SIZE, TO BE USED IN ASSESSING HYDROMORPHOLOGY AT SITES WHERE EXEMPTIONS DO NOT APPLY..... 74
TABLE 4.3. CANAL HYDROMORPHOLOGY ASSESSMENT SHEET SHOWING SCORES FOR DIFFERENT ATTRIBUTES. 76
TABLE 4.4. SUGGESTED MINIMUM OPEN CHANNEL STANDARDS TO ACHIEVE HYDROCOMPLIANCE WHERE FORMAL ASSESSMENTS HAVE NOT BEEN UNDERTAKEN..... 77
TABLE 4.5. EXAMPLE SITE SHOWING COMPLETED HYDROMORPHOLOGY ASSESSMENT SHEET FOR WORKED EXAMPLE SITE. RELEVANT ATTRIBUTES AND THEIR SCORES ARE HIGHLIGHTED. 81
TABLE 5.1. INDICATED MEAN CONCENTRATIONS OF AMMONIA (μGL^{-1}) AT THE MAXIMUM/GOOD AND GOOD/MODERATE EP BOUNDARIES BASED ON SIGNIFICANT RELATIONSHIPS BETWEEN AMMONIA AND DIFFERENT INVERTEBRATE METRICS 83
TABLE 5.2. INDICATED VALUES FOR BOD (MGL^{-1}) AT THE MAXIMUM/GOOD AND GOOD/MODERATE EP BOUNDARIES BASED ON SIGNIFICANT RELATIONSHIPS BETWEEN BOD AND DIFFERENT INVERTEBRATE METRICS 84
TABLE 5.3. INDICATED VALUES FOR CHLORIDE (MG/L) AT THE MAXIMUM/GOOD AND GOOD/MODERATE EP BOUNDARIES BASED ON RELATIONSHIPS BETWEEN CHLORIDE AND DIFFERENT INVERTEBRATE METRICS 84
TABLE 5.4. INDICATED VALUES FOR SRP (μGL^{-1}) AND TON (MGL^{-1}) AT THE MAXIMUM/GOOD AND GOOD/MODERATE ECOLOGICAL POTENTIAL BOUNDARIES BASED ON THEIR GLOBAL RELATIONSHIPS WITH MFI. NOTE THAT THE SRP VALUES CANNOT BE APPLIED TO IRISH CANALS. 86
TABLE 5.5. AQUATIC PLANT BIOMASS AND SPECIES RICHNESS EXPECTED AT MAXIMUM ECOLOGICAL POTENTIAL AND GEP/MEP BOUNDARY IN CANALS AT VARYING BOAT TRAFFIC DENSITIES. 91
TABLE 5.6. TSS (MG/L) AND TURBIDITY (AS NTU) ASSOCIATED WITH THRESHOLD VALUES OF AQUA_N AND AQUA_B AT THE G/M BOUNDARY AT FOUR CONTRASTING BOAT DENSITIES..... 91
TABLE 5.7. PROPOSED UPPER LIMITS OF ANNUAL MEAN CHLOROPHYLL (UG/L) IN CANALS NEEDED TO ACHIEVE GOOD ECOLOGICAL POTENTIAL. VALUES FOR HG AND GM BOUNDARIES FOR THE NEAREST EQUIVALENT LAKE TYPES ARE ALSO GIVEN. 93
TABLE 5.8. SUMMARY OF WATER QUALITY PARAMETERS AT MAXIMUM/GOOD AND GOOD/MODERATE ECOLOGICAL POTENTIAL BOUNDARIES..... 94
TABLE 6.1. STATUS OF QUALITY ELEMENTS CONSIDERED IN CLASSIFICATION OF ECOLOGICAL POTENTIAL OF CANAL WATER BODIES. 95
TABLE 6.2. NUMBER, DISTRIBUTION AND FREQUENCY OF SAMPLING FOR DIFFERENT QUALITY ELEMENTS IN A TYPICAL SIZED CANAL WATER BODY (E.G. 20KM IN LENGTH)..... 100
TABLE 6.3. WORKED EXAMPLE SHOWING MECHANISM FOR COMBINING INFORMATION FROM DIFFERENT QUALITY ELEMENTS TO ACHIEVE AN OVERALL CLASSIFICATION OF ECOLOGICAL POTENTIAL FOR A CANAL WATER BODY..... 102

ACRONYMS USED

ASPT	Average Score per Taxon
AWB	Artificial Water Body
BAP	Biodiversity Action Plan
BMWP	Biological Monitoring Working Party
BOD	Biochemical Oxygen Demand
BQE	Biological Quality Element.
BW	British Waterways
CEH	Centre for Ecology & Hydrology
CIS	Common Implementation Strategy
CoC	Confidence of Class
CPET	Chironomid Pupal Exuviae Technique
CSM	Common Standards Monitoring
EA	Environment Agency
DCA	Detrended Correspondence Analysis
DO	Dissolved Oxygen
EPT	Ephemeroptera, Plecoptera and Trichoptera
EQI	Ecological Index Index
EQR	Ecological Quality Ratio
GEP	Good Ecological Potential
GLM	General Linear Model
GQA	General Quality Assessment
HMWB	Heavily Modified Water Body
INNS	Invasive Non-Native Species
LEAFPACS	WFD Classification Tool for assessing ecological quality of lakes and rivers using aquatic macrophytes
MDA	Multiple Discriminant Analysis
MEP	Maximum Ecological Potential
MOC	Minimum Open Channel

PCT	Ponds Conservation Trust
PSYM	Predictive System for Multimetrics
RIVPACS	River Invertebrate Prediction and Classification System
SAC	Special Area for Conservation
SEPA	Scottish Environment Protection Agency
SRP	Soluble Reactive Phosphorus
SSSI	Site of Special Scientific Interest
STARBUGS	STAR project biological uncertainty guidance software
TWINSPAN	Two Way Indicator Species Analysis
UKTAG	UK Technical Advisory Group on WFD implementation
WFD	Water Framework Directive

1 INTRODUCTION

1.1 Background to the project and organisation of this report

Under the EU Water Framework Directive (WFD; European Union, 2000) a method is required to enable the classification of water bodies on the basis of their ecological status. Such classifications will be used to support operational and surveillance monitoring undertaken by the regulatory bodies for the water environment in Britain and Ireland and to select appropriate programmes of measures for incorporation into River Basin Management Plans. The Directive develops the concept of 'ecological potential' in artificial or heavily modified water bodies as a parallel to ecological status in natural water bodies, such as lakes and rivers, for which separate tools have been developed.

This report was commissioned by SNIFFER as the third and final phase in the development of a system for classifying the ecological status of canals in Britain and Ireland. This report builds on the Phase 1 work (WFD61a: Willby et al., 2008) but fully integrates data collected in Scotland and Ireland in Phase 2 and incorporates the consequences of hydromorphological assessments on water body classification.

The organisation of this report is summarised below:

- **Chapter 1** considers the nature of the resource and issue of Ecological Potential and how this can be interpreted with particular reference to canal water bodies.
- **Chapter 2** considers the classification of ecological potential based on data obtained from standardised sampling of macroinvertebrates. Specifically it addresses the treatment of the available data, how data from different sources has been harmonised and linked to supporting environmental data and the potential implications of invertebrate-based classifications.
- **Chapter 3** discusses canal classification based on standard macrophyte survey data, focussing on the consequences of a revised approach to reference site selection and the integration of recent survey data from Ireland and Scotland. Implications of water body classification based on a 'one-out, all out' approach are then considered.
- **Chapter 4** deals with hydromorphological assessment and explores a streamlined approach for direct hydromorphological assessment. A suitable protocol for the latter is proposed and discussed with worked examples.
- **Chapter 5** considers the relationship between biology and physico-chemistry and uses these relationships to suggest suitable standards for a range of variables which should support biology in canals at Good Ecological Potential. Comparisons are then made with the standards proposed by UKTAG for lakes and rivers.
- **Chapter 6** discusses suitable approaches, consistent with UKTAG guidance, for combining information from different groups of quality elements in order to achieve an overall classification of Ecological Potential for a canal water body.

1.2 Canals in Britain and Ireland

Canals in the UK and Ireland were constructed primarily during the period 1750-1870, the greatest density being found in the lowlands of central England. Channels are typically 10-20m wide with a maximum depth of 1-2m. In some cases these artificial 'cuts' were designed to connect with navigable lakes or rivers that were subsequently modified for the purpose of navigation. Following a prolonged period of decline, and in some cases abandonment and infilling that continued until the 1950s, canals have enjoyed a revival as a major recreational boating resource. In Britain some 2500 km are currently in water, the total having increased significantly over the last two decades through major programmes to restore derelict lengths to navigation. Several hundred km of canal in Britain are not currently navigable, some of which are the subject of restoration campaigns at various stages of completion. These include sections not currently in water. In Ireland approximately 330 km of canals are currently navigable and are managed by Waterways Ireland. In Northern Ireland canals that are outside the remit of Waterways Ireland are managed by a combination of borough Councils and Trusts.

1.3 Interpreting Ecological Potential

Navigation, including port facilities or recreation, is recognised within the WFD as one reason why artificial water bodies may be created, or why the hydromorphological characteristics of natural water bodies may be modified. In most cases the changes that would be required for canals to meet Good Ecological Status (GES) would be such that their original function could no longer be met and, independently of this, would be difficult to justify on the grounds of cost.

Sites are classified under the WFD using the Ecological Quality Ratios (EQR) of different elements of biology and physico-chemistry. The calculation of an EQR requires the observed value and the value that would be expected for that element under reference conditions. In the case of AWB/ HMWBs the term Ecological Potential replaces Ecological Status, while Maximum Ecological Potential (MEP) is the reference state against which observed values are assessed. For biological quality elements this is defined as being:

> *"The values of the relevant biological quality elements (that) reflect, as far as possible, those associated with the closest comparable surface water body type, given the physical conditions which result from the artificial or heavily modified characteristics of the water body"*

and for physico-chemical elements as corresponding

> *"totally or nearly totally to the undisturbed conditions associated with the surface water body type most closely comparable to the artificial or heavily modified body concerned"*

Good Ecological Potential (GEP) is the basic objective for AWB/HMWBs such as canals. GEP is considered to constitute a slight change in the values of the biological quality elements and for values of physico-chemical elements not to deviate to the extent that ecosystem functioning is affected. However, as with natural water bodies, there will be circumstances under which a less stringent objective than GEP is appropriate. Such circumstances include the provision of services that cannot be provided in other ways without disproportionate cost.

Canals are subject to a variety of external and internal pressures. The external pressures are common to many natural lowland water bodies and include nutrient enrichment from diffuse and point sources, water abstraction, and contamination of sediments by waterside industries. The internal pressures relate to the management of canals as navigations, and in some cases, as recreational fisheries. How these internal pressures are interpreted is critical to the definition of MEP. Several approaches to this issue were considered in Phase 1, viz;

- Comparison with other natural water bodies, such as lowland rivers
- Comparison with canals not carrying boat traffic
- Comparison with the best sites at a given level of use

Of these approaches the first two both present significant problems. Thus, it is difficult to find a satisfactory natural analogue for canals since the primary objective is to retain, rather than to convey water, and canals shift from pond-like environments at very low traffic to large river-like environments as traffic increases. The second option is an improvement in the sense that it allows for the specific physical attributes of canals but it essentially penalises the purpose for which canals were constructed by using an absence of navigation as the reference point. Within the third option boat traffic *per se* becomes an integral characteristic of the system and is therefore considered to be a source of permitted environmental variation, in the same way as intrinsic factors such as slope, depth and alkalinity in natural rivers or lakes. With the approval of DEFRA this is the approach that has been used to define MEP for canals because it makes explicit allowance for the impacts associated with a given level of the permitted use. Equally this definition does not exempt navigation from being a cause of failure to meet MEP because there are a range of activities associated with the use and 'servicing' of navigations that will regulate the impact of boat traffic on the aquatic environment.

2 INVERTEBRATE-BASED CLASSIFICATION OF CANALS

2.1 Sampling protocol

2.1.1 Sample collection

Samples are collected from the bank edge using a standard 1000µm mesh size pond net on a 1.5m pole. Samples are obtained based on a timed 3 minute sweep of aquatic and emergent vegetation and sediment within an area accessible from the canal bank and by walking along a ~10m section of channel.

Samples can be collected from either bank, although it is assumed that the towpath (i.e. nearside) bank will be used by default.

This method of sample collection will work in the vast majority of cases. However, if the channel is very deep (>1m) immediately adjacent to the bank, or the substrate is very coarse, this method is liable to be inefficient. In such cases a Naturalists dredge (305mm x 100mm, fitted with a 1000µm mesh bag) that can be thrown into the channel from the bank should be used instead, or to complement the sample obtained with a net.

2.1.2 Timing and frequency of sampling

Samples should be collected in spring (March 1st to May 31st). At the present time it is recommended that sampling is carried out twice in each six year monitoring cycle, with samples separated by approximately three years. Water bodies that have been disturbed by major maintenance activities, such as dredging or dewatering, must be allowed sufficient time to recover before sampling is undertaken. Sampling should therefore not be undertaken less than 2 years after this level of disturbance. If weed cutting is to be carried out sampling should be undertaken prior to this or postponed until the year following weed cutting.

2.1.3 Where and how many sites to sample

The invertebrate sampling protocol can be used to assess the quality of any section of canal, including both reinforced and natural bank sections. Specifically, the sampling area should be a representative section of the canal and should be as homogenous as possible. The channel type must be constant over the length of the site that is assessed. It is important not to seek out only vegetated sections (the only exception to this being on Scottish highland canals where sampling of the naturally sparse vegetation may be needed in order to obtain an adequate sample), or lengths with soft banks, if these are not typical of the canal. The hydromorphological assessment, in conjunction with maps, GIS and Google Earth aerial imagery should be used to judge the representativeness of potential sites.

Sampling sites should not be situated within 100m of operational features such as locks, moorings, wharves or marina entrances. Samples should be collected from lengths in which a single channel type is present, and not, for example, along a length where the channel type changes from a natural trapezoidal to hard - engineered box shaped profile. Heavily poached banks should be avoided unless they are a typical feature of the overall water body. Unless it would be

unrepresentative to avoid them samples should not be obtained immediately downstream of potential point sources of pollution.

It will be beneficial if invertebrate sampling is carried out at the same locations used for macrophyte surveys. This is not essential but it is likely to be logistically easier, will maximize the value of sampling and will ease the interpretation of results. Note, however, that macrophyte surveys must be carried out during the summer (1 June to 31 September) which means that invertebrate sampling and macrophyte surveys cannot be undertaken simultaneously.

The number of sites to sample in a water body will depend on water body length. The number of samples per water body should range from one-three, with a density of approximately one sample site per 10km of channel. Where multiple samples are obtained from a water body sites should be located in different pounds, or roughly evenly spaced throughout the water body. If water body length is very small (<1km) and/or composed largely of operational features, consideration could be given to deriving the status of invertebrates from the average of samples obtained from upstream and downstream water bodies.

2.1.4 Sample processing

Samples collected in the field can be stored unpreserved at 5-8^{o}C and sorted live within 24 to 48 hours of collection or preserved at base using ethanol and/or formaldehyde. Sub-sampling is a legitimate method to deal with large volume samples, for example, from densely vegetated sites.

Samples should be hand-sorted to remove animals and identified to family level (excluding Chironomidae and Oligochaeta) using appropriate identification keys. The currently available sample data contains representatives from 81 scoring families. Note that for quality control and audit purposes all taxa should be recorded, even if they are 'non-scoring' taxa for the purposes of metric calculation. Commonly observed families in samples from canals, which do not contribute to values of biological metrics, include Hydracarinae (mites), several common families of dipteran larvae (Ceratopogonidae, Psychodidae and Stratiomyidae), and various micro-crustaceans (ostracods, cladocerans and copepods).

Animals should be enumerated and their numbers recorded according to $\log_{10}$ abundance categories (0 = absent; 1-9 individuals = 1; 10-99 individuals = 2; 100-999 individuals = 3).

2.1.5 Collection of environmental data

In order to relate the observed fauna at a site to the fauna that would be expected at a comparable site in the absence of anthropogenic disturbance a basic subset of environmental data is required. Some of this data can be obtained by reference to other monitoring networks but other information must be acquired by direct observation at the time of biological sampling. The required information is;

- Alkalinity, as mg/L $CaCO_3$, based on the long term annual mean of the closest monitoring site within the same water body. Alkalinity is commonly measured as apart of a suite of chemical determinands in

routine chemistry monitoring. If measured alkalinity is not available it should be estimated from conductivity (as µS/cm) using the function

Alkalinity = 10^(2.8109*Ln(Conductivity)-0.6533).

For the purposes of this tool the alkalinity value used is the mean measured alkalinity for the water body. Alkalinity in lowland water bodies may be influenced may a number of factors including agricultural fertiliser input and discharges from STWs. For the purposes of this tool it is assumed that all alkalinity values for canals, whether reference sites or not, are 'influenced' rather than truly natural but are equally not 'grossly distorted'. Thus, alkalinity values should not be taken from routine monitoring directly downstream of point sources and values used should be verified as being typical of the overall waterbody. If necessary, this may require spot sampling from a number of discrete locations.

- If conductivity is also not available then alkalinity must be estimated from the values applicable to the nearest major feeder.
- Boat traffic, as passages per year. This data is available from British Waterways or other operators in terms of traffic per waterway length. The value used must be appropriate to the site in question and should not be an average of the entire canal or associated waterbody.
- Channel type, taking account of profile and bank material. Channel type is defined as either 'Box shaped', where both banks are hard engineered, 'asymmetric', where one bank (usually towpath side or nearside) is hard engineered and the other (offside) is soft and sloping, or 'Trapezoidal', where both banks are soft and sloping. Most canal sites can be accommodated easily within this scheme. Possible exceptions are: (i) sites with a trapezoidal channel profile where banks are composed of riprap or rock. These should be recorded as box shaped. (ii) sites where one or either bank is not hard engineered but is composed of vertical natural rock. These should be recorded as box shaped or asymmetric according to the above scheme, even if covered with terrestrial vegetation. (iii) sites where one or both banks are vertical but are composed of soft eroding material. These should be recorded as asymmetric or trapezoidal according to the above scheme.
- Substrate particle size based on a visual assessment, and expressed as a percentage. Percentages should be given for each of the following size classes; Boulders/cobbles (phi -7.75), pebbles/gravel (phi -3.25), sand (phi 2.0) and silt/clay (phi 8.0). If the substrate surface is not visible from the bank then the substrate must be characterised based on the material collected in a net during invertebrate sampling, or from a substrate sample obtained independently using a grab or dredge.
- Altitude, expressed in metres. This can be obtained in the field using a handheld GPS, or retrospectively by reference to 1:50000 scale Ordnance Survey maps.

2.1.6 Data storage

Data should be stored in an electronic format, such as a Microsoft Access database or Microsoft Excel spreadsheet, or within an appropriate national agency recording system (e.g. BIOSys). An Excel spreadsheet is provided which will calculate the metrics required in classification.

2.2 Calculating metrics used in classification

2.2.1 Screening of metrics

The PSYM project (Biggs et al., 2001) and WFD61a (Willby et al., 2008) identified a number of candidate metrics for use in reflecting a range of pressures on canal biota. A subset of metrics is used here that show reasonably good relationships with measured environmental data, and appear to offer complementary sensitivity to different types of pressure. Expansion of the available data to include sites in Scotland and Ireland has expanded the range of potential metrics from that identified in Phase 1. Of the metrics discussed below ASPT and N_EPT can be considered to be particularly indicative of water quality pressures, while N_FAM, N_MOLL+ODON and N_COL+BUGS are more reflective of the quality and diversity of physical habitats and are therefore better aligned with hydro-morphological pressures. The sensitivity of the various metrics to different pressures, or indicators of pressures, is summarised in Table 2.1. Samples obtained and classified using the CPET technique (Green, 2009), were cross matched in GIS to the macroinvertebrate sample sites and their associated environmental data. The pressure sensitivity of CPET (based on the reported EQI rather than the raw metric) is shown alongside the other metrics for comparative purposes.

Table 2.1. Relationships between invertebrate metrics used in classification and a suite of pressure variables, or indicators of pressures, relevant to canals in GB & Ireland.

Pressure/Indicator	ASPT	N_EPT	CPET	N_FAM	N_COL+ BUG	N_MOLL+ ODON
Ammonia	- -	-	-			+ +
BOD	- -	-			-	+
Chloride		-	- -	-		- -
Copper		-		-		-
Nitrate		- -	- -		-	+
Orthophosphate	+		- -	-		+
Zinc	-					+
Bank reinforcement		-		- -	- -	-
Boat traffic			- -	- -	- -	-
Macrophyte cover		-		+ +		+
Susp Solids			-		-	

Key: ASPT = Average Score Per Taxon, N_EPT = Number of Ephemeroptera, Plecoptera and Trichoptera families, N_FAM = Total number of scoring families, N_COL+BUGS = Number of Coleoptera and Hemiptera families, N_MOLL+ODON = Number of Mollusca and Odonata families.

Table 2.2. Indicative values for invertebrate metrics based on the mean (upper line) and inter quartile range (lower line) of all available spring samples from canals in England & Wales, Scotland and Ireland.

Country	N_FAM	ASPT	N_EPT	N_COL+ BUG	N_MOLL+ ODON
England & Wales	13.0	4.0	1.0	1.0	4.0
	8 - 18	3.6 - 4.5	0 - 3	0 - 2	2 - 6
Scotland	17.5	4.1	2.0	3.0	6.0
	12.5 - 20	3.9 - 4.5	1 - 3	2 - 4	1.5 - 7
Ireland	13.0	4.3	2.0	4.0	4.0
	10 - 16	4.1 - 4.5	1 - 2	2 - 4.5	3 - 5

2.2.2 Number of families (N_FAM)

N_FAM simply represents a count of the total number of scoring taxa in a sample (based on family level or coarser taxonomic level). Scoring taxa are listed in Table 2.3.

The N_FAM metric is weakly responsive to several water quality parameters but is highly sensitive to boat traffic and channel type and increases with macrophyte cover.

Table 2.3. Full list of scoring families, revised BMWP scores for calculating revised ASPT and membership of EPT, COL+BUG and MOLL+ODON groups for calculating additional richness metrics.

Family	Common name	rev BMWP	EPT	COL+ BUG	MOLL+ ODON	Family	Common name	rev BMWP	EPT	COL+ BUG	MOLL+ ODON
Planariidae	Flatworms	4.2				Cordulegasteridae	Dragonflies	8.6			*
Dendrocoelidae	Flatworms	3.1				Aeshnidae	Dragonflies	6.1			*
Neritidae	Snails	7.5			*	Corduliidae	Dragonflies	8†			*
Viviparidae	Snails	6.3			*	Libellulidae	Dragonflies	5			*
Valvatidae	Snails	2.8			*	Hydrometridae	Bugs	5.3		*	
Hydrobiidae	Snails	3.9			*	Gerridae	Bugs	4.7		*	
Lymnaeidae	Snails	3			*	Nepidae	Bugs	4.3		*	
Physidae	Snails	1.8			*	Naucoridae	Bugs	4.3		*	
Planorbidae	Snails	2.9			*	Aphelocheiridae	Bugs	8.9		*	
Ancylidae	Limpets	5.6			*	Notonectidae	Bugs	3.8		*	
Unionidae	Mussels	5.2			*	Pleidae	Bugs	3.9		*	
Sphaeriidae	Mussels	3.6			*	Corixidae	Bugs	3.7		*	
Oligochaeta	Worms	3.5				Haliplidae	Beetles	4		*	
Piscicolidae	Leeches	5				Hygrobiidae	Beetles	2.6		*	
Glossiphoniidae	Leeches	3.1				Dytiscidae	Beetles	4.8		*	
Hirudididae	Leeches	0				Gyrinidae	Beetles	7.8		*	
Erpobdellidae	Leeches	2.8				Hydrophilidae	Beetles	5.1		*	
Asellidae	Crustaceans	2.1				Scirtidae	Beetles	6.5		*	
Corophiidae	Crustaceans	6.1				Dryopidae	Beetles	6.5		*	
Gammaridae *	Crustaceans	4.5				Elmidae	Beetles	6.4		*	
Astacidae	Crustaceans	9				Sialidae	Alderflies	4.5			
Siphlonuridae	Mayflies	11	*			Rhyacophilidae	Caddisflies	8.3	*		
Baetidae	Mayflies	5.3	*			Philopotamidae	Caddisflies	10.6	*		
Heptageniidae	Mayflies	9.8	*			Polycentropidae	Caddisflies	8.6	*		
Leptophlebiidae	Mayflies	8.9	*			Psychomyiidae	Caddisflies	6.9	*		
Ephemerellidae	Mayflies	7.7	*			Hydropsychidae	Caddisflies	6.6	*		
Potamanthidae	Mayflies	7.6	*			Hydroptilidae	Caddisflies	6.7	*		
Ephemeridae	Mayflies	9.3	*			Phryganeidae	Caddisflies	7	*		
Caenidae	Mayflies	7.1	*			Limnephilidae	Caddisflies	6.9	*		
Taeniopterygidae	Stoneflies	10.8	*			Molannidae	Caddisflies	8.9	*		
Nemouridae	Stoneflies	9.1	*			Beraeidae	Caddisflies	9	*		
Leuctridae	Stoneflies	9.9	*			Odontoceridae	Caddisflies	10.9	*		
Capniidae	Stoneflies	10	*			Leptoceridae	Caddisflies	7.8	*		
Perlodidae	Stoneflies	10.7	*			Goeridae	Caddisflies	9.9	*		
Perlidae	Stoneflies	12.5	*			Lepidostomatidae	Caddisflies	10.4	*		
Chloroperlidae	Stoneflies	12.4	*			Brachycentridae	Caddisflies	9.4	*		
Platycnemidae	Damselflies	5.1			*	Sericostomatidae	Caddisflies	9.2	*		
Coenagriidae	Damselflies	3.5			*	Tipulidae	Trueflies	5.5			
Lestidae	Damselflies	5.4			*	Chironomidae	Trueflies	3.7			
Calopterygidae	Damselflies	6.4			*	Simuliidae	Trueflies	5.8			
Gomphidae	Dragonflies	8†			*						

* including Crangonyctidae † Based on original BMWP score

2.2.3 Average Score Per Taxon (ASPT)

ASPT is the Average Score Per Taxon where the taxon scores (Table 2.3) are based on the published revision of the original BMWP scores by Walley & Hawkes (2001). In this case ASPT is calculated based on a numerically weighted average using the $\log_{10}$ abundance categories to provide the weighting. Several alternative sets of scores were trialled including the original BMWP scores and new scores specific to either riffle or pools (Walley & Hawkes, 2001) but the generic revised BMWP scores produced the best correlations with measured pressure data.

ASPT displays clear sensitivity to ammonia and BOD and therefore clearly merits a role in the classification of ecological status on canals due to its potential to indicate organic pollution. It is insensitive to a range of pressures associated with the operation and design of canals.

2.2.4 Number of Ephemeroptera, Plecoptera and Trichoptera families (N_EPT)

N_EPT is the number of families of Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies) in a standard sample. This metric is commonly used in conjunction with ASPT to improve detection of water quality pressures in rivers. Qualifying families are listed in Table 2.3.

The present screening illustrates that N_EPT is weakly sensitive to various water quality pressures but is particularly responsive to nitrate concentrations.

2.2.5 Number of Mollusc and Odonata families (N_MOLL+ODON)

N_MOLL+ODON is the number of Mollusc (snail, limpet and mussel) families plus the number of Odonata (dragonfly and damselfly) families present in a sample. Qualifying families are listed in Table 2.3.

The screening shows that this is not a suitable metric to use for assessment of water quality (although there is some sensitivity to heavy metals) because snails in particular are tolerant of low dissolved oxygen concentrations and are often favoured by high fertility. However, both these groups are strongly dependent on macrophytes for habitat structure and consequently are sensitive to pressures that reduce macrophyte cover or eliminate marginal habitat.

This metric was originally conceived as two discrete metrics reflecting the number of mollusc families or the number of Odonata families. However, the two metrics were found to be well correlated. They also had limited independent value since the observed number of taxa in either group was often very small which led to high variability in metric values at reference sites. Consequently these metrics were merged.

2.2.6 Number of Coleoptera and Hemiptera families (N_COL+BUGS)

This metric described the number of Coleoptera (beetle) families plus the number of Hemiptera (bug) families in a sample. Qualifying families are listed in Table 2.3.

There is little evidence that this metric is sensitive to water quality-related pressures but it is clearly responsive to boat traffic and channel type. This reflects the importance of emergent vegetation in linking terrestrial and aquatic habitats and in providing hunting, concealment, and oviposition sites for Coleoptera and Hemiptera.

As with N_MOLL+ODON this metric was originally conceived as two discrete metrics but both were well correlated. Both sub metrics were shown in the WFD61a report to have potential for reflecting pressures on marginal habitats but in that case the variability of values for these metrics at reference sites was too high for the metrics to be used effectively in classification. Now that data from Scotland and Ireland has been incorporated into this study and existing data has been reassessed to determine channel type it has proved possible to utilise N_COL+BUGS in macroinvertebrate-based assessment.

2.2.7 CPET

The Chironomid Pupal Exuviae Technique (CPET) was developed as part of a sister project (Green, 2009). The CPET method requires a different sampling technique (skimming floating debris from the water surface with a fine net and aggregating three samples collected between April – September), and the identification of pupa skins to species or genus level. The distribution of species appears to be especially sensitive to Total Oxidised Nitrogen (TON) and the optima of the different species along this gradient therefore forms the basis of the metric that is applied to canals (Green, 2009). A weighted average of the mean, based on the proportion of different species in a sample, provides the observed metric value.

Integration of CPET into the project database indicates that it is less sensitive to ammonia and BOD than ASPT but is especially responsive to dissolved inorganic nutrients. It is also sensitive to boat traffic.

2.2.8 Data quality and sampling efficiency

In this system of classification an Ecological Quality Index (EQI) is calculated for each metric based on the ratio of Observed: Expected value. The method will take no account of data quality and will simply classify the data with which it is provided.

Attention should therefore be given to the quality of samples submitted for classification. Inspection of the available biological data indicates that samples containing ≤ 6 taxa in which no taxa are represented by >9 individuals are extremely abnormal and should be treated as suspect for the purposes of classification. Such samples should preferably be discarded and repeated unless it is obvious from the sample volume (i.e. amount of sediment or aquatic vegetation) that the sample is of a normal size.

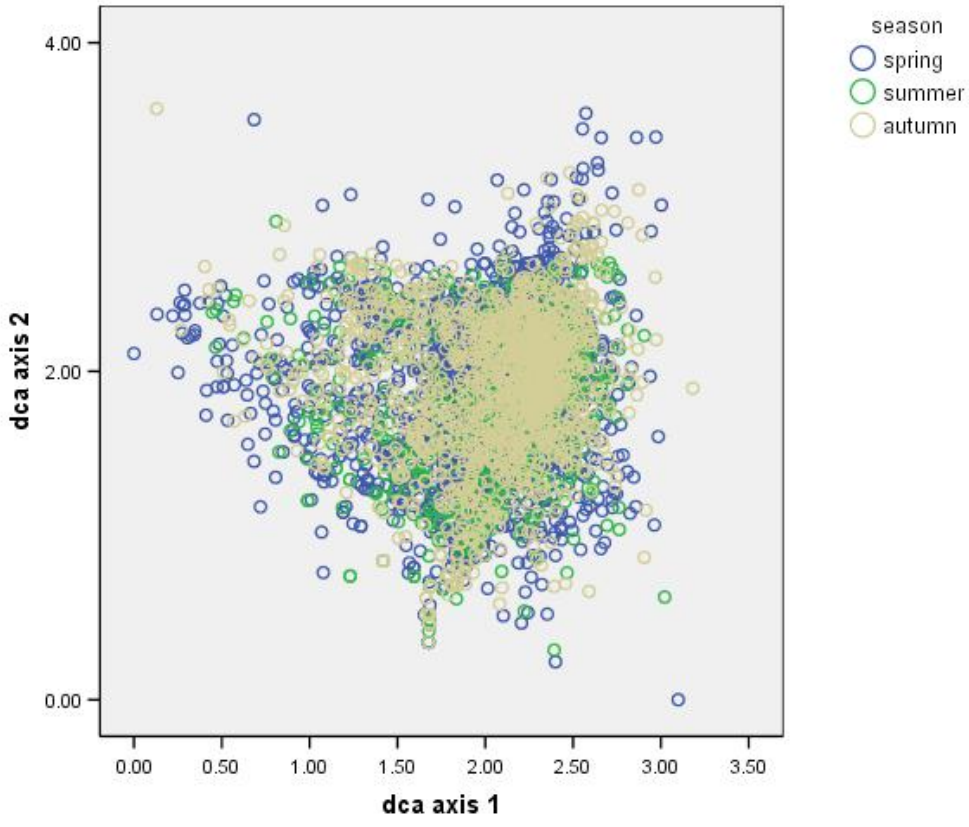
2.3 Identifying reference sites

2.3.1 Data availability

Invertebrate family level data was available for 4100 samples collected on canals between 1990 and 2006. The vast majority of data (95%) was from canals in England & Wales, with compliant data having been obtained from canals in Scotland and Ireland, specifically for this project, in 2006.

The majority of samples (2100) had been obtained in spring (01 March – 31 May). Spring samples were available across the widest geographical extent of sites and range of conditions encountered on canals in GB and Ireland. An analysis in Phase 1 raised the possibility of using single-season data for invertebrate assessment given the apparent lack of seasonality in canal invertebrate samples. The integration of data from Scotland and Ireland collected in spring and autumn 2006 showed that this pattern was retained (Figure 2.1). The use of spring only data also enabled the integration of data collected from canals in spring 1997 as part of the PSYM project. The decision was therefore made to restrict classification based on invertebrates to spring samples.

Figure 2.1. Ordination based on Detrended Correspondence Analysis (dca) of family level invertebrate data showing distribution of samples in full database according to season of collection



Sites that had been sampled in spring in England & Wales were plotted in GIS to confirm that they were situated on a canal itself. At this stage 452 samples from 39 sites that had been retained in Phase 1 were excluded as the samples derived from canal feeders, adjacent streams or navigable rivers. These are likely to have a different fauna that is not representative of canals.

2.3.2 Environmental data availability GIS etc

Sampling sites were located in a GIS which was used to cross match biological data with environmental data collected at the same or other sites within a water body. A nearest neighbour matching procedure was used to resolve the nearest spatial match. This was especially effective in resolving links between invertebrate sampling sites and those used for CPET, macrophytes or routine chemistry sampling.

Biological sample sites were also linked to spatial data provided by BW on boat traffic, bank type and condition from towpath surveys, location of linear mooring sites and embankments and hydrocompliance status.

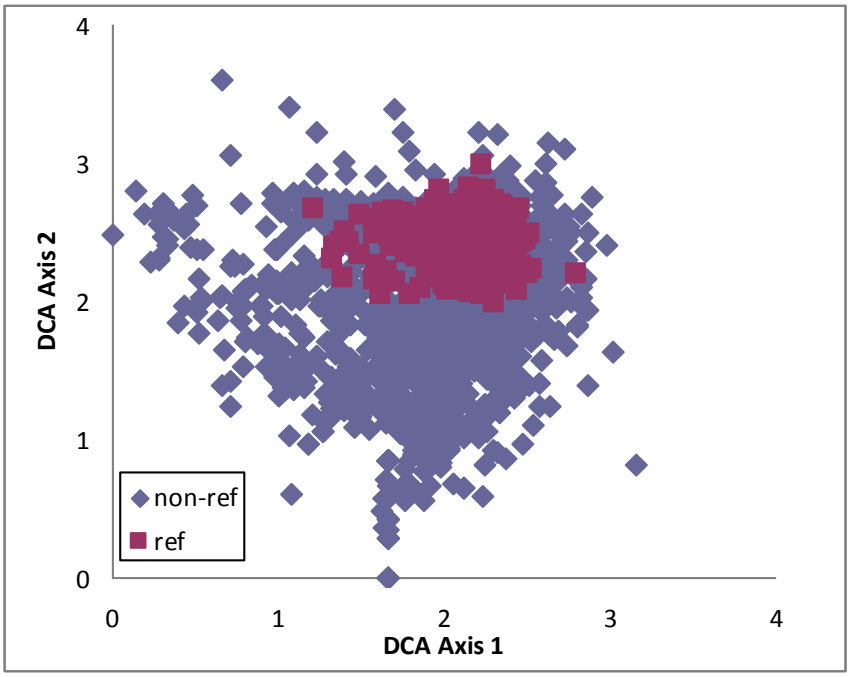
Data on channel type was not available for Ireland and was therefore derived from an inventory of site photographs in conjunction with Google Earth.

2.3.3 Data screening

To screen out potential reference sites the lower 25th percentile of ASPT and N_FAM was calculated for the samples available from each country. A subset of sites from each country was then created where the values for <u>both</u> metrics exceeded the lower quartile of their distribution. This approach ensured that the potential reference sites were of generally good biological quality and did not contain deviant data where, for example, high ASPT values occurred in association with low numbers of taxa, or vice versa. Analysing the data on a country-by-country basis ensured that high thresholds for a metric in one country could not preclude sites in a different country from being in reference condition. For example, average taxon richness was 25% higher on unimpacted canals in England & Wales than in Ireland. A threshold derived from the global dataset (which is dominated by samples from England & Wales) would have automatically excluded naturally less diverse sites in Ireland from being potential reference sites.

The pool of sites of good biological quality was then filtered using matched abiotic data relevant to water quality pressures on canals (organic pollution, eutrophication, heavy metal pollution). The match between sites with biological data to sites with routine chemical monitoring was relatively good (50-60% of samples can be matched to measured data for the most well monitored variables) but only a relatively small number of determinands were consistently measured. Thus, proper screening could only be accomplished in relation to ammonia, BOD, TON, SRP, chloride, suspended solids, and the metals copper and zinc. The objective of filtering by abiotic data was to obtain a pool of minimally impacted sites with high biological quality of low variability. This was obtained by plotting potential reference and non-reference sites in ordination space and then systematically removing outliers that could be accounted for by elevated values of abiotic variables (Figure 2.2).

Figure 2.2. Ordination based on Detrended Correspondence Analysis (DCA) of family level invertebrate data showing distribution of invertebrate reference (ref) and non-reference (non-ref) sites after screening by biology and pressure data.



The remaining pool of sites were also checked against the smaller body of data available for other metals, including mercury, cadmium, chromium and lead, to confirm that concentrations were not elevated above background at these sites. Sites were also checked in GIS to ensure that they were not located in close proximity (<500m) to moorings or known marinas (since these are likely to elevate disturbance above that associated with the reported traffic figures) and they did not fail Hydrocompliance standards according to data provided by BW. In this case, boat traffic is considered a 'permitted disturbance', since it represents the basic function of canals and we are seeking to establish ecological potential, taking account of the reasons for which a water body was designed or modified. Therefore reference sites were not specifically screened in relation to their boat traffic. Similarly, under certain circumstances, channel engineering to support navigation is an expected feature of canals. Thus, potential reference sites with hard engineering on both banks were not specifically excluded but were treated as described in 2.4.1 to determine if hard engineering was an expected feature of the combination of conditions at that site. The result is a set of biologically coherent sites in which concentrations of the main stressors can be considered to be within the low impact range for navigable canals (Table 2.4).

The final population of canal reference sites comprised data from 165 sites of which 127 were in England & Wales, 20 in Ireland and 18 in Scotland. When spring sample data was available from multiple years at a site (England & Wales only) the year with values of ASPT and N_FAM closest to the mean value of all spring samples from that site was used.

Table 2.4. Annual mean values of selected pressure-indicating variables (median and lower or upper percentiles) in reference sites.

	Ammonia –N	BOD	Chloride	Conduc - tivity	Copper	Total Oxidised Nitrogen	Soluble Reactive Phosphor -us	Dissolved Oxygen saturation	pH	Suspe -nded Solids	Zinc
units	mg/L	mg/L	mg/L	uScm	mg/L	mg/L	mg/L	%		mg/L	mg/L
median	0.059	2.00	30	331	0.003	1.26	0.053	88	7.70	9.68	0.042
90th %tile	0.094	2.83	67	572	0.027	3.64	0.104	78†	7.98	53.13	0.126

† Expressed as 10th percentile

2.4 Calculating expected values for invertebrate metrics

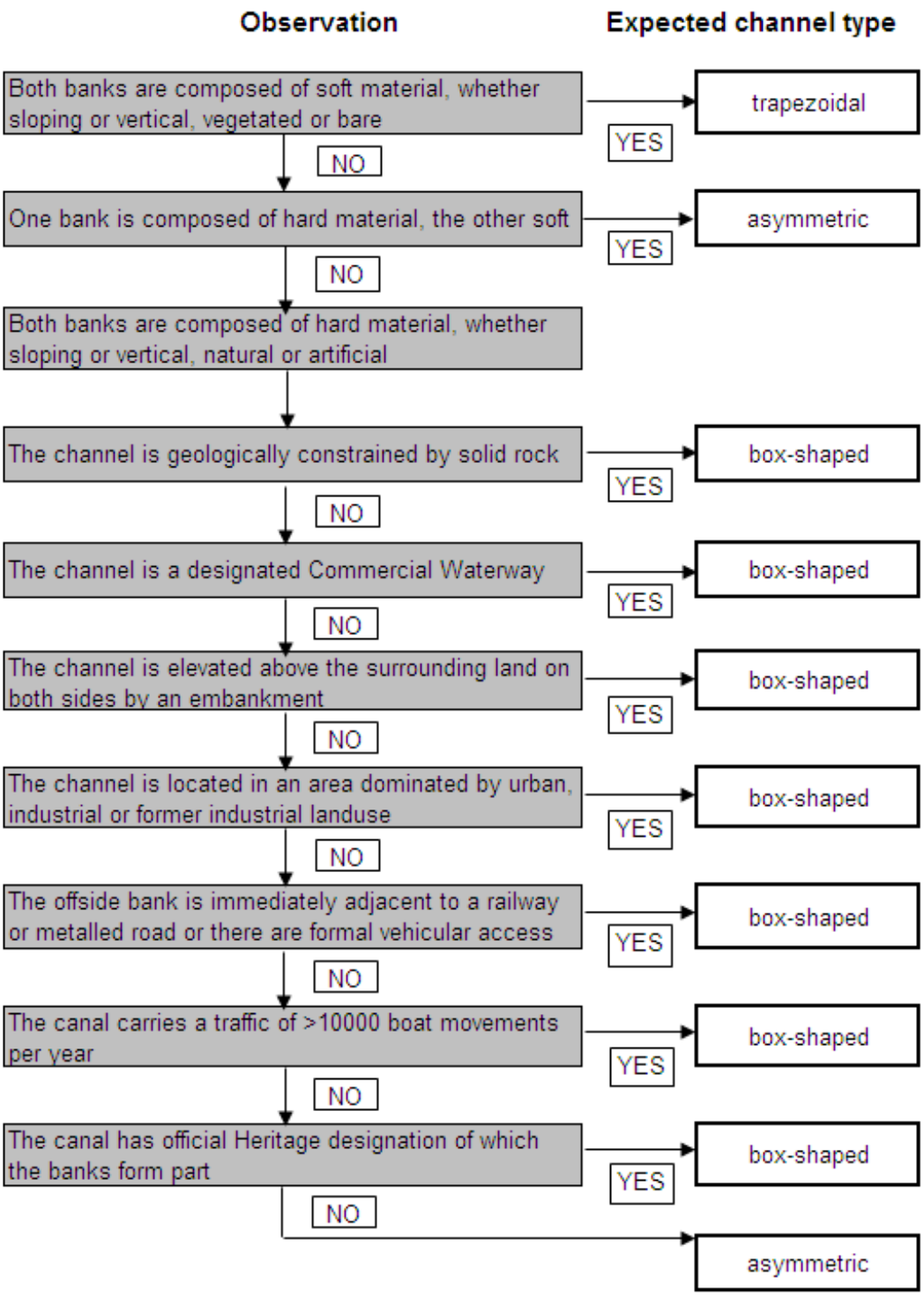
2.4.1 Data requirements

In order to model the expected values for the various invertebrate metrics at the reference sites biological data was linked to environmental data covering boat traffic density, channel type (reflecting the availability of marginal habitat), alkalinity (as an indicator of underlying geology), altitude and substrate (expressed as an estimate of the proportion of the sediment composed of sand, or silt-clay respectively). The purpose of these models is to determine the expected value of a biological metric in the absence of anthropogenic pressure.

With the exception of channel type, the values used are taken at face value and are therefore as recorded in the field, or derived from separate monitoring. Channel type presents a special case because the observed channel type may reflect a level of hydro-morphological impact beyond that appropriate to the specific setting. Bank engineering is likely to significantly compromise lateral connectivity between aquatic and marginal habitats unless effectively mitigated. Taking the observed channel type, without first checking the expected channel type for that site, could result in hydro-morphological pressures being concealed.

Figure 2.3 illustrates how to evaluate the expected channel type at a site. If an asymmetric or trapezoidal channel is recorded in the field no further check is required. If a box-shaped channel (as defined in 2.1.5) is recorded it is necessary to check if this is the expected channel type in that setting and in the absence of hydro-morphological impacts. The flow chart therefore provides a number of questions to determine if a box-shaped channel can be justified under the conditions at the site. In essence, the argument would be that it is difficult to explain the presence of an artificial, vertical-sided channel in a non-commercial waterway, with moderate to low boat traffic, in a rural setting, where there are no specific access requirements on the offside bank, the channel is not situated on an embankment and the banks are not geologically constrained or part of a formal built heritage designation. Under such circumstances it is expected that the offside bank should be soft and sloping and would thus default to an asymmetric channel. It is estimated that c.10% of instances where a box-shaped channel is observed in the field should be expected to be an asymmetric channel.

Figure 2.3. Flowchart for determining expected channel type



2.4.2 Expected N_FAM

While there is a considerable amount of unexplained variability, the number of scoring families is clearly sensitive to boat traffic and channel type (Figure 2.4). These variables must therefore be taken into consideration when predicting the numbers of families to be expected at a site in the absence of pressure. Expected values of N_FAM are calculated based on the function in Table 2.6.

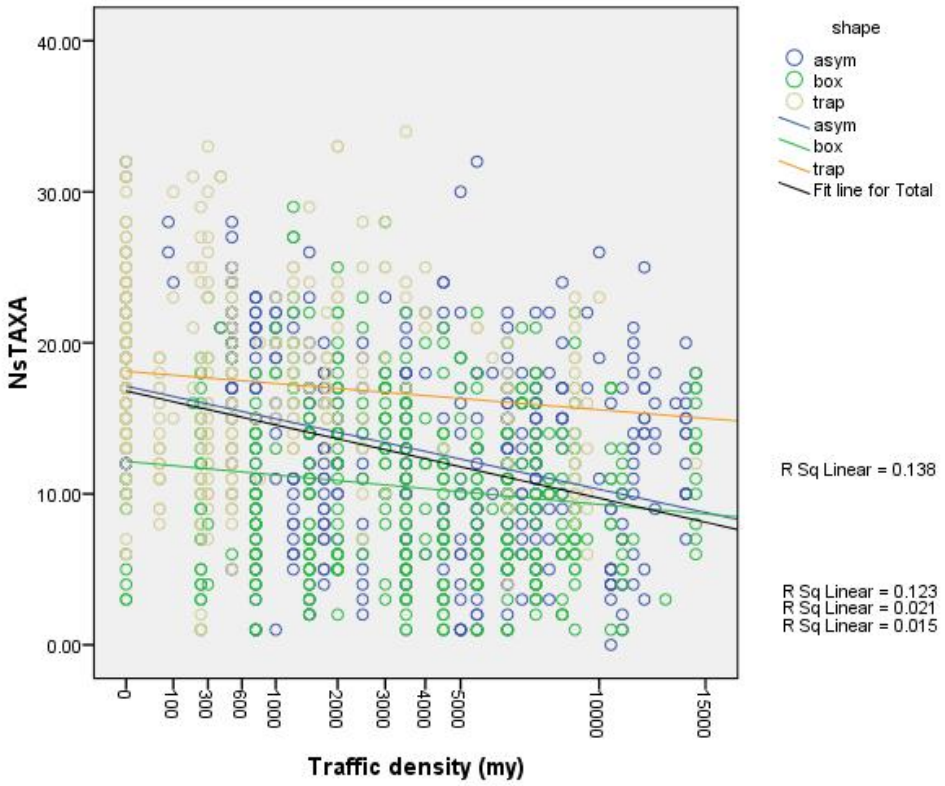
However, there is a further complication in modelling expected numbers of families because the median number of scoring families at reference sites differs strongly between some countries or regions in this project (Table 2.5). This almost certainly reflects biogeographical factors, possibly allied to major differences in water chemistry. It should be noted that canals in Ireland and Scotland *per se* are, on average, much more diverse than canals in England & Wales since the latter contain a significantly higher proportion of lower quality sites. However, within reference sites, canals in England & Wales are more taxon-rich.

These national or regional differences in richness can be accommodated but a further problem is that the range of conditions (in terms of combinations of traffic and channel type) covered by canals outside of England & Wales is highly constrained and it would be unwise to base models on the very limited amount of available data. To resolve this, the model developed for England & Wales is applied to all countries and regions because it is assumed that invertebrates will respond in a predictable manner to traffic and bank type, regardless of where they are. A correction factor based on Table 2.5 is then applied to the expected value. Thus, for example, in Ireland the expected number of families would be 76% of the number expected in England & Wales under the same conditions of boat traffic and channel type. For the Scottish lowland canals only a small correction is required (93%), but in the Scottish highland canals the correction is much larger (65%). This reflects the lower alkalinity, coarse substrate and sparse aquatic vegetation associated with Scottish highland canals.

Table 2.5. Median family richness in reference sites in different countries/ regions, and appropriate correction factors for calculating expected values for the metric N_FAM.

	England & Wales	Ireland	Scotland lowland	Scotland highland
median N_FAM	21	16	19.5	13.6
correction factor	1	0.76	0.93	0.65

Figure 2.4. Relationship between the number of scoring taxa and boat traffic in three different channel types



Asym= asymmetric or semi-trapezoidal; box = box shaped (i.e. both banks hard engineered); trap= trapezoidal

2.4.3 Expected ASPT

ASPT is primarily sensitive to alkalinity and substrate (Table 2.6). Its sensitivity to boat traffic within the population of reference sites reflects the slight enhancement of ASPT values at higher boat traffic due to the more river-like conditions that then occur in canals which favour some more oxygen-demanding taxa. By contrast, in densely vegetated channels with low boat traffic those taxa tolerant of sluggish conditions with lower oxygen concentrations naturally tend to dominate. ASPT at reference sites is insensitive to channel type (Figure 2.5).

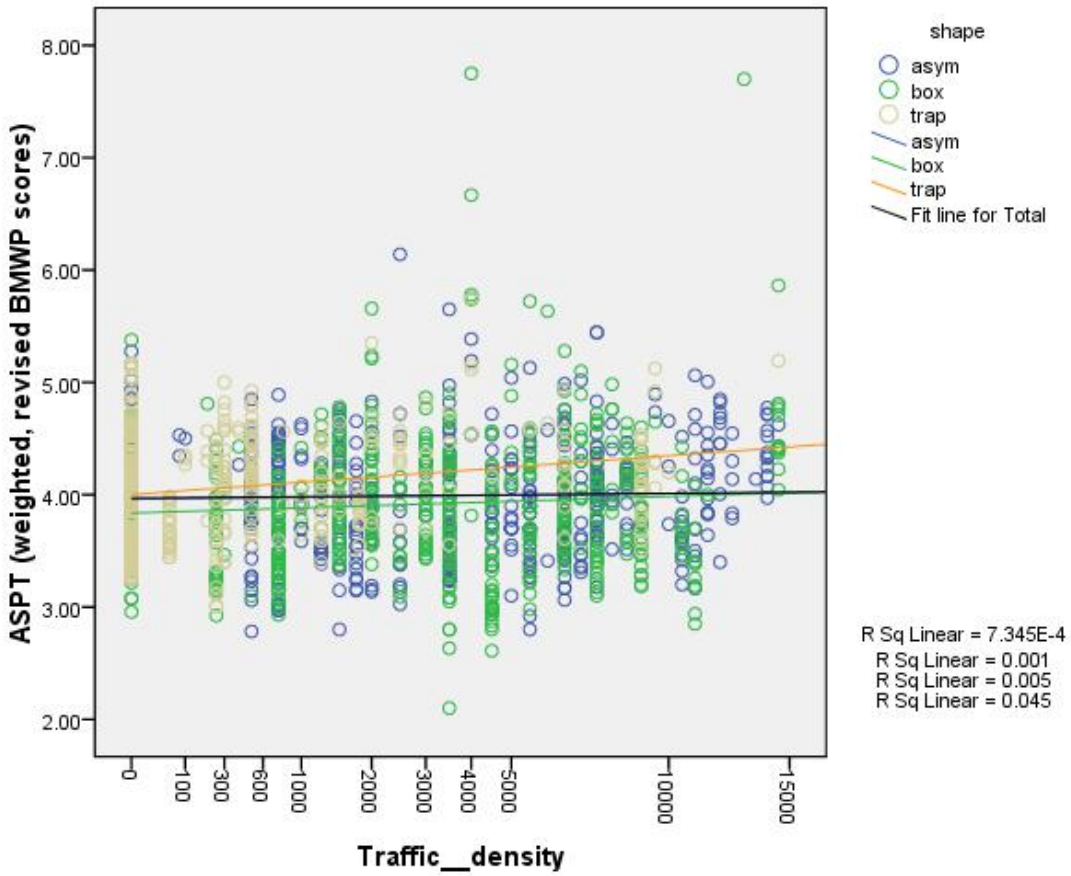
Two models can be used to predict ASPT at reference sites. The preferred model includes the visual assessment of the proportion of silt-clay in the substrate as a predictor. ASPT can still be predicted without substrate data (if the substrate is not visible or cannot easily be sampled) but with much greater uncertainty. The ASPT values based on the revised BMWP system (Walley & Hawkes, 1997) could not be modelled quite as effectively as those based on the system of original scores but the revised version of BMWP was retained to maintain compatibility with the River Invertebrate Classification Tool (RICT). Expected ASPT values normally range between 4.1 – 4.5. For guidance, values calculated using the revised BMWP scores tend to be fractionally lower (~5%) for canal samples than the equivalent scores based on the original BMWP system.

Table 2.6. Functions and coefficients for modeling expected values of macroinvertebrate metrics.

Metric	Conditions	r^2	Constant	Traffic (my)	$\log_{10}$ (traffic+1)	Shape†	Alkalinity (mg/L $CaCO_3$)	$\log_{10}$ (alkalinity)	Altitude (m)	%Silt-clay
N_FAM	apply country/region corrections	23.1	14.62	-0.001	1.07	2.128				
ASPT	preferred model	28.6	5.072	0.0000278				-0.243		-0.005
ASPT	if no substrate data available	15.4	5.701	0.0000255			0.002	-0.83		
EPT	preferred model	12.8	4.151							-0.018
EPT	if no substrate data available		3.00							
COL+BUGS	no correction required	27.7	1.409	-0.0002		0.868			0.012	
MOLL+ODON	apply country/region corrections	13.2	2.257					1.999		

Shape†: where 1 = box shaped; 2 = asymmetric; 3 = trapezoidal

Figure 2.5. Relationship between ASPT and boat traffic in three different channel types



Asym= asymmetric or semi-trapezoidal; box = box shaped (i.e. both banks hard engineered); trap= trapezoidal

2.4.4 Expected N_EPT

This metric is difficult to model since it appears to be responsive only to substrate characteristics (Table 2.6). If substrate data is available and the proportion of silt-clay is greater than zero N_EPT can be modelled with low precision. The expectation is that there will typically be 2.5 - 4.0 EPT taxa at a site. If no substrate data is available then the expected value of EPT is derived simply from the global median of all values across the 165 reference sites. This provides a value of 3.

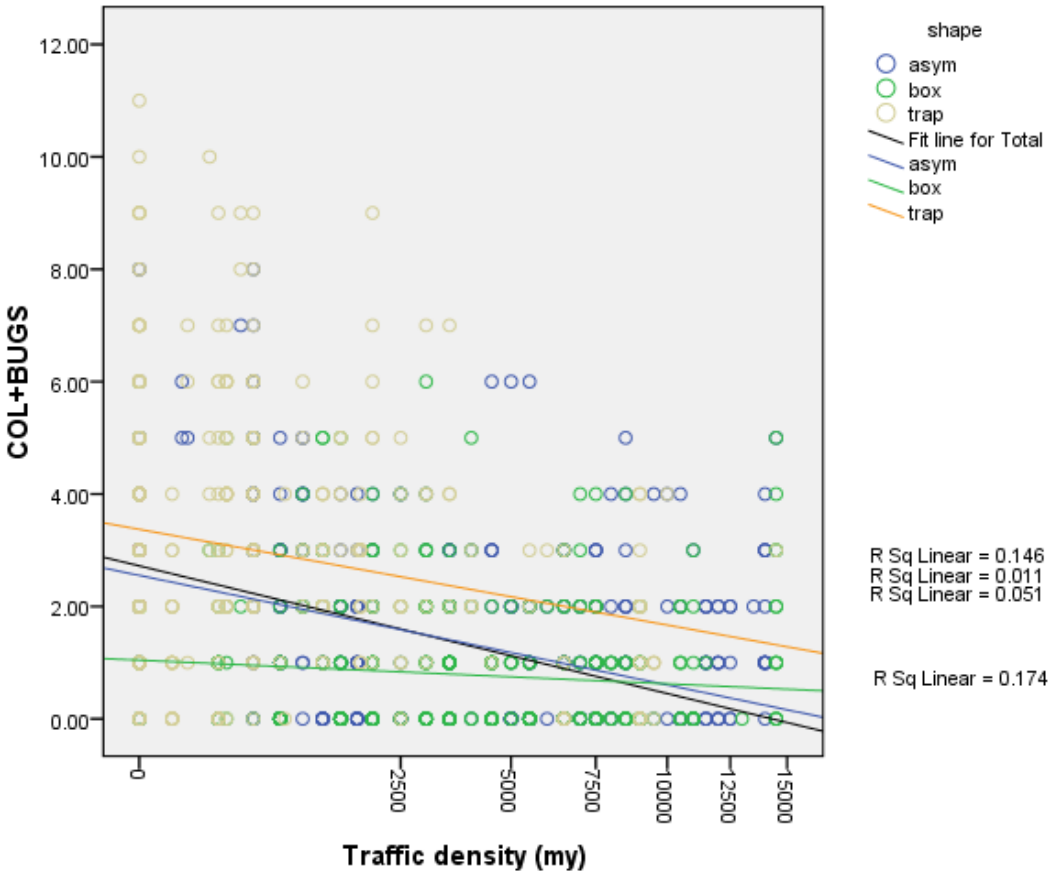
There is no clear evidence in the case of this metric that different geographical groupings of sites support different diversities of EPT taxa so no regional adjustments have been applied to the expected values. Scottish Highland canals often have slightly higher values for this metric which is to be expected given their close proximity to high quality upland streams but the number of sites is too small to allow a reliable adjustment to be made to take account of this.

2.4.5 Expected N_COL+BUGS

This metric is relatively strongly responsive to boat traffic and channel type within the population of reference sites. Altitude is an additional minor predictor (Table 2.6). Sites with soft sloping banks that are likely to support extensive emergent vegetation support larger numbers of Coleoptera and Hemiptera (bugs) at almost all levels of traffic and both groups are very poorly represented at sites with hard engineered banks (Figure 2.6).

There is no evidence in the case of this metric that different geographical groupings of sites support different diversities of Coleoptera or Hemiptera so no regional adjustments have been applied to the expected values.

Figure 2.6. Relationship between the N_COL+BUGS and boat traffic in three different channel types



Asym= asymmetric or semi-trapezoidal; box = box shaped (i.e. both banks hard engineered); trap= trapezoidal

2.4.6 Expected N_MOLL+ODON

Within reference sites this metric only varies clearly in relation to alkalinity (Table 2.6). The use of alkalinity therefore allows this metric to be modelled with low precision.

There is clear evidence in the case of this metric that different geographical groupings of sites support different diversities of Mollusca and Odonata. Reference sites in Ireland supported 65% of the numbers of Mollusca and Odonata compared to sites in England & Wales. This feature is puzzling given the very high alkalinity of Irish canals and their plentiful aquatic vegetation and may reflect the low mobility of molluscs. Geographical isolation coupled with relatively frequent weed cutting may also be a factor. Nominally there was no difference in the number of Mollusca and Odonata at reference sites in lowland Scotland but the number of such sites is comparatively small so the same adjustment was applied in the case of lowland Scotland as was used for the metric N_FAM (93%). In the case of the Scottish highland canals a more significant adjustment was required since these sites only supported about 15% of the Mollusca and Odonata as sites in England & Wales. This is almost certainly a reflection of the low alkalinity and sparse aquatic vegetation characteristic of highland Scottish canals.

2.5 Calculating an Ecological Quality Index (EQI) for invertebrate metrics

2.5.1 N_FAM

For this metric:

EQI_{N_FAM} = Observed N_FAM / Expected N_FAM

Where the Observed N_FAM is derived from a sample collected between 1 March and 31 May according to the prescribed method and represents only scoring families in Table 2.3, and Expected N_FAM is calculated according to the equation in Table 2.6, and adjusted for the country/region in which the site is located.

2.5.2 ASPT

For this metric:

EQI_{ASPT} = (Observed ASPT) / (Expected ASPT)

Where the Observed ASPT is derived from a sample collected between 1 March and 31 May according to the prescribed method and is calculated using only the scoring families and scores as listed in Table 2.3. Expected ASPT is calculated according to the equation in Table 2.6.

2.5.3 N_EPT

For this metric:

EQI_{N_EPT} = Observed N_EPT / Expected N_EPT

Where the Observed EPT is derived from a sample collected between 1 March and 31 May according to the prescribed method and Expected EPT is calculated according to the equation in Table 2.6.

2.5.4 N_COL_BUGS

For this metric:

$EQI_{N_COL+BUG}$ = Observed N_COL+BUG / Expected N_COL+BUG

Where the Observed N_COL+BUG is derived from a sample collected between 1 March and 31 May according to the prescribed method and Expected COL+BUG is calculated according to the equation in Table 2.6.

2.5.5 N_MOLL_ODON

For this metric:

$EQI_{N_MOLL+ODON}$ =Observed N_MOLL+ODON / Expected N_MOLL+ODON

Where the Observed N_MOLL+ODON is derived from a sample collected between 1 March and 31 May according to the prescribed method and Expected N_MOLL+ODON is calculated according to the equation in Table 2.6, and adjusted for the country/region in which the site is located.

2.6 Combining EQIs to achieve an invertebrate-based classification

2.6.1 Establishing class boundaries

In this project class boundaries for the different metrics have been established statistically by examining the distribution of EQI values for the various metrics in samples from reference sites. The advantages of this approach is that (i) the class boundaries reflect the natural variability found in each metric in the population of reference sites, without setting boundaries that are unrealistically stringent or lax in relation to this variability, and (ii) the same boundary setting protocol is applied to all metrics. The major disadvantage is that a class boundary has little ecological basis. Other approaches have been employed, for example, using the relative proportions of pressure sensitive and tolerant taxa to set a particular boundary but even the rules underlying this type of approach are largely arbitrary. In reality consensus regarding placement of boundaries between Maximum/Good or Good/Moderate Ecological Potential can only be established through the process of intercalibration of national assessment methods between Member States.

In this project the reference dataset is relatively small, canals in general are relatively homogenous (as opposed to the difference between, say, small upland streams and large lowland rivers), and richness-based metrics (all those used here except ASPT) tend to be inherently rather variable and more difficult to model effectively. This means that there is a large amount of unexplained variation in the metrics within the population of reference sites and class widths must be correspondingly large enough to reflect this. The variation in EQI values within reference sites is reflected in the wide spread of the histograms in Figure 2.7.

In this project the boundary between Maximum and Good Ecological Potential was based on the lower 20^{th} percentile of the distribution of EQI values in reference sites. This is a relatively precautionary approach (normally the lower 5^{th} or 10^{th} percentile is used) but it ensures that all metrics can be subdivided into at least three classes. Subsequent class boundaries are then based on the width of the Maximum Ecological Potential class. The resulting boundaries are reviewed in Table 2.7. It should be noted that for two metrics, EPT and N_COL+BUG, the variation in reference sites was sufficiently high to dictate that only three quality classes could be recognised based on these metrics, while for MOLL+ODON only four classes could be defined. This reflects the fact that a relatively small number of taxa contribute to observed values of these metrics meaning that the variation in the EQI is naturally high.

Figure 2.7. Histograms showing frequency distribution of EQI values for different metrics in a population of 165 reference sites.

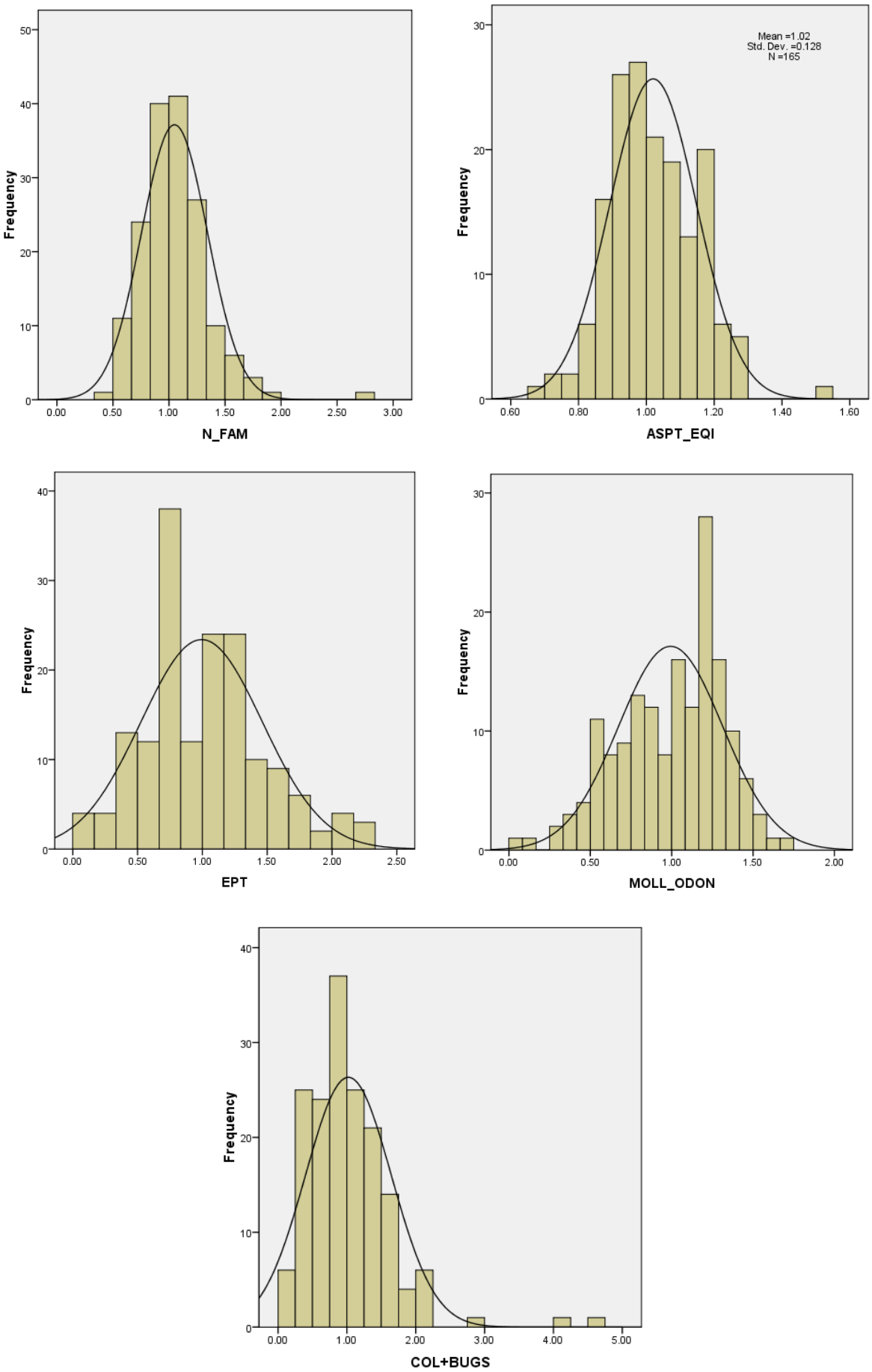


Table 2.7. Class boundaries for different levels of Ecological Potential in five macroinvertebrate metric EQIs.

EP boundary	Max/Good	Good/Mod	Mod/Poor	Poor/Bad
N_FAM	0.80	0.60	0.40	0.20
ASPT	0.92	0.84	0.76	0.68
N_EPT	0.66	0.33		
N_COL+BUG	0.66	0.33		
N_MOLL+ODON	0.75	0.50	0.25	

2.6.2 Rescaling EQI values to an EQR

In multimetric approaches such as the one described here metrics need to be combined to provide an overall classification for a site. Before this is possible it is essential to map all the metrics onto a common class boundary system. A class width of 0.2 is normally used. Hence the MEP/GEP boundary in each metric is then mapped to a value of 0.8. This approach requires a 'piecewise linear transformation' of each metric from its original class boundaries so that the values of all metrics can be compared directly. As an example, consider that for the ASPT metric the MEP/GEP and GEP/ModEP boundaries lie at 0.92 and 0.84 respectively. Thus, a site with an EQI of 0.88 lies exactly in the middle of Good status, or, on a 0.2 unit system, half way between 0.6 and 0.8 (i.e. 0.7). In the metric N_MOLL+ODON, if the MEP/GEP and GEP/ModEP boundaries lie at 0.75 and 0.5 respectively, a value of 0.63 lies exactly in the middle of Good status, or, on a 0.2 unit system, again half way between 0.6 and 0.8 (i.e. 0.7). Thus, despite the raw EQI values differing, they are identical on a transformed scale. Conversely, a site with an EQI of 0.88 in each metric would be classified as Good EP, according to the ASPT class boundaries, but Maximum EP according to the N_MOLL+ODON class boundaries.

The piecewise linear transformation ensures the conversion of all EQI values to a standardised set of EQRs ranging from 0 to 1 with class boundaries at intervals of 0.2 (Table 2.8).

Table 2.8. Ecological Potential boundaries, after implementation of piecewise linear transformation to standardise EQI values to an EQR scaling.

	Max	Max/Good	Good/Mod	Mod/Poor	Poor/Bad	Min
N_FAM	1.0	0.8	0.6	0.4	0.2	0.0
ASPT	1.0	0.8	0.6	0.4	0.2	0.0
EPT	1.0	0.8	0.6			0.0
COL+BUG	1.0	0.8	0.6			0.0
MOLL+ODON	1.0	0.8	0.6	0.4		0.0

2.6.3 Rules for combining EQR values

A series of potential rules for combining EQR values for individual metrics were considered. These included

1. the average of all metrics
2. a weighted average of all metrics, with ASPT and N_FAM receiving more weight
3. the minimum of all metrics
4. the average of the lowest two or lowest three metrics
5. the average of different 'families' of metrics
6. a rule-based approach taking the average of the minima of different families of metrics
7. a rule-based approach taking a weighted average of a group of water quality metrics and a group of hydromorphological metrics, followed by taking the lowest EQR of these two groups.

The effect of using different approaches to combine EQR values on the overall EQR for the site is illustrated in Table 2.9. The average of all metrics is clearly the most relaxed approach, whilst the minimum of all is the most precautionary. Averaging multiple metrics when it is not possible to express all metrics equally in terms of five classes is probably not wise. General averaging also ignores the fact that different metrics have inherently higher variability and therefore should probably carry less weight as they are less reliable indicators. In this study the metrics N_EPT, N_COL+BUG and N_MOLL+ODON were therefore arbitrarily assigned half the weight of the metrics ASPT and N_FAM.

Based on an analysis of relationships between different quality elements, common sense, and advice from the project steering group the chosen approach for combining metrics was to calculate a weighted average of the combination of metrics from each of the two groups of metrics reflecting water quality (ASPT, EPT) and hydro-morphological (N_FAM, N_MOLL+ODON, N_COL+BUGS) pressures and to take the minima of these two averages as the EQR for the site.

Table 2.9. Proportion of samples assigned to different classes of Ecological Potential based on different rules for combining metrics

combination option	Max	Good	Mod	Poor	Bad
average of all	37.8	25.8	16.3	14.5	5.5
wtd average of all	39.6	26.3	16.8	12.9	4.4
average of lowest 3 metrics	18.9	23.9	14.2	14.2	28.7
average of lowest 2 metrics	16.6	24.2	10.8	17.8	30.7
minimum of all	12.7	22.3	14.0	1.4	49.5
average of WQ/HM mins	20.7	24.1	11.9	16.9	26.3
minima of wtd WQ/HM avs	26.4	25.0	15.5	20.6	12.5

2.6.4 Implications for national classifications

Using the scheme described for combining metrics to produce an overall classification it is possible to determine the proportion of sites in different countries that will be classified at different levels of Ecological Potential. Table 2.10 provides this data. Scotland is split into lowland and highland canals due to their marked differences. The data for Scotland and Ireland should be treated as indicative since it is based on only a single year of assessment. The results for England & Wales are usually based on the average of assessments made in 3-5 years at each site and should therefore provide a reliable picture. The most directly comparable sites are probably those in England & Wales and lowland Scotland. The quality of the latter is readily evident. Classifications of the Scottish highland canals should be treated with caution due to the potential problems of sampling efficiency discussed earlier. As a general picture 38% of sites on canals in GB and Ireland are likely to fall below Good Ecological Potential, of which 90% are England & Wales.

Table 2.10. Numbers and proportions of sites in different countries/regions classified at differing levels of Ecological Potential according to a combination rule based on the minimum of either 'water quality' or 'hydro-morphological' groups of metrics.

Country		Max	Good	Moderate	Poor	Bad
England & Wales	n	96	112	67	61	21
	%	26.9	31.4	18.8	17.1	5.9
Ireland	n	18†	13	7	3	0
	%	43.9	31.7	17.1	7.3	0.0
Scotland - lowland	n	15	13	2	0	0
	%	50.0	43.3	6.7	0.0	0.0
Scotland - highland	n	3	4	2	2	0*
	%	27.3	36.4	18.2	18.2	0.0

* Three sites classified at Bad status were excluded because samples contained either no animals or only a single species.

† Includes two sites in Northern Ireland (Newry Canal at Carnmeen and Coalisland Canal at Anaghbeg)

A detailed example is given below concerning the classification of sites on Irish canal water bodies. A total of thirty nine sites were assessed belonging to different water bodies. Of these, three water bodies were small and therefore assessed based on a single site. Figure 2.8 illustrates the invertebrate-based EQR for sites on the Grand and Royal Canal systems, with sites ranked from west to east. The global average EQR of all sites assessed was 0.73 (middle of Good Ecological Potential) reflecting their generally high quality. Sixteen sites were classified at Maximum Ecological Potential (EQR ≥ 0.8), with the Barrow Line of the Grand Canal being especially striking. Thirteen sites were classified at Good Ecological Potential, with only ten sites (26%) being classified below Good Ecological Potential, of which three were classified as Poor status. At a water body level all waterbodies would achieve Good Ecological Potential, based on the

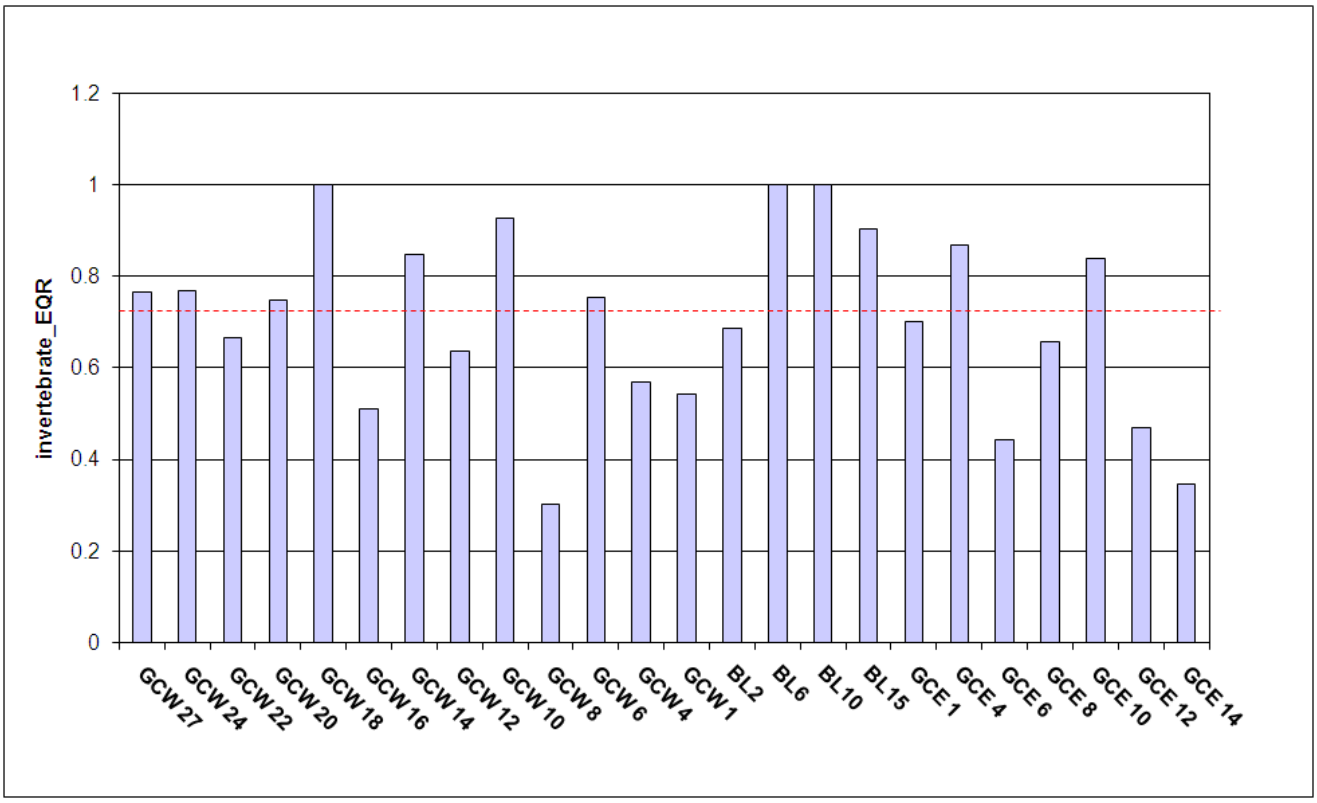
averaging of the EQRs of those sampling sites belonging to each water body, with the exception of one water body on the Grand Canal (SeRBD-GCW) represented by five sampling sites (GCW1, GCW4, GCW8, GCW10 and GCW12) which lies on the boundary between Good and Moderate Ecological Potential (average EQR = 0.59). The first three of these sampling sites exhibit low values for metrics associated with hydro-morphological pressures, suggesting the possibility of recent disturbance in this water body. The most obvious geographical trend from this analysis is for lower quality of sites on both the Grand and Royal canals in the vicinity of Dublin, especially in relation to water quality.

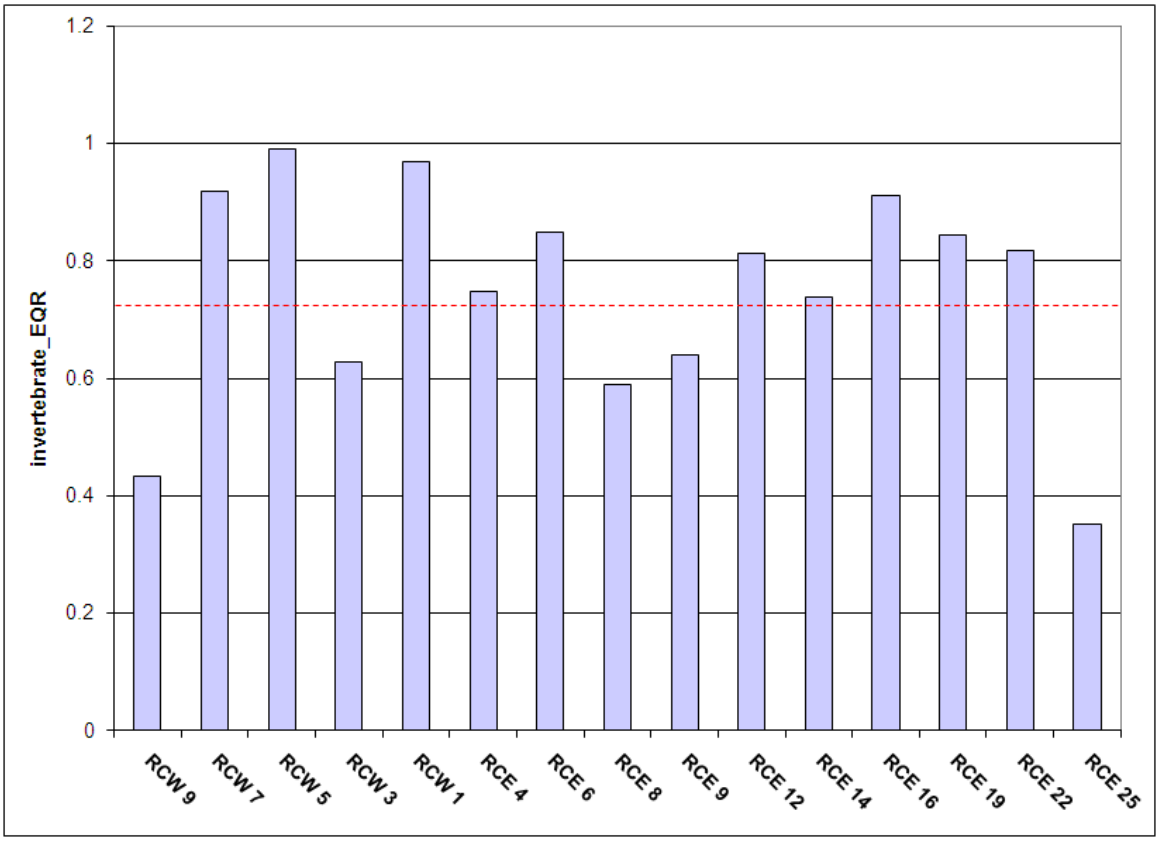
From a summary of site-based classifications, it is also possible to determine the proportion of sites failing to achieve Good Ecological Potential for different reasons. Of the 163 sites in GB and Ireland that fail to achieve GEP, 44% do so due to low values of metrics sensitive to hydro-morphological pressures, 28% due to water quality pressures and 28% due to a combination of both water quality and hydro-morphological pressures.

Table 2.11. Proportion of times different groups of metrics dictated the classification of a subset of 106 sites from GB, showing the effect of integration of CPET on the overall assessment.

Major pressure	without CPET	with CPET
Water quality	24.5	40.6
Hydromorphology	31.1	21.7
Both	44.3	37.7

Figure 2.8. Classification of sites on Irish Canals based on macroinvertebrates. Upper panel Grand Canal, including Barrow Line (BL), lower panel Royal Canal. Red dashed line is the global mean value across all sites assessed. Classifications based on samples collected spring 2006 by Central Fisheries Board.





2.7 Integrating CPET-based classifications

It is relatively straightforward to integrate the results of CPET-based classifications into the present classification format. The pressure-based screening of metrics indicates that the principle sensitivity of CPET is to water quality. Therefore, rather than take the weighted average of the two macroinvertebrate metrics focused on water quality (ASPT and EPT) one would take the weighted average across the minimum of either ASPT or CPET, and EPT, whenever CPET was available, giving equal weight to CPET and ASPT.

Thus, without CPET, the formula is:

$$WQ = (EQR_{ASPT} + 0.5 \times EQR_{N_EPT}) / 1.5$$

When CPET is available this becomes

$$WQ = (\min (EQR_{CPET}, EQR_{ASPT}) + 0.5 \times EQR_{N_EPT}) / 1.5$$

By integrating the CPET results into the present classification it is possible to determine the relative value of different metrics in classification, and to infer the relative importance of different types of pressures on canals.

By linking the CPET and macroinvertebrate sample sites in GIS a subset of 106 canal sites in GB was obtained where CPET and macroinvertebrate samples were collected from the same water body, within 1km of each other. No CPET data was available for Ireland. The effect of integrating CPET was then assessed by comparing the proportion of occasions when water quality, hydromorphology or both pressures would determine the classification of a site (Table 2.11). The inclusion of CPET clearly increases sensitivity to water quality related pressures, increasing the frequency of occasions where this is the major pressure at a site and reducing the incidence that hydromorphology is the main pressure.

Unfortunately it is difficult to compare CPET directly with the other metrics since the CPET classification was derived using an independent reference dataset of only seven sites and CPET may therefore be intrinsically more precautionary. However, the incorporation of CPET probably serves to increase sensitivity to nutrient-related pressures since ASPT is more sensitive to organic pollution. Looking at the relative importance of hydro-morphological versus water quality pressures, it is clear that no one group clearly dominates and that in many cases both pressures are probably acting simultaneously.

2.8 Determining the EQR for a water body

The above details will enable an EQR to be calculated for a single site based on a single sample. If the waterbody is only assessed at a single location, and has been sampled only once in a monitoring cycle the EQR calculated will represent the EQR for the water body as a whole. If a water body is sampled at multiple locations on a single ‘date’ (i.e. within the same season in the same year), the average EQR for all sites sampled should be determined to given an EQR for the water body. If the water body is sampled more than once in a monitoring cycle, its overall EQR is the average of the separate waterbody EQRs calculated for each year of sampling.

2.9 Worked example

2.9.1 Raw data

The data used in this example was collected by SEPA in spring 2006 from the Forth & Clyde Canal at Lambhill which is located in the north of Glasgow. The site in question (Figure 2.9) lies at the edge of an industrial area and borders a Local Nature Reserve. Both banks are soft and well vegetated and the channel type is therefore trapezoidal. Boat traffic is light (500 my). The environmental data pertaining to this site is shown in Table 2.12.

2.9.2 Calculating observed metric values

The raw biological data for this site is shown in Table 2.13.
The value for N_FAM is simply determined by counting the scoring taxa, in this case 18. For ASPT the metric is a weighted average of the ASPT values given in Table 2.13 column 3, where the $\log_{10}$ abundance of each taxon provides the weighting factor. Hence the value of ASPT for the site is:

(1 * 4.8 + 1 * 4.0 + 1 * 3.7 + 1 * 7.8……2 * 3.7 + 1 * 3.5) /
(1 + 1 + 1 + 1……2 + 1) = 3.93

In the case of N_EPT only two scoring taxa are present, Leptoceridae and Limnephilidae. Therefore N_EPT = 2.

For N_COL+BUG there are three scoring taxa, Dytiscidae, Haliplidae and Corixidae. Thus N_COL+BUG = 3.

For N_MOLL+ODON there are seven scoring taxa, six molluscs and the damselfly, Coenagrionidae. Thus N_MOLL+ODON = 7.

Figure 2.9. Map details and image for site used in worked example, Lambhill, Glasgow, Forth & Clyde Canal.



Table 2.12. Environmental data for case study site at Lambhill, Glasgow, Forth & Clyde Canal.

	SEPA19
Canal Name	Forth and Clyde
Site Name	Lambhill
NGR	NS5863469702
Site code	
SEPA location code	121436
Channel type	Trapezoidal
Traffic (my)	500
Alkalinity (mg/L $CaCO_3$)	97.2
Altitude (m)	54
% Sand	0
% Silt-clay	50

Table 2.13. Raw biological data for case study site at Lambhill, Glasgow, Forth & Clyde Canal, as sampled by SEPA in spring 2006.

Family	**log_{10} abundance**	**ASPT**	**EPT**	**COL + BUG**	**MOLL + ODON**
Dytiscidae	1	4.8		*	
Haliplidae	1	4.0		*	
Corixidae	1	3.7		*	
Leptoceridae	1	7.8	*		
Limnephilidae	1	6.9	*		
Asellidae	2	2.1			
Gammaridae	2	4.5			
Coenagrionidae	1	3.5			*
Planariidae	1	4.2			
Erpobdellidae	1	2.8			
Ancylidae	1	5.6			*
Sphaeriidae	1	3.6			*
Lymnaeidae	1	3.0			*
Physidae	1	1.8			*
Planorbidae	1	2.9			*
Hydrobiidae	2	3.9			*
Chironomidae	2	3.7			
Oligochaeta	1	3.5			
Metric value		**3.93**	**2**	**3**	**7**

2.9.3 Calculating Expected metric values

The expected values for each of the five metrics are calculated using the functions in Table 2.6. Values are summarized in Table 2.14.

Thus, for N_FAM

Expected N_FAM = 14.62 + -0.001 * Boat traffic + 2.128 * Shape + 1.07 * $\log_{10}$ (Boat traffic +1)

Where shape = 1 if box shaped, 2 if asymmetric and 3 if trapezoidal.

= 14.62 + -0.001 * 500 + 2.128 * 3 + 1.07 * 2.7 = 23.4.

Then applying the correction of 0.93 for Scotland (Table 2.5) gives the value 21.76.

Thus for ASPT

Expected ASPT = 5.072 + 0.0000278 * Boat traffic + -0.243 * $\log_{10}$ (Alkalinity) +-0.005 * % silt-clay

= 5.072 + 0.0000278 * 500 + -0.243 * 1.99 + -0.005 * 50 = 4.35

Expected values for other metrics are given in Table 2.14.

Table 2.14. Summary of Observed and Expected metric values, EQI and EQR values and overall Ecological Potential for case study site at Lambhill, Glasgow, Forth & Clyde Canal.

Invertebrate metrics	Observed	Expected	EQI	EQR
N_FAM	18	21.76	0.83	0.83
ASPT	3.93	4.35	0.90	0.76
EPT	2	3.25	0.62	0.77
N_COL+BUG	3	4.58	0.65	0.80
N_MOLL+ODON	7	5.8	1.21	1.00
1. wtd average of WQ metrics				0.76
2. wtd average of HM metrics				0.85
minimum of 1 and 2				0.76
Ecological Potential				Good

2.9.4 Calculating an EQI for biological metrics

For each metric the EQI is calculated simply by dividing the Observed value by the Expected value. Thus;

EQI N_FAM = 18 / 21.76 = 0.83

EQI ASPT = 3.93 / 4.35 = 0.90

EQI N_EPT = 2 / 3.25 = 0.62

EQI N_COL+BUG = 3 / 4.58 = 0.65

EQI N_MOLL+ODON = 7 / 5.8 = 1.21

Values are summarised in Table 2.14

2.9.5 Converting the EQI to an EQR

Before these metrics can be combined they must be converted into an EQR where the class widths of all metrics are the same. This means adopting a class system where each of the five possible classes of Ecological Potential occupies 0.2 units of the EQ gradient.

For the metric N_FAM no change is needed to the EQI since the boundary between Maximum and Good Ecological Potential falls at 0.8 and all subsequent classes are therefore 0.2 units apart (Table 2.8).

For the metric ASPT the class boundaries are at 0.92, 0.84, 0.76 and 0.68, i.e. the class width on the EQI scale is 0.08. This width requires to be 'stretched' to 0.2 units so that the ASPT EQI is directly comparable to the N_FAM EQI. Thus, we apply the following formula for a piecewise linear transformation:

=IF(EQI $_{ASPT}$>=1,1,
IF(EQI $_{ASPT}$>=0.92,((EQI $_{ASPT}$-0.92)/(1-0.8))*0.2+0.8,
IF(EQI $_{ASPT}$>=0.84,((EQI $_{ASPT}$-0.84)/(0.92-0.84))*0.2+0.6,
IF(EQI $_{ASPT}$>=0.76,((EQI $_{ASPT}$-0.76)/(0.84-0.76))*0.2+0.4,
IF(EQI $_{ASPT}$>=0.68,((EQI $_{ASPT}$-0.68)/(0.76-0.68))*0.2+0.2,
IF(EQI $_{ASPT}$<0.68,(EQI $_{ASPT}$/0.68)*0.2))))))

Thus in the present case:

IF: EQI $_{ASPT}$>=0.84 ((0.90-0.84)/(0.92-0.84))*0.2+0.6 = 0.75.

The same procedure is then applied to the other metrics. Thus, for N_EPT

=IF(EQI $_{N_EPT}$>=1,1,
IF(EQI $_{N_EPT}$ >=0.66,((EQI $_{N_EPT}$ -0.66)/(1-0.66))*0.2+0.8,
IF(EQI $_{N_EPT}$ >=0.33,((EQI $_{N_EPT}$ -0.33)/(0.66-0.33))*0.2+0.6,
IF(EQI $_{N_EPT}$ <0.33,(EQI $_{N_EPT}$ /0.33)*0.2+0.6))))

Values for each metric in EQR terms are summarised in Table 2.14.

2.9.6 Combining EQRs into a value for the site

The individual EQRs shown in Table 2.14 belong to two basic families – those principally indicative of pressures associated with water quality (shaded green in Table 2.14), and those more responsive to hydromorphological pressures (shaded brown in Table 2.14). The classification of the site is determined by the lowest of each of these groups of pressures.

For the water quality component the aggregated EQR is determined by taking the weighted average of EQR_{ASPT} and EQR_{N_EPT} where EQR_{N_EPT} receives half the weight to reflect its inherent higher variability within the population of reference sites. Thus:

$$WQ\ component = (EQR_{ASPT} + 0.5 * EQR_{N_EPT}) / 1.5$$
$$= (0.76 + 0.5 \times 0.77) / 1.5 = 0.76$$

For the hydromorphological component the aggregated EQR is determined by taking the weighted average of EQR_{N_FAM}, $EQR_{COL+BUG}$ and $EQR_{MOLL+ODON}$ where the latter two EQRs receive half the weight to reflect their higher variability within the reference sites. Thus:

$$HM\ component = (EQR_{N_FAM} + 0.5 * ((EQR_{COL+BUG} + EQR_{MOLL+ODON}) / 2)) / 1.5$$
$$= (0.83 + 0.5 * ((0.8 + 1.0)/ 2)) / 1.5 = 0.853$$

Since the value for WQ component (0.76) is lower than the value for the HM component (0.85) the EQR for this site is 0.76. This would place the site towards the upper end of Good Ecological Potential.

3 MACROPHYTE-BASED CLASSIFICATION OF CANALS

3.1 Survey protocol

3.1.1 Survey methodology

Surveys are conducted on sections of canal 150m in length. For reasons of safety and reducing inter-surveyor variability it is desirable if surveyors work in pairs. Surveys can be conducted from either bank, although it is assumed that the towpath bank will be used by default. There is no requirement to enter standing water within this method. A pair of close-focusing binoculars will be beneficial to allow identification of emergent or floating-leaved plants on the far bank.

Within a reach there are 10 equidistant recording points. At each point a standard design of grapnel is thrown and retrieved to establish the biomass of submerged aquatic vegetation. A record is made of the submerged plants recorded and any floating-leaved or emergent plants that are present at that location. The distance between the bank and the outer edge of the reed fringe is then measured or estimated. Information from the ten sampling positions is then pooled to provide a taxa list for the site where aquatic and emergent taxa are scored on a scale of 1-5 according to their percentage cover. A mean value of submerged plant biomass is obtained based on the drained fresh weight of material on a grapnel. The proportion of the channel width occupied by emergent vegetation is derived from calculation of the average width of fringing vegetation and the average channel width over the reach surveyed.

This survey methodology will work in the vast majority of cases. However, if the substrate is very coarse, or the vegetation is very sparse or highly patchily distributed, this method is liable to be inefficient. In these cases a walkover survey with direct visual observation of vegetation wherever present will be required to provide a representative list of taxa present at a site. Use of a double-headed rake instead of a grapnel is recommended when sampling Scottish highland canals.

3.1.2 Timing and frequency of surveying

Surveys should be undertaken in summer (June 1st to September 31st). At the present time it is recommended that surveys are carried out twice in each six year monitoring cycle, with surveys separated by approximately three years. If it can be established that temporal variation in survey data over this time scale is suficiently low to have a small effect on the confidence of classification then survey frequency could be reduced to once per six year cycle. Water bodies that have been disturbed by major maintenance activities, such as dredging or dewatering, must be allowed sufficient time to recover before sampling is undertaken. Surveys should therefore not be undertaken less than 2 years after this level of disturbance. If weed cutting is to be carried out sampling should be undertaken prior to this or postponed until the year following weed cutting.

3.1.3 Where and how many sites to survey

The macrophyte survey protocol can be used to assess the quality of any section of canal, including both reinforced and natural bank sections. Specifically, the survey length should be a representative section of the canal and should be as

homogenous as possible. The channel type must be constant over the length that is assessed. It is important not to seek out only vegetated sections (the only exception to this being on Scottish highland canals where targeted sampling of the sparse vegetation may be needed in order to obtain an adequate sample), or lengths with soft banks, if these are not typical of the canal as a whole. The hydro-morphological assessment, in conjunction with maps, GIS and Google Earth aerial imagery should be used to judge the representativeness of potential sites.

Survey lengths must not be situated within 100m of operational features such as locks, moorings, wharves or marina entrances. Surveys should be conducted in lengths in which a single channel type is present, and not, for example, along a length where the channel type changes from a natural trapezoidal to hard engineered box shaped profile. Heavily poached banks should be avoided unless they are a typical feature of the overall water body. Unless it would be unrepresentative to avoid them (e.g. because they occur at high density) samples should not be obtained immediately downstream of potential point sources of pollution.

For ease of interpretation of results it will beneficial if macrophyte surveys are carried out at the same locations used for invertebrate sampling. Note, however, that timing requirements mean that macrophyte surveys and invertebrate sampling cannot be undertaken simultaneously.

The number of sites to sample in a water body will depend on water body length and environmental heterogeneity within a water body. The number of samples per water body should typically range from one-three, with a density of approximately one sample site per 10km of channel. Where multiple samples are obtained from a water body sites should be located in different pounds, or roughly evenly spaced throughout the water body. If water body length is very small (<1km) and/or composed largely of operational features, consideration could be given to inferring the status of macrophytes from the average of the closest surveys in adjacent upstream and downstream water bodies. In cases where environmental heterogeneity within a water body is higher than average the number of surveys should be increased to cover the range of conditions present. Examples include a marked gradient in alkalinity from one end of a water body to the other caused by different water sources, or contrasting levels of boat traffic for lengths upstream or downstream of a key junction or flight of locks. Normally this situation will only arise in very large water bodies (>30km length).

3.1.4 Collection of environmental data

In order to relate the observed vegetation at a site to the vegetation that would be expected at a comparable site in the absence of anthropogenic disturbance (excluding navigation) a basic subset of environmental data is required. Some of this data can be obtained by reference to other monitoring networks but other information must be acquired by direct observation at the time of biological sampling. The required information is;

- Alkalinity, as mg/L $CaCO_3$, based on the long term annual mean of the closest monitoring site within the same water body. Alkalinity is commonly measured as part of a suite of chemical determinands in routine chemistry monitoring. If measured alkalinity is not available it should be estimated from conductivity (as µS/cm) using the function

Alkalinity = 10^(2.8109*Ln(Conductivity)-0.6533)

- If conductivity is also not available then alkalinity must be estimated from the values applicable to the nearest major feeder.
- Boat traffic, as passages per year. This data is available from British Waterways or other operators in terms of traffic per waterway length. The value used must be appropriate to the site in question and should not be an average of the entire canal or associated waterbody. When using boat traffic it is important to be sure that 'through traffic', on which traffic estimates are usually based, does not significantly underestimate local (e.g. day-trip) traffic.
- Channel type, taking account of profile and bank material. Channel type is defined as either 'Box shaped', where both banks are hard engineered, 'asymmetric', where one bank (usually towpath side) is hard engineered and the other is soft and sloping, or 'Trapezoidal', where both banks are soft and sloping. Most canal sites can be accommodated easily within this scheme. Exceptions are detailed in 2.1.4.

3.1.5 Data storage

Data should be stored in a suitable electronic format, such as a Microsoft Access database or Microsoft Excel spreadsheet, or within an appropriate national agency recording system (e.g. BIOSys). An Excel spreadsheet is provided which will calculate the metrics required in classification.

3.2 Calculating metrics used in classification

3.2.1 Screening of metrics

WFD61a identified a number of candidate metrics for use in reflecting a range of pressures on canal macrophytes. Expansion of the available data to include more sites in Scotland and Ireland has not changed the range of preferred metrics to use. Macrophyte survey data was cross matched in GIS to the closest sites with associated environmental data. The metrics used here show reasonably good relationships with measured environmental data, and appear to offer complementary sensitivity to different types of pressure. Of the metrics discussed below MFI is exclusively indicative of water quality pressures. The metrics APB and N_AQUA are weakly correlated with water quality pressures but are clearly sensitive to within-channel pressures associated with boat traffic and related pressures, such as suspended solids concentrations. The metrics N_EMERG and REL_EMERG show some sensitivity to various aspects of water quality (especially chlorophyll concentrations) but are primarily affected by the extent of bank reinforcement. The sensitivity of the various metrics to different pressures, or indicators of pressures, is summarised in Table 3.1.

3.2.2 Aquatic Plant Biomass (APB)

Aquatic Plant Biomass, or APB, is a measure of the amount of aquatic plant material with the channel, obtained using a standardised sampling approach. The sampling is based on ten throws with a standard grapnel at equally spaced intervals along the survey length. Material is then drained in the field and its fresh

weight determined using a spring balance. A conversion factor that takes account of the length of transect sampled (as dictated by the length of grapnel rope) is then used to convert mean fresh weight from 10 grapnel throws to mean dry weight per square metre of channel ($gDWm^{-2}$). Since floating-leaved species are likely to be underestimated using this sampling method additional estimates are made using visual estimates of cover combined with standard biomass figures. The approach is described in detail in WFD61a.

The metric APB is especially sensitive to densities of boat traffic and is an effective indicator of within-channel disturbance (Figure 3.1). Higher biomass is generally expected in hard engineered channels at very low traffic presumably because soft banked channels are more vulnerable to encroachment by emergent vegetation when traffic is low.

Table 3.1. Relationships between macrophyte metrics used in classification and a suite of pressure variables, or indicators of pressures, relevant to canals in GB & Ireland.

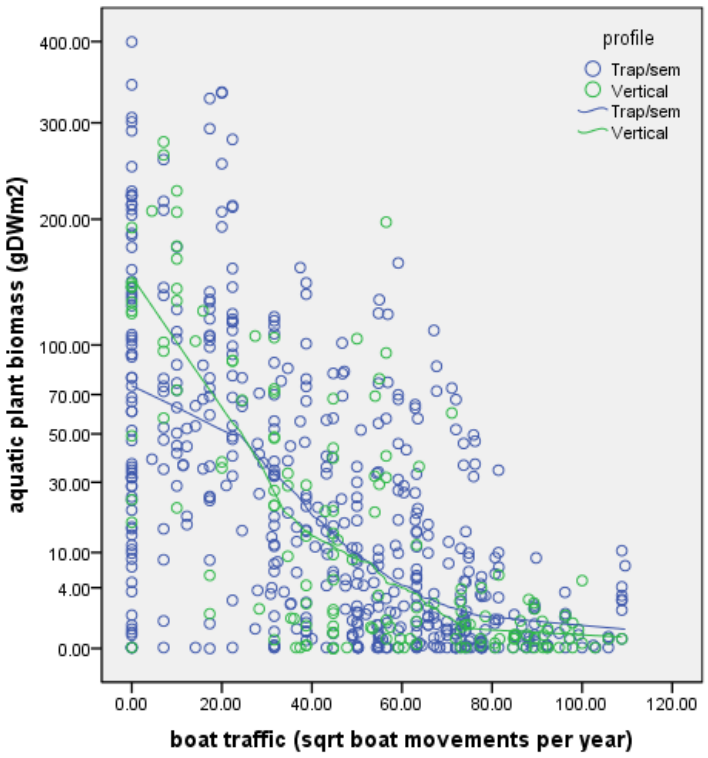
Pressure/Indicator	APB	N_AQUA	MFI	N_EMERG	REL_EMERG
Ammonia					
BOD		-		++	
Chloride					
Chlorophyll		-	- -	- -	- -
Copper	-	-		-	-
Nitrate			+	-	
Orthophosphate		-	+ +	-	
Zinc	-	-		-	-
Bank reinforcement		-		- -	- -
Boat traffic	- -	- -			- -
Suspended Solids	-	- -			

3.2.3 Number of scoring aquatic plant taxa (N_AQUA)

N_AQUA simply represents a count of the total number of scoring aquatic plant taxa in a survey. Scoring taxa are listed in Table 3.2. Although this is not desirable the majority of records of *Chara* have only been determined to genus level in the surveys available and most members of this genus are therefore aggregated for the purposes of calculating this metric. Metrics based on numbers of taxa tend to be subject to greater variability due to differences in surveyor efficiency, expertise and effort. The sampling design should constrain the variability in this metric in surveys of canals because N_AQUA is determined from observations or material harvested from 10 locations at each site.

N_AQUA is weakly sensitive to various water quality parameters but is highly correlated with densities of boat traffic (Figure 3.2) and suspended solids.

Figure 3.1. Relationship between APB and boat traffic in all surveys with respect to channel type.



Trap/sem = Trapezoidal or semi-trapezoidal channel profile, Vertical = box shaped profile

Table 3.2. Full list of scoring macrophyte taxa used in canal assessment.

Species	MFI	habit	status	Species	MFI	habit	status	Species	MFI	habit	status
Acorus calamus	9.49	e	NN	Lemna minor	8.80	a		Riccia sp.	4.86	a	
Alisma lanceolatum	8.47	e		Lemna minuta	9.21	a	NN	Ricciocarpus natans	6.30	a	
Alisma plantago-aquatica	7.82	e		Lemna trisulca	8.21	a		Rorippa amphibia	9.20	e	
Apium inundatum	4.34	a		Littorella uniflora	1.96	a		Rorippa nasturtium-aquaticum	8.42	e	
Apium nodiflorum	8.64	e		Lobelia dortmanna	2.72	a		Rorippa palustris	7.32	e	
Aponogeton distachyos		a	NN	Luronium natans	4.37	a		Rumex hydrolapathum	8.65	e	
Azolla filiculoides	9.71	a	INN	Lysimachia thrysiflora		e		Sagittaria latifolia		a	NN
Baldellia ranunculoides	4.34	a		Lythrum portula		e		Sagittaria rigida		ae	NN
Berula erecta	8.24	e		Lythrum salicaria	7.33	e		Sagittaria sagittifolia	9.24	ae	
Bidens cernua	8.13	e		Mentha aquatica	6.27	e		Schoenoplectus lacustris	8.44	ae	
Bidens tripartita	8.39	e		Menyanthes trifoliata	3.14	e		Schoenoplectus tabernaemont	7.43	e	
Bolboschoenus maritimus	7.65	e		Mimulus sp./hybrid	5.60	e	NN	Scirpus sylvaticus	6.45	e	
Butomus umbellatus	8.89	e		Myosotis laxa	4.82	e		Sparganium angustifolium	2.26	a	
Cabomba caroliniana		a	NN	Myosotis scorpioides	6.83	e		Sparganium emersum	8.32	a	
Callitriche brutia var hamulata	4.51	a		Myriophyllum alterniflorum	3.44	a		Sparganium erectum	8.34	e	
Callitriche hermaphroditica	5.75	a		Myriophyllum aquaticum		a	INN	Spirodela polyrhiza	8.99	a	
Callitriche obtusangula	8.04	a		Myriophyllum spicatum	8.26	a		Spirogyra sp	6.45	a	
Callitriche platycarpa	7.56	a		Myriophyllum verticillatum	7.53	a		Stratiotes aloides	8.51	a	(NN)
Callitriche spp.	6.67	a		Nitella flexilis (agg.)	4.39	a		Triglochin palustris	4.07	e	
Callitriche stagnalis	6.47	a		Nitella mucronata		a		Typha angustifolia	7.57	e	
Callitriche truncata	6.47	a		Nuphar lutea	8.42	a		Typha latifolia	8.87	e	
Caltha palustris	4.20	e		Nymphaea alba	5.69	a		Ulva flexuosa	9.52	a	
Carex acuta	7.19	e		Nymphoides peltata	9.37	a	(NN)	Utricularia australis	4.75	a	
Carex acutiformis	8.21	e		Oenanthe aquatica	6.06	ae		Utricularia vulgaris s.l.	5.28	a	
Carex elata	4.54	e		Oenanthe crocata	6.22	e		Vallisneria spiralis		a	NN
Carex paniculata	7.49	e		Oenanthe fistulosa	8.27	e		Vaucheria sp(p)	8.41	a	
Carex riparia	9.06	e		Oenanthe fluviatilis	8.57	a		Veronica anagallis-aquatica ag	8.34	e	
Carex rostrata	2.64	e		Persicaria amphibia	8.20	a		Veronica beccabunga	7.31	e	
Carex spp.		e		Persicaria hydropiper	6.97	e		Zannichellia palustris	9.01	a	
Carex vesicaria	3.68	e		Phalaris arundinacea	7.52	e					
Ceratophyllum demersum	9.73	a		Phragmites australis	7.70	e					
Ceratophyllum submersum		a		Pistia stratiotes		a	INN				
Chara globularis	3.30	a		Potamogeton alpinus	4.96	a					
Chara sp.	3.85	a		Potamogeton berchtoldii	7.35	a					
Chara vulgaris	3.77	a		Potamogeton compressus	8.33	a					
Cladophora/Rhizoclonium	8.66	a		Potamogeton crispus	8.02	a					
Crassula helmsii		a	INN	Potamogeton epihydrus		a	(NN)				
Elatine hexandra	4.17	a		Potamogeton friesii	8.19	a					
Eleocharis acicularis	5.35	a		Potamogeton gramineus	4.24	a					
Eleocharis palustris	4.54	e		Potamogeton lucens	8.54	a					
Eleogiton fluitans	2.06	a		Potamogeton natans	5.69	a					
Elodea canadensis	7.65	a	NN	Potamogeton obtusifolius	5.84	a					
Elodea nuttallii	9.44	a	NN	Potamogeton pectinatus	9.59	a					
Equisetum fluviatile	3.92	e		Potamogeton perfoliatus	8.16	a					
Filamentous green algae	7.61	a		Potamogeton polygonifolius	1.71	a					
Fontinalis antipyretica	5.40	a		Potamogeton praelongus	7.81	a					
Glyceria declinata	6.66	ae		Potamogeton pusillus	7.47	a					
Glyceria fluitans	5.25	ae		Potamogeton trichoides	7.24	a					
Glyceria maxima	9.64	e		Potamogeton x bennettii		a					
Glyceria x pedicillata	7.12	ae		Potamogeton x cooperi	6.07	a					
Groenlandia densa	7.96	a		Potamogeton x lintonii		a					
Hippuris vulgaris	5.94	ae		Potamogeton x salicifolius	6.36	a					
Hottonia palustris	6.93	a		Potentilla palustris	2.88	a					
Hydrocharis morsus-ranae	8.77	a	(NN)	Ranunculus aquatilis var aquatilis	5.67	a					
Hydrocotyle ranunculoides		e	INN	Ranunculus aquatilis var diffusus	7.65	a					
Hydrodictyon reticulatum	8.79	a		Ranunculus circinatus	9.42	a					
Iris pseudacorus	6.92	e		Ranunculus flammula	2.56	e					
Isoetes echinospora	4.06	a		Ranunculus fluitans	7.97	a					
Isoetes lacustris	3.02	a		Ranunculus peltatus var baudotii	9.06	a					
Juncus articulatus	3.10	e		Ranunculus hederaceus	5.47	e					
Juncus acutiflorus		e		Ranunculus penicillatus ssp. penicillati	6.29	a					
Juncus bulbosus	1.89	a		Ranunculus peltatus var peltatus	6.22	a					
Juncus effusus		e		Ranunculus penicillatus subsp vertumr	5.87	a					
Juncus inflexus		e		Ranunculus penicillatus ssp. pseudofl	7.92	a					
Lagarosiphon major	7.42	a	INN	Ranunculus lingua	7.61	ae					
Lemna gibba	10.00	a		Ranunculus sceleratus	9.86	e					

MFI refers to the species scores to be used in the calculation of the site MFI score

Habit refers to whether the plant is normally aquatic (a) or emergent (e). These designations are to be used in deriving the observed number of N_AQUA or N_EMERG at a site. For some taxa that may occur in either habits (ae) the species may contribute to one or both metrics depending on which growth form(s) is present.

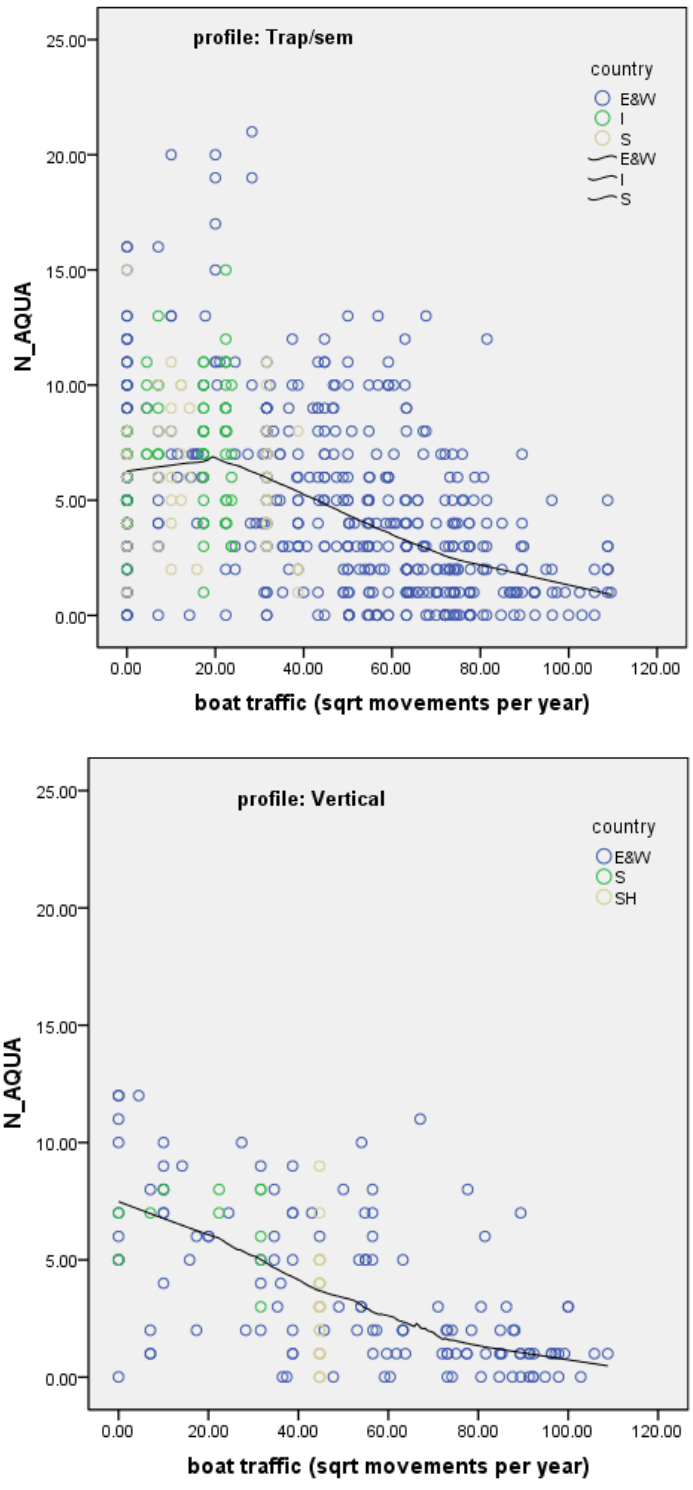
Status refers to the native status of the plant. INN = invasive or potentially invasive non native. NN = non native or naturalised species. (NN) = native species whose canal populations have arisen mainly or exclusively from deliberate introduction, often outside their native range.

3.2.4 Macrophyte Fertility Index (MFI)

The Macrophyte Fertility Index (MFI) is a compositional metric that is analogous to the macroinvertebrate Average Score Per Taxon metric. MFI represents a cover-weighted index score for a site based on a set of fertility ranks for different species. The origin of these ranks is the revision of the Ellenberg scoring system for European plants that assigned values (on a scale of 1-8) for different environmental variables to individual species according to their optimum growing conditions based on a combination of expert opinion and autecological studies (Hill et al.,2000). These scores have been adjusted and extensively tested via the LEAFPACS project (Willby et al., 2009). There is a strong correlation between these scores and those used in the Mean Trophic Rank (MTR) system but it has been possible to adjust where necessary and generate scores for other species not considered by MTR, using a reciprocal weighted averaging approach described by Hill et al. (2000). As a result of rescaling, the site scores range between 1.5 and 9.5 (the theoretical limits of 1 and 10 are never achieved because they would require the taxa associated with the most extreme ranks to occur in the absence of all other taxa), and in practice, on canals most scores lie in the range 6-8.5, the higher values indicating greater fertility. MFI can only be used reliably when the number of species present is four or more, of which at least two must be aquatic. In calculating the macrophyte inferred fertility the calculation is based on all in-channel species not solely the hydrophyte component. It should be noted that some hydrophyte taxa that are currently very rare in canals (including some non-native taxa) lack a species score and therefore do not contribute to the site value of this metric.

Of all the metrics MFI is the most strongly correlated with orthophosphate concentrations and therefore clearly merits a role in canal classification through incorporating sensitivity to eutrophication.

Figure 3.2. Relationship between N_AQUA and boat traffic in all surveys with respect to country and channel type.



E&W = England & Wales, S=Scotland, SH=Scottish Highlands, I=Ireland

3.2.5 Number of scoring emergent taxa (N_EMERG)

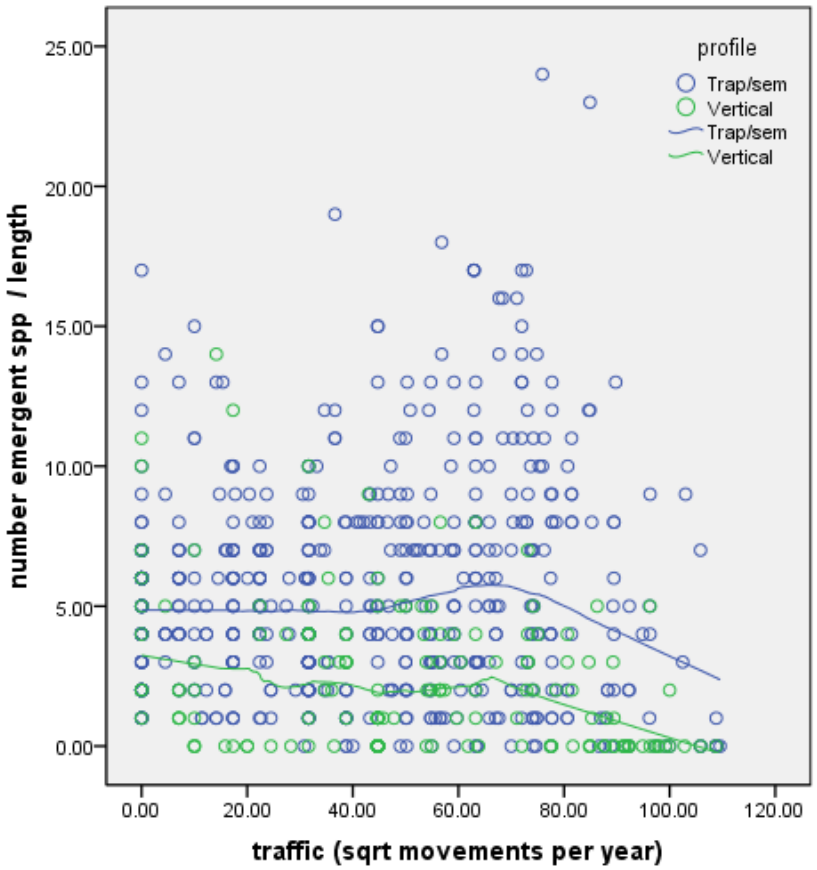
N_EMERG simply represents a count of the total number of scoring emergent plant taxa in a survey. Scoring taxa are listed in Table 3.2. To qualify for inclusion species must be growing with their basal parts submerged or rooted in fully saturated substrate at the time of survey.

It should be noted that a large number of species associated with fens or damp grasslands can occur alongside canals but are only loosely associated with the presence of standing water. N_EMERG is based on a fixed list of scoring species. The list is not an exhaustive list of the emergent plants that occur in canals but is an exclusive list for the purposes of determining observed values of this metric.

N_EMERG is well correlated with several water quality parameters, including BOD and chlorophyll, but its primary value is its sensitivity to bank reinforcement.

Figure 3.3 documents the relationship between the number of emergent species in standing sampling units at different boat traffic demsities and in different channel types. The relationship with boat traffic is very weak and the major feature is that soft engineered channels maintain a consistently higher number of species than hard engineered channels at all traffic densities.

Figure 3.3. Relationship between REL_EMERG and boat traffic in all surveys with respect to channel type.



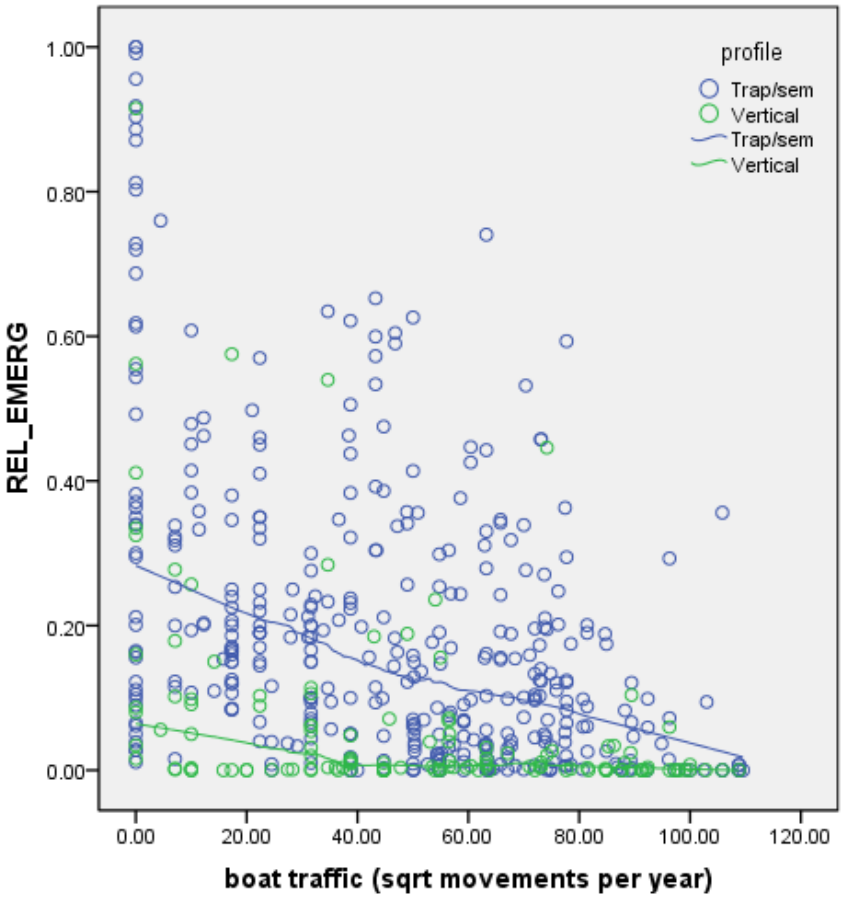
Trap/sem = Trapezoidal or semi-trapezoidal channel profile, Vertical = box shaped profile

3.2.6 Relative extent of emergent vegetation (REL_EMERG)

This metric refers to the proportion of the channel that is occupied by species with aerial foliage. Values range from 0 (i.e. no fringing vegetation) to 1 (i.e. the entire channel width is occupied by emergent vegetation). It is based on the survey of a standard length and requires information on the lateral extent of the reed fringe (or other species) on both banks and the width of the channel. The distance between the bank edge and outer edge of each reed fringe is normally estimated at the same 10 locations used for the grapnel throws. Access to the offside bank, or good quality aerial photography, will usually be required to verify the width of both reed fringes.

REL_EMERG is sensitive to both the density of boat traffic and the type of bank reinforcement (Figure 3.4). In all channel types thre is a steep reduction in the extent of emeregnt vegetation with increasing traffic caused by contraction and fragmentation of the reed fringe. In all channel types emergent vegetation is normally extremely restricted at traffic densities >10000 movements per year (my), although emergent vegetation is typically almost absent from channels where both banks are hard engineered once traffic exceeds 2000 my.

Figure 3.4. Relationship between REL_EMERG and boat traffic in all surveys with respect to channel type.



Trap/sem = Trapezoidal or semi-trapezoidal channel profile, Vertical = box shaped profile

3.2.7 Data quality and sampling efficiency

In this system of classification an Ecological Quality Index (EQI) is calculated for each metric based on the ratio of Observed: Expected value. The method will take no account of data quality and will simply classify the data with which it is provided.

Attention should therefore be given to the quality of survey data submitted for classification. Inspection of the available biological data indicates that MFI values calculated from surveys recording <5 taxa tend to be unstable and correlate poorly with pressure data. If the observed values of metrics such as APB or N_AQUA are abnormally low relative to expectations consideration should be given to sampling efficiency or whether a site has been affected by disturbances such as dredging or dewatering.

3.3 Identifying reference sites

3.3.1 Data availability

Macrophyte survey data was available for 790 surveys conducted on canals between 1985 and 2007. The vast majority of data (80%) was from canals in England & Wales, with compliant data having been obtained from canals in Scotland and Ireland, specifically for this project, in 2006. All data was collected in summer and as a minimum provided cover scores on a scale of 1-5 for aquatic plant taxa. Due to the use of data from a variety of sources the full range of metrics could only be calculated on half the survey data, while at least four of the five metrics could be calculated for 80% of surveys.

An ordination of the pool of macrophyte survey data illustrates the marked distinction between the vegetation of canals in (i) England & Wales or lowland Scotland (ii) Ireland and (iii) Scottish highlands (Figure 3.5). The differences between these sites are much greater for macrophytes than are seen at a family level in relation to macroinvertebrates, and highlight the need to take account of regional or national differences.

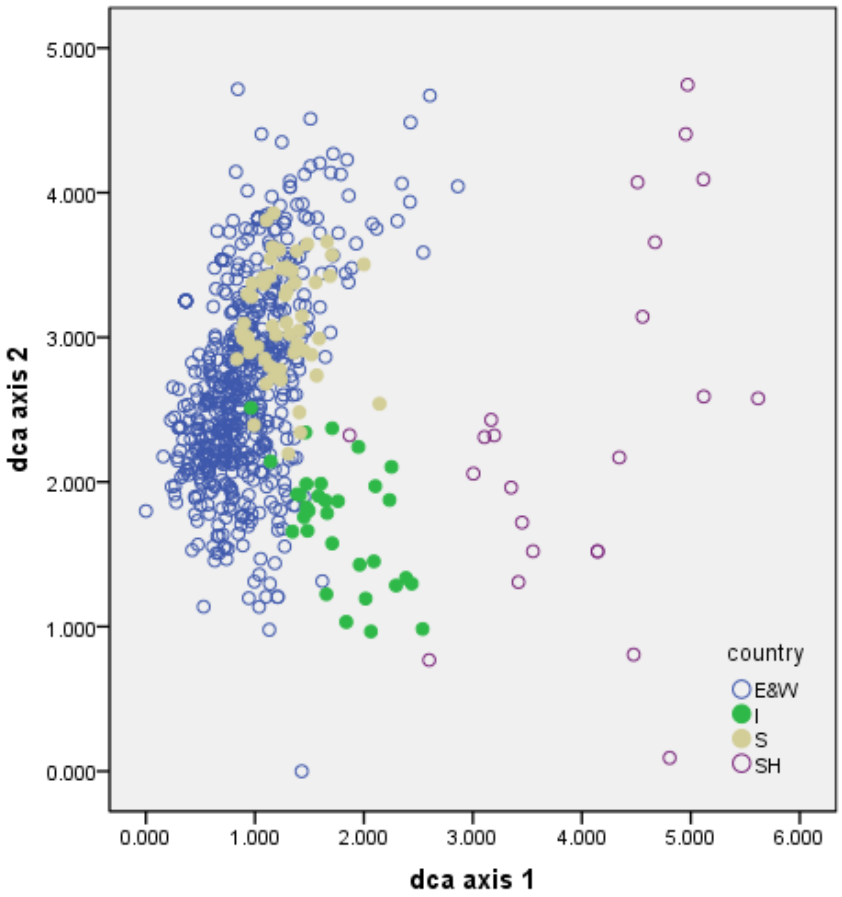
3.3.2 Environmental data availability

Sampling sites were located in a GIS which was used to cross match biological data with environmental data collected at the same or other sites within a water body. A nearest neighbour matching procedure was used to resolve the nearest spatial and temporal match. Matches >1km apart or displaced by >2 years were excluded. Because most macrophyte surveys were not undertaken for GQA purposes the links to water quality data obtained from routine monitoring sites was much more limited for macrophytes than invertebrates. Hence, data on the most frequently recorded chemical determinands were only available for about 25% of the macrophyte surveys.

Survey sites were also linked to spatial data provided by BW on boat traffic, bank type and condition from towpath surveys, location of linear mooring sites and embankments and hydrocompliance status.

Data on channel type was not available for Ireland and was therefore derived from an inventory of site photographs in conjunction with the use of Google Earth.

Figure 3.5. Ordination by Detrended Correspondence Analysis (dca) of macrophyte survey data showing differences in composition at national or regional levels.



E&W = England & Wales, S=Scotland, SH=Scottish Highlands, I=Ireland

3.3.3 Data screening

To screen out potential reference sites the major relationships between each metric and the environmental variables referred to in 3.2 were used. In each case the lower 25th percentile of the residuals were flagged to identify those surveys with good biology for a given level of boat traffic, alkalinity or channel type. To arbitrarily select, for example, the upper quartile of N_AQUA values, all of which will occur at very low traffic, would discriminate against boat traffic and would ensure that sites at high traffic were always classified at low ecological potential. Taking account of the underlying relationship between a metric and boat traffic avoids this discrimination. Analysing the data on a country-by-country basis ensured that thresholds based on data from England & Wales (which dominate the dataset) cannot biass the selection of reference sites by excluding sites in a different country that fall below this threshold. A subset of surveys from each country was then identified where values for the majority of metrics calculated fell within the lower quartile of the distribution of residuals. This approach ensured that the potential reference sites were of generally good biological quality and did not contain deviant data where, for example, low (i.e. good) MFI values occurred in association with low numbers of aquatic taxa, or vice versa.

This pool of sites of 'good' biological quality was then filtered using available matched abiotic data relevant to water quality pressures on canals (organic pollution, eutrophication, heavy metal pollution). The match between sites with macrophyte survey data to sites with routine chemical monitoring was relatively poor (only 20-30% of surveys could be matched to measured data for the most well monitored variables). Thus, proper screening could only be accomplished in relation to ammonia, BOD, TON, SRP, chloride, chlorophyll a, suspended solids, and the metals copper and zinc (water column concentrations). The objective of filtering by abiotic data was to obtain a pool of minimally impacted sites with high biological quality of low variability. This was obtained by plotting potential reference and non-reference sites in ordination space and then systematically removing outliers that could be accounted for by elevated values of abiotic variables.

The remaining pool of sites were also checked against the smaller body of data available for other metals, including mercury, cadmium, chromium and lead, to confirm that concentrations were not elevated above background at these sites. Sites were also checked in GIS to ensure that they were not located close to moorings or known marinas (since these are likely to elevate disturbance above that associated with the reported traffic figures) and they did not fail Hydrocompliance standards according to data provided by BW.

The final population of canal reference sites comprised data from 127 sites of which 89 (70%) were in England & Wales, 18 (14%) in Ireland and 20 (16%) in Scotland.

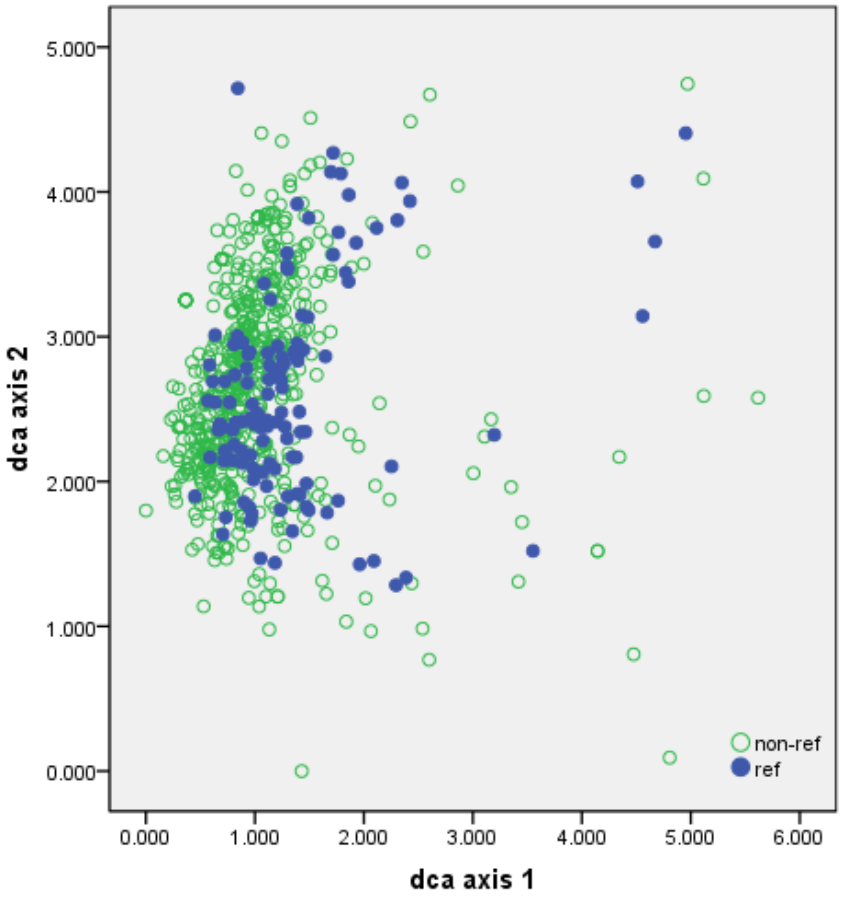
An ordination of reference and non-reference sites using detrended correspondence analysis (Figure 3.6) illustrates that after screening there is still a relatively high dispersion of macrophyte reference sites when compared to invertebrate reference sites (Figure 2.2) which reflects the marked difference in vegetation composition between high alkalinity canals in central and southern England and low alkalinity canals in the Scottish highlands.

Table 3.3. Annual mean values of selected pressure indicating variables (median and lower or upper percentiles) in macrophyte reference sites.

	Ammonia –N	BOD	Chloride	Chloro-phyll a	Copper	Total Oxidised Nitrogen	Soluble Reactive P	% DO Satura-tion	Suspen -ded Solids	Zinc
units	mg/L	mg/L	mg/L	uScm	mg/L	mg/L	mg/L	%	mg/L	mg/L
median	0.086	2.74	37.0	0.007	0.002	1.67	0.027	92.0	12.0	0.010
90th %tile	0.134	3.87	75.2	0.019	0.004	2.27	0.066	76.3 †	51.0	0.025

† Expressed as 10th percentile

Figure 3.6. Ordination based on Detrended Correspondence Analysis (dca) of macrophyte survey showing distribution of macrophyte reference (ref) sites and non-reference (non-ref) after screening by biology and pressure data.



3.4 Calculating expected values for macrophyte metrics

3.4.1 Data requirements

In order to model the expected values for the various macrophyte metrics at the reference sites biological data was linked to environmental data covering boat traffic density, channel type (reflecting the type of marginal habitat), alkalinity (as an indicator of underlying geology). The purpose of these models was to determine the expected value of a biological metric at reference sites where the impacts of anthropogenic pressures excluding navigation are minimal.

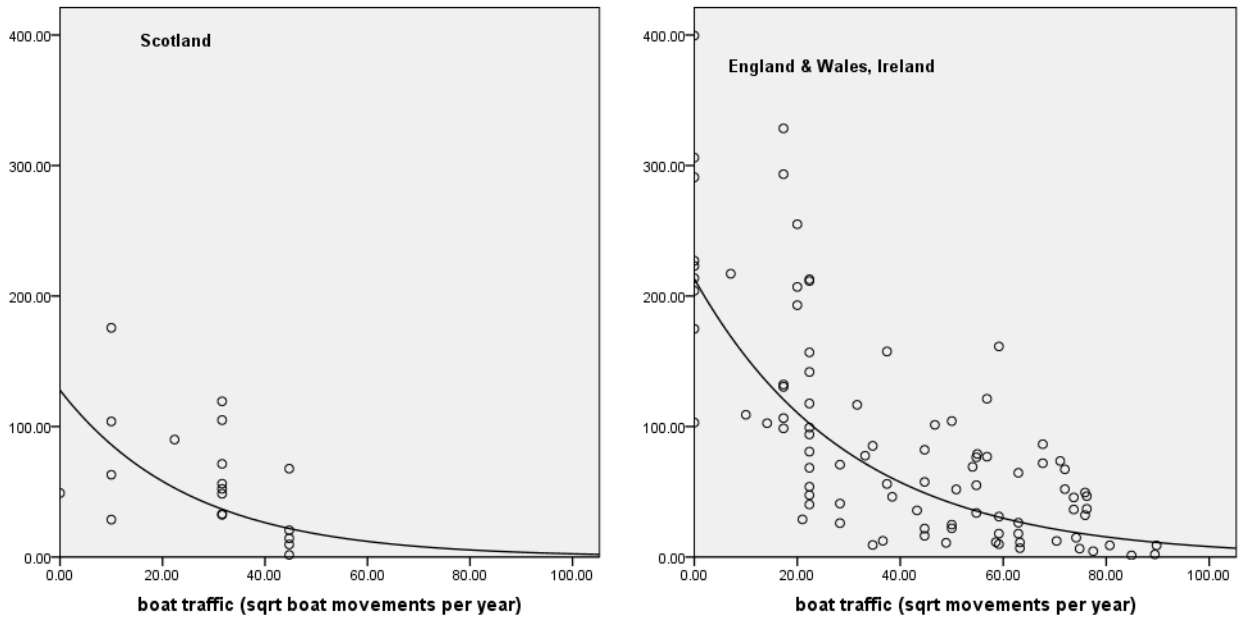
With the exception of channel type, the values used are taken at face value and are therefore as recorded in the field, or derived from separate monitoring. As explained in section 2.4.1 channel type presents a special case because the observed channel type may reflect a level of hydro-morphological impact beyond that appropriate to the specific setting. Figure 2.3 should therefore be used to evaluate the expected channel type at a site.

3.4.2 Expected APB

While there is a considerable amount of unexplained variability, the amount of aquatic plant biomass in reference sites is clearly sensitive to boat traffic (Figure 3.7).

However, there is a further complication in modelling expected biomass because the value at reference sites differs strongly between some countries or regions in this project. This almost certainly reflects biogeographical factors, possibly allied to major differences in water chemistry. Thus, biomass tendes to be markedly lower at canals in Scotland, for the same level of boat traffic, than it is at reference sites elsewhere. In Scotland this is not only a feature of the highland canals and probably reflects the fact that the water feed to the lowland canals is also generally derived from relatively base- and nutrient-poor sources compared to lowland canal sites in other countries in this study. The consequence of this is that different models are required to predict expected plant biomass at reference sites in (i) Scotland and (ii) England & Wales and Ireland.

Figure 3.7. Relationship between aquatic plant biomass (APB, y axis) and boat traffic in reference sites in Scotland (left) and England & Wales and Ireland (right).



3.4.3 Expected N_AQUA

N_AQUA decreases strongly with traffic, this relationship being sublty different between soft and hard engineered channels (Figure 3.8). The tendency for increased richness of aquatic plants in hard engineered channels at low traffic is not reproduced within the reference site model.

There is some evidence that the median richness of low traffic sites in both Scotland and Ireland is lower than sites in England & Wales with the same level

of boat traffic. Unfortunately the number of reference sites outwith England & Wales is rather small and it is therefore difficult to verify this pattern. A recent adjustment of the LEAFPACS river classification method which had access to almost 400 reference sites suggested that a slightly lower reference number of aquatic taxa would be appropriate when classifying rivers in Northern Ireland, with the standard being set at 80% of the expected number of taxa at otherwise similar sites in GB. This same adjustment has therefore been applied in this instance but also including Scotland. This takes into consideration the likely effect of catchment isolation and lower alkalinity on numbers of taxa.

Figure 3.8. Relationship between aquatic plant species richness (NAQUA, y axis) and boat traffic in reference sites in soft (left) and hard engineered channels (right).

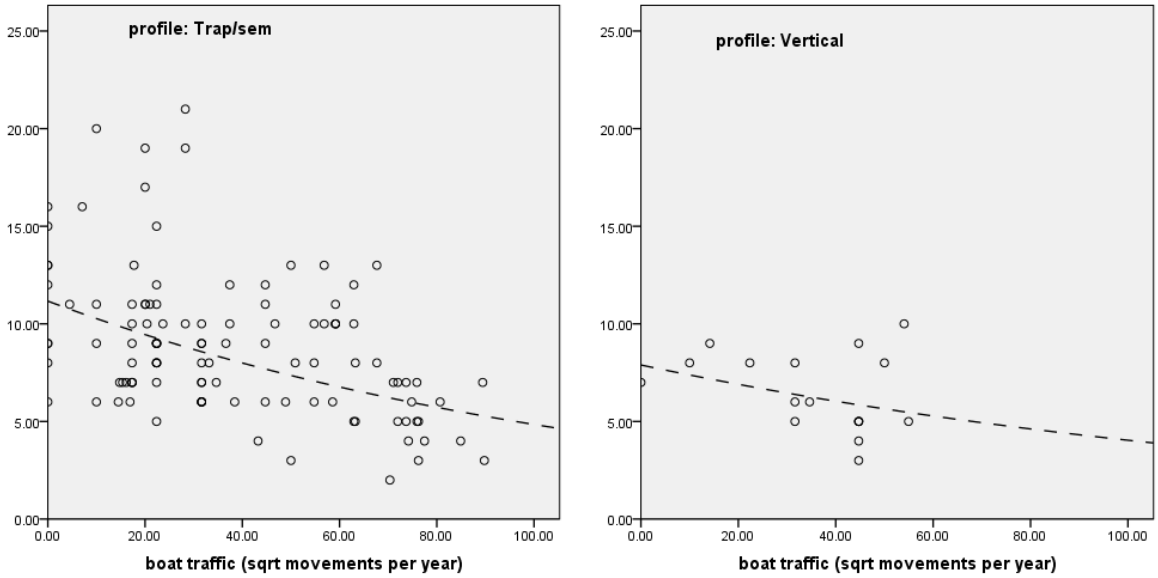


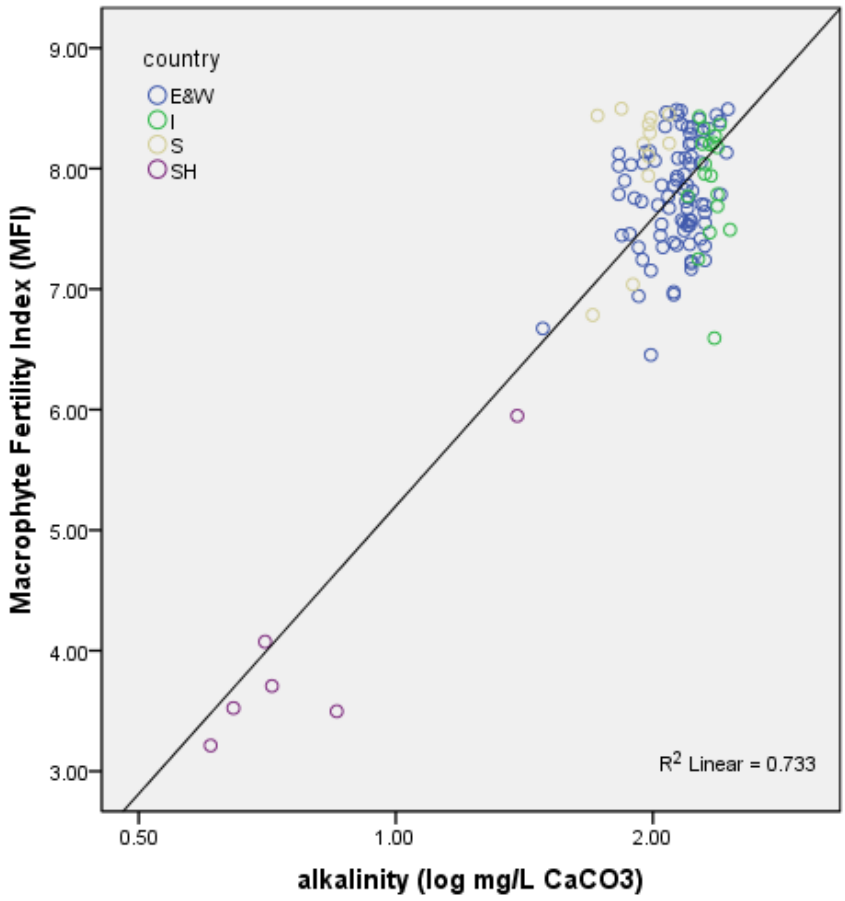
Table 3.4. Functions and coefficients for modeling expected values of macrophyte metrics.

metric	conditions	country/region	r^2	model or fixed value
APB	all channel types	England, Wales & Ireland	0.51	= 213.20 * EXP(-0.033 * sqrt(boat traffic))
		Scotland	0.27	= 127.85 * EXP(-0.039 * sqrt(boat traffic))
NAQUA	Box-shaped channel	England & Wales	0.10	= 7.89 * EXP(-0.0067 * sqrt(boat traffic))
		Scotland & Ireland	0.10	= 7.89 * EXP(-0.0067 * sqrt(boat traffic)) * 0.8
	Trapezoid/asymetric channel	England & Wales	0.24	= 11.16 * EXP(-0.008 * sqrt(boat traffic))
		Scotland & Ireland	0.24	= 11.16 * EXP(-0.008 * sqrt(boat traffic)) * 0.8
MFI	use only if no. scoring taxa ≥4 and N_AQUA>1	all countries, all channel types	0.76	= 5.124 + 3.539 * LN (log alkalinity)
N_EMERG	Trapezoid/asymetric channel	England & Wales	fixed	10
		Ireland	fixed	6
		Scottish lowland	fixed	6
	Box-shaped channel	England & Wales	fixed	5
		Scottish lowland	fixed	4.5
		Scottish highland		not used
REL_EMERG	Trapezoid/asymetric channel	all countries	0.10	= 0.315 * EXP(-0.0069 * sqrt (boat traffic))
	Box-shaped channel	all except Scottish highland	0.27	= 0.449 * EXP(-0.0806 * sqrt (boat traffic))
		Scottish highland		not used

3.4.4 Expected MFI

MFI is highly dependent on alkalinity since this reflects the underlying geology and the potential supply of phosphorus from natural sources by rock weathering. The relationship is best described by a logarithmic relationship over the range of alkalinity values sampled (Figure 3.9). The reference MFI values in Scottish highland canals are equivalent to those of low gradient base-poor upland rivers or shallow low alkalinity lakes. At the opposite end of the scale the upper reference value of MFI for very high alkalinity canals in England or Ireland is ~8.3. Note that the upper expected River Macrophyte Nutrient Index (RMNI) values for very high alkalinity, very low gradient rivers, of which canals are the closest parallel, is 7.9. The higher value for canals reflects high siltation rates and the potential for extensive vegetation development due to low water velocity combined with shallow depth.

Figure 3.9. Relationship between alkalinity and MFI at canal reference sites illustrating distribution of regional surveys.



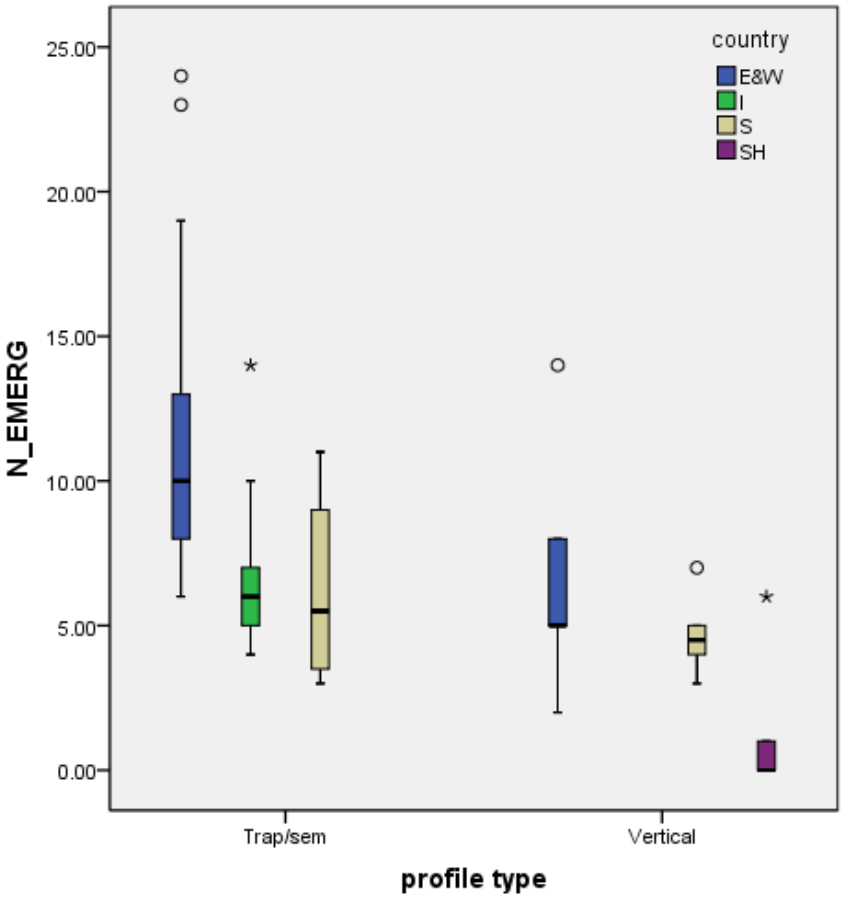
3.4.5 Expected N_EMERG

N_EMERG is highly affected by the type of channel cross-section because vertical banks more or less eliminate shallow marginal habitat. It is insensitive to boat traffic, perhaps because losses of some species as traffic increases are offset by increased recruitment of smaller species that is facilitated by reduction

in the dominance of common larger swamp-forming taxa such as *Glyceria maxima* or *Sparganium erectum*.

The complication with N_EMERG is that the standards at reference sites for different channel types show strong regional or national differences (Figure 3.10). Thus, in sites with a trapezoidal or semi-trapezoidal profile the number of emergent species is about 40% lower at sites in lowland Scotland or Ireland than it is in England & Wales. Therefore, for this channel type, a reference value of 10 is applied to sites in England & Wales, but 6 in lowland Scotland or Ireland. In hard-engineered channels, the median number of emergent species is constrained in all sites but is still higher in England & Wales (6) than in lowland Scotland (4.5). Emergent species are present erratically and in such small numbers fringing Scottish highland canals that this metric has no utility in that environment.

Figure 3.10. Relationship between N_EMERG and channel type at canal reference sites illustrating regional differences, notably in relation to trapezoidal channels.



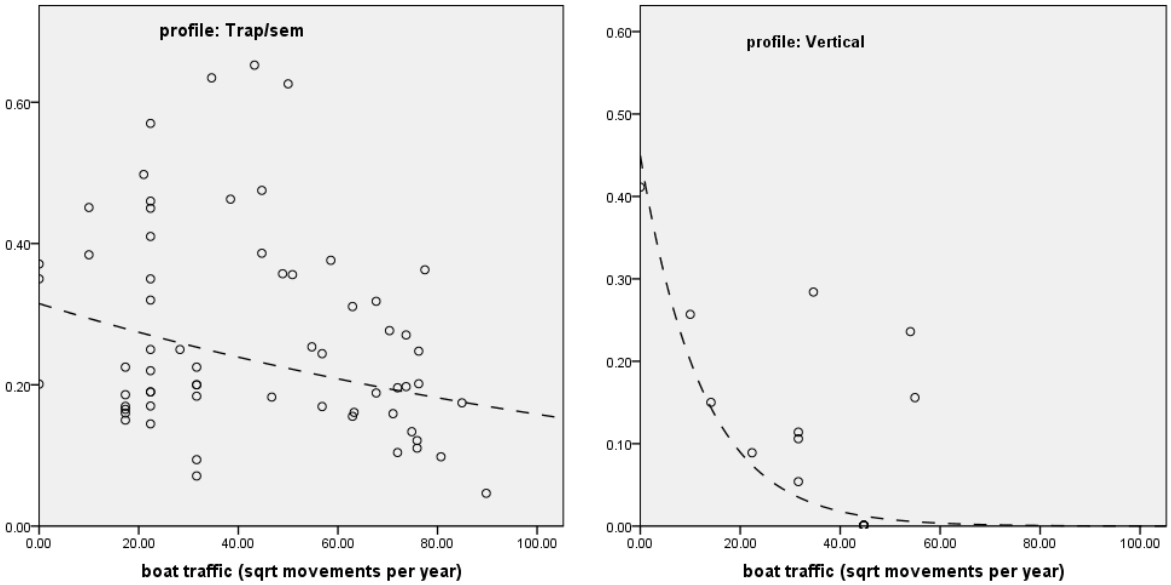
E&W = England & Wales, S=Scotland, SH=Scottish Highlands, I=Ireland
Trap/sem = Trapezoidal or semi-trapezoidal channel profile, Vertical = box shaped profile

3.4.6 Expected REL_EMERG

At reference sites REL_EMERG shows a clear reduction as traffic increases with the rate of reduction being dependent on channel type and therefore occurring more rapidly in hard engineered channels (Figure 3.11).

There is no evidence in the case of this metric that different geographical groupings of sites support relative quantities of emergent vegetation. Thus, for a given channel type and boat traffic the relative proportion of the channel occupied by emergent vegetation will be the same in Scotland, Ireland or southern England. The exception to the use of this metric is provided by the Scottish Highland canals where the combination of low fertility and banks composed of sloping embedded rock results in an almost complete absence of emergent vegetation from reference sites. Consequently, as with N_EMERG, this metric is not employed in assessment of Scottish highland canals.

Figure 3.11. Relationship boat traffic and REL_EMERG at canal reference sites illustrating dependency on channel type.



3.5 Calculating an Ecological Quality Index (EQI) for macrophyte metrics

3.5.1 APB

For this metric:

EQI_{APB} = Observed APB / Expected APB

Where the Observed APB is derived from a a survey conducted between 1 June and 31 September according to the prescribed method and Expected APB is calculated according to the equation in Table 3.4, and adjusted for the country/region in which the survey site is located.

3.5.2 N_AQUA

For this metric:

EQI_{N_AQUA} = (Observed N_AQUA) / (Expected N_AQUA)

Where the Observed N_AQUA is derived from a survey conducted between 1 June and 31 September according to the prescribed method and is calculated using only the scoring taxa as listed in Table 3.2. Expected N_AQUA is calculated according to the equation in Table 3.4.

3.5.3 MFI

For this metric:

EQI_{MFI} = (Observed MFI -10)/ (Expected MFI-10)

Where the Observed MFI is derived from a survey conducted between 1 June and 31 September according to the prescribed method and Expected MFI is calculated according to the equation in Table 3.4. MFI can only be used reliably when the total number of taxa recorded is ≥ 4, of which ≥ 2 taxa are aquatic. Note that in this equation, 10 is deducted from both Observed and Expected MFI values to ensure that low values of MFI (less fertile) are associated with higher quality.

3.5.4 N_EMERG

For this metric:

EQI_{N_EMERG} = Observed N_EMERG / Expected N_EMERG

Where the Observed N_EMERG is derived from a survey conducted between 1 June and 31 September according to the prescribed method and is calculated using only the scoring taxa as listed in Table 3.2. Expected N_EMERG is based on the standards in Table 3.4 that take account of channel type and country of origin. N_EMERG is not suitable for assessment of Scottish Highland canals.

3.5.5 REL_EMERG

For this metric:

EQI_{REL_EMERG} =Observed REL_EMERG / Expected REL_EMERG

Where the Observed REL_EMERG is derived from a survey conducted between 1 June and 31 September according to the prescribed method and Expected REL_EMERG is calculated according to the equation in Table 3.4. REL_EMERG is not used for assessment of Scottish Highland canals.

3.6 Combining EQIs to achieve a macrophyte-based classification

3.6.1 Establishing class boundaries

In this project class boundaries for the different metrics were mainly established statistically by examining the distribution of EQI values for the various metrics in samples from reference sites. The advantages and disadvantages of this approach are discussed in section 2.6.1 in relation to invertebrates. In reality consensus regarding placement of boundaries between Maximum/Good or

Good/Moderate Ecological Potential can only be established through the process of intercalibration of national assessment methods between Member States.

In this project the reference dataset is relatively small (127 surveys), canals in general are relatively homogenous (as opposed to the difference between say small upland streams and large lowland rivers), and non-compositional metrics (all those used here except MFI) tend to be inherently rather variable and more difficult to model effectively. This means that there is often a large amount of unexplained variation in the metrics within the population of reference sites and class widths must be correspondingly large enough to reflect this. The variation in EQI values within reference sites is reflected in the wide spread of the histograms in Figure 3.12.

In this project the boundary between Maximum and Good Ecological Potential was based on the lower 20th percentile of the distribution of EQI values in reference sites. This is a relatively precautionary approach (normally the lower 5th or 10th percentile is used) but it ensures that all metrics can be subdivided into at least four classes. Subsequent class boundaries are then based on the width of the Maximum Ecological Potential class. The resulting boundaries are reviewed in Table 3.5. It should be noted that for two metrics, APB and REL_EMERG the variation in reference sites was sufficiently high to dictate that only three quality classes could be potentially recognised based on these metrics and taking a lower 20th percentile to form the boundary between Maximum and Good Ecological Potential. In these two cases it was necessary to adopt an arbitrary boundary which is equivalent to taking a more precautionary upper percentile. Thus for APB and REL_EMERG the boundary between Maximum and Good Ecological Potential was set at 0.75 and and between Good and Moderate Ecological Potential at 0.5. This is equivalent to stating that sites at the Max/Good EP boundary must have 75% of the expected aquatic plant biomass, or extent of reed fringe, of reference sites. Those at the Good/Moderate EP boundary must have 50% of the value expected for these metrics under reference conditions.

Figure 3.12. Histograms showing frequency distribution of EQI values for different macrophyte metrics in a population of 127 reference sites

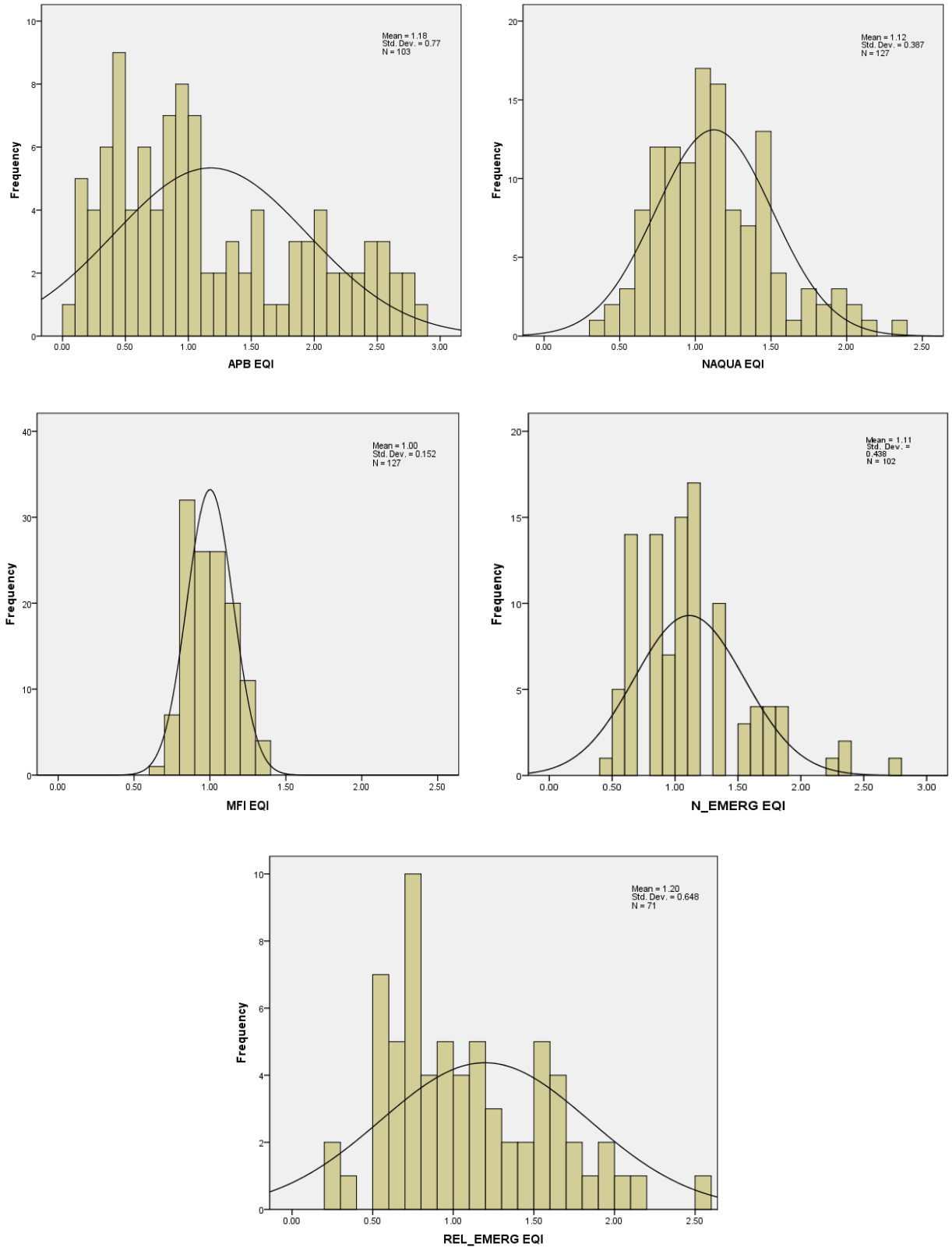


Table 3.5. Class boundaries for different levels of Ecological Potential in five macrophyte metric EQIs.

EP boundary	Max/Good	Good/Mod	Mod/Poor	Poor/Bad
APB	0.75	0.50	0.25	
N_AQUA	0.80	0.60	0.40	0.20
MFI	0.85	0.70	0.55	0.40
N_EMERG	0.80	0.60	0.40	0.20
REL_EMERG	0.75	0.50	0.25	

3.6.2 Rescaling EQI values to an EQR

In multimetric approaches such as the one described here metrics need to be combined to provide an overall classification for a site. Before this is possible it is essential to map all the metrics onto a common class boundary system. A class width of 0.2 is normally used. Hence the MEP/GEP boundary in each metric is then mapped to a value of 0.8. This approach requires a piecewise linear transformation of each metric from its original class boundaries so that the values of all metrics can be compared directly. The rationale and an example are given for invertebrates in section 2.6.2 and a worked example using macrophytes can be found in section 3.8.5.

The piecewise linear transformation ensures the conversion of all EQI values to a standardised set of EQRs ranging from 0 to 1 with class boundaries at intervals of 0.2 (Table 3.6).

Table 3.6. Ecological Potential boundaries after implementation of piecewise linear transformation to standardise EQI values to an EQR scaling.

	Max	Max/Good	Good/Mod	Mod/Poor	Poor/Bad	Min
APB	1.0	0.8	0.6	0.4		0.0
N_AQUA	1.0	0.8	0.6	0.4	0.2	0.0
MFI	1.0	0.8	0.6	0.4	0.2	0.0
N_EMERG	1.0	0.8	0.6	0.4	0.2	0.0
REL_EMERG	1.0	0.8	0.6	0.4		0.0

3.6.3 Rules for combining EQR values

A series of potential rules for combining EQR values for individual metrics were considered in line with those trialled for invertebrates. These included

- the average of all metrics
- the minimum of all metrics
- the average of the lowest two or lowest three metrics
- the average of different ‘families’ of metrics

The effect of using different approaches to combine EQR values on the overall EQR for the site is illustrated in Table 3.7. The average of all metrics was the most relaxed simple approach, whilst the minimum of all is the most precautionary. Averaging multiple metrics when some metrics have inherently lower levels of precision is probably unwise since thess metrics should strictly carry less weight. In this study the metrics APB and REL_EMERG were therefore arbitrarily assigned half the weight of the metrics MFI, N_AQUA and N_EMERG. Averaging multiple metrics is also inadvisable when the combination of available metrics may differ between sites

Based on an analysis of relationships between different quality elements, the chosen approach for combining metrics was to calculate an average value from three groups of metrics (i) MFI reflecting water quality (ii) APB and N_AQUA reflecting in-channel disturbance and (iii) N_EMERG and REL_EMERG reflecting bank-related disturbance. Thus the overall macrophyte EQR is defined as:

$$\text{Final EQR} = (EQR_{MFI} + ((EQR_{APB} \times 0.5 + EQR_{N_AQUA})/1.5) + ((EQR_{REL_EMERG} \times 0.5 + EQR_{N_EMERG})/1.5)) / 3$$

The formula is modified accordingly if certain individual metrics cannot be used or cannot be calculated. No site should be assessed on the basis of less than three metrics. Although this in fact represents the most relaxed combination rule it is important to take account of the classification results already established using invertebrates. If the combination rule for macrophytes provides a very different set of classification results to that obtained using invertebrates one or other quality element will effectively become redundant once overall classification results are obtained using the one-out, all-out rule. The macrophyte combination rule based on the average of three groups produces the most similar distribution of surveys between the different classes of Ecological Potential to the rule established for invertebrate classifications.

Table 3.7. Proportion of surveys assigned to different classes of Ecological Potential based on different rules for combining metrics, based on 762 surveys for which at least three metrics were available.

Combination	Max	Good	Moderate	Poor	Bad
minimum of all	6.2	10.4	15.2	14.8	53.4
average of all	22.4	26.6	24.8	15.4	10.8
minimum of grps	10.0	16.8	19.0	22.3	31.9
average of 3 grps	23.5	29.4	25.3	11.0	10.8
average of lowest 2 grps	15.5	23.8	28.1	20.3	12.3

3.6.4 Implications for national classifications

Using the scheme described for combining metrics to produce an overall classification it is possible to determine the proportion of sites in different countries that will be classified at different levels of Ecological Potential. Table 3.8 provides this data. Scotland is split into lowland and highland canals due to their marked differences. The data for Scotland and Ireland is based mostly on a

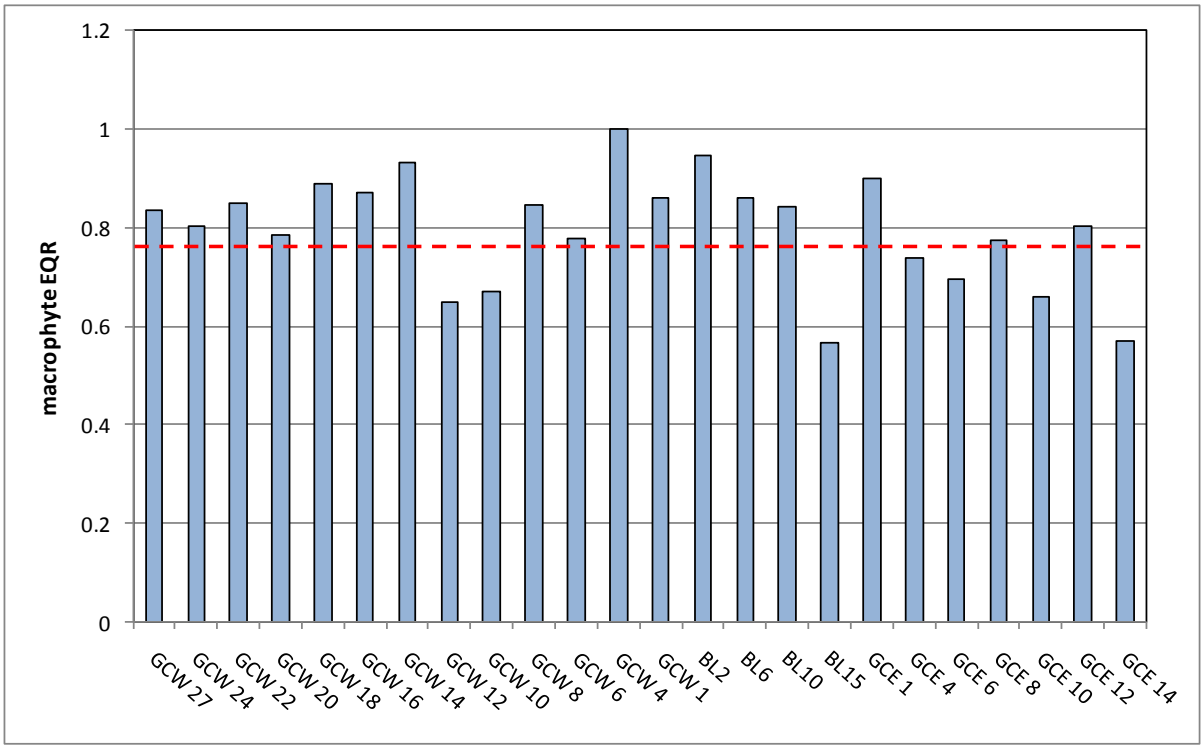
single year of assessment but probably offers a reliable general picture of the current status of canals in these countries. For England & Wales the amount of data is large but the majority of assessments are based on surveys conducted 10-20 years ago and this therefore cannot be assumed to offer a reliable picture of the current status of these canals. Failing classifications of the Scottish highland canals should be treated with caution due to the potential problems of sampling efficiency in relation to habitat that have been discussed earlier. As a general picture 47% of surveys of canals in GB and Ireland are likely to fall below Good Ecological Potential, of which 93% are in England & Wales.

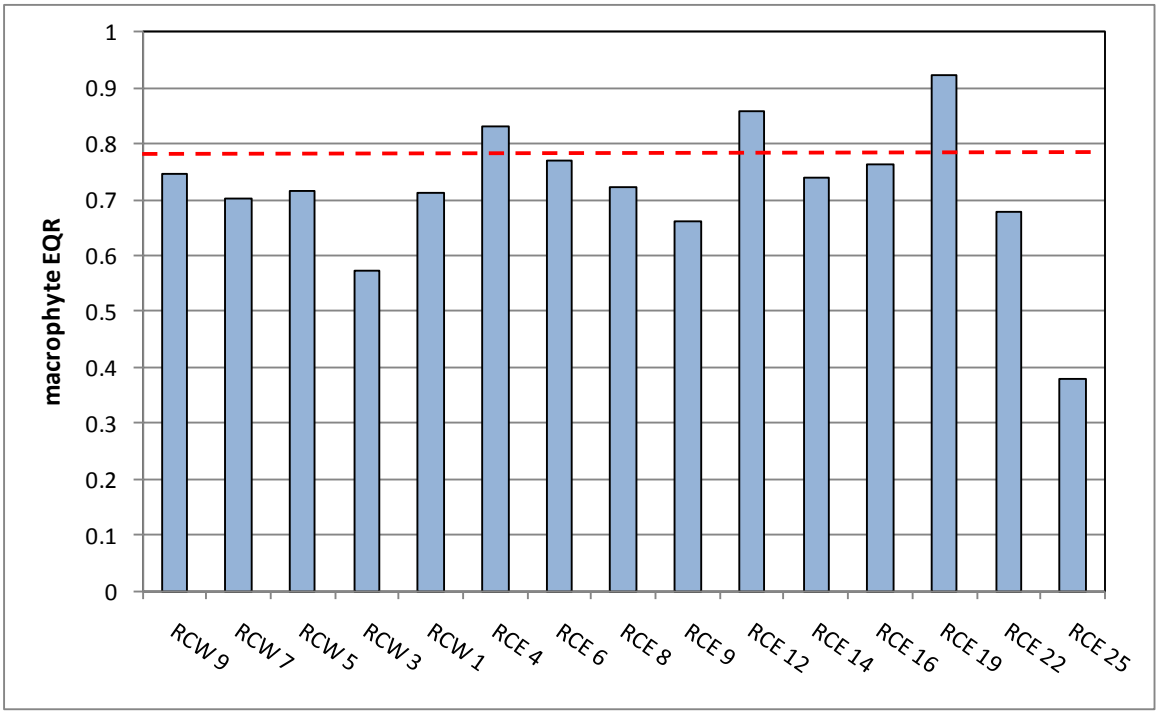
Table 3.8. Numbers and proportions of surveys in different countries/regions classified at differing levels of Ecological Potential according to a combination rule based on the average of three groups of metrics.

Country		Max	Good	Moderate	Poor	Bad
England & Wales	n	114	173	177	80	89
	%	18.0	27.3	28.0	12.6	14.1
Ireland	n	39	26	4	1	0
	%	55.7	37.1	5.7	1.4	0.0
Scotland - lowland	n	22	22	9	4	0
	%	38.6	38.6	15.8	7.0	0.0
Scotland - highland	n	12	4	5	2	1
	%	50.0	16.7	20.8	8.3	4.2

A detailed example is given below concerning the classification of sites on Irish canal water bodies to match the assessment provided in section 2.6.4 using invertebrates. A total of thirty nine sites were surveyed in 2006 belonging to different water bodies. Figure 3.13 illustrates the macrophyte-based EQR for sites on the Grand and Royal Canal systems, with sites ranked from west to east. The global average EQR of all sites assessed was 0.77 (upper end of Good Ecological Potential) reflecting their generally high quality, and is consistent with the invertebrate-based assessment. Seventeen sites were classified at Maximum Ecological Potential (EQR ≥ 0.8), while eighteen sites were classified at Good Ecological Potential, with only four sites (10%) being classified below Good Ecological Potential. The quality of the Grand Canal (average EQR = 0.8) was generally higher than that of the Royal Canal (average EQR = 0.72) on the basis of its macrophytes. At a water body level all waterbodies on both waterways would be classified at or above the middle of Good Ecological Potential, based on the averaging of the EQRs of the sites belonging to each water body.

Figure 3.13. Classification of sites on Irish Canals based on macrophytes. Upper panel Grand Canal, including Barrow Line, lower panel Royal Canal. Red dashed line is the global mean value across all sites assessed. Classifications based on samples collected summer 2006 by Central Fisheries Board.





The only geographical trend from this analysis is for slightly lower quality of sites on both the Grand and Royal canals in the vicinity of Dublin.

From a summary of site-based classifications, it is also possible to determine the proportion of sites across GB and Ireland failing to achieve Good Ecological Potential for different reasons. The three major groups of metrics were all assessed at 542 surveys. Of these sites MFI is the lowest metric in 15% of cases whilst there is a roughly equal split between pressures associated with in-channel or bank-based disturbance. Taking the 184 surveys where the overall macrophyte assessment reports a value below GEP failure on the basis of MFI appears to be very uncommon whilst bank-based disturbance is the major likely reason for failure (Table 3.9). The implication is that water quality-related pressures can discriminate between Maximum and Good status but are rarely a direct cause of failure to achieve GEP. Conversely the majority of sites that fail to achieve GEP appear to do so because they have unusually species-poor or restricted development of emergent vegetation relative to what would be expected for the channel type and density of boat traffic.

The proportion of sites that are likely to be failing to achieve GEP for macrophytes specifically due to water quality pressures is noticeably lower than for invertebrates (28%). However, this finding does not mean that water quality pressures are unimportant at other sites, simply that other metrics linked to in-channel or bank-based pressures have lower values, which may themselves be attributable indirectly to water quality pressures.

Figure 3.14. Percentage of occasions different groups of metrics dictated the classification of a subset of sites from GB and Ireland.

Major pressure	**All sites** (n=542)	**Sites below GEP** (n=184)
Water quality (MFI)	15	4
In-channel (APB+NAQUA)	43	36
Bank-based (N_EMERG+REL_EMERG)	47	60

3.6.5 Comparing classifications of ecological potential based on macrophytes and invertebrates

When data is spatially and temporally contemporaneous it is possible to directly compare classifications based on macrophytes or macroinvertebrates and to examine the implications of basing an overall classification on the one-out all-out approach. This is possible using the data collected in 2006 in Ireland and Scotland. In each country 39 sites were assessed for both macrophytes and macroinvertebrates.

A test of the correlation between EQR values for invertebrates and macrophytes indicates that the classifications of the two quality elements are uncorrelated. In some respects this is surprising although it is reassuring that neither quality element can be considered to be redundant, as would be the case if the EQR values were highly correlated. In general this suggests that macrophytes and invertebrates are responding to different pressures in canals, are responding to common pressures but at different rates, or both of these.

The results of comparing classifications of sites in Ireland and Scotland using macroinvertebrates and macrophytes are summarised in Table 3.10. Values are directly comparable between countries as the same numbers of sites were assessed in each. A number of key points emerge:

- Classifications of sites in the two countries are very similar, whether based on macrophytes, macroinvertebrates or both.

- Neither macrophytes nor invertebrates are conspicuously more precautionary which lends supports to the combination rules employed in this project. When classifications are determined using the worst of either macrophytes or invertebrates across the 78 sites assesed in both countries macrophytes determine the classification in 40% of cases, invertebrates in 37%, while in 23% both quality elements differ by <0.05 EQR units (one quarter of a class) and can then be said to jointly determine the classification.

- The one-out all-out approach is, by its nature, more precautionary. Thus, using either macrophytes or invertebrates for classification results in 81% of sites across the two countries being classified at Maximum or Good Ecological Potential. Using the one out all out approach this proportion drops to 67%.

Table 3.9. Comparison of classifications of canal sites in Ireland and Scotland sampled in 2006 and assessed using invertebrates, macrophytes or both quality elements.

Country	Quality element	Max	Good	Moderate	Poor	Mean EQR	Class-driver
Ireland	Invertebrates	16	13	7	3	0.72	17
	Macrophytes	17	18	3	1	0.77	14
	both	6	21	9	3	0.66	8
Scotland	Invertebrates	18	16	4	1	0.77	12
	Macrophytes	13	16	7	3	0.71	17
	both	8	17	10	4	0.65	10

3.7 Determining the EQR for a water body

The above details will enable an EQR to be calculated for a single site based on a single survey. If the waterbody is only assessed at a single location, and has been surveyed only once in a monitoring cycle the EQR calculated will represent the EQR for the water body as a whole. If a water body is surveyed at multiple locations on a single ‘date’ (i.e. all in the summer of the same year), the average EQR for all sites surveyed should be determined to given an EQR for the water body. If the water body is surveyed more than once in a monitoring cycle, its overall EQR is the average of the separate waterbody EQRs calculated for each year of sampling.

3.8 Worked example

3.8.1 Raw data

The data used in this example was collected by SEPA in summer 2006 from the Forth & Clyde Canal at Lambhill which is located in the north of Glasgow. The site in question (Figure 1.9) lies at the edge of an industrial area and borders a Local Nature Reserve. Both banks are soft and well vegetated and the channel type is therefore trapezoidal. Boat traffic is light (500 my). The environmental data pertaining to this site is shown in Table 3.11. Data from this site is used in the worked example of the invertebrate classification. Further details are given in section 2.9.

3.8.2 Calculating observed metric values

The raw biological data for this site is shown in Table 3.10.

For APB we take the mean drained weight of material on a grapnel as the basic figure. At the ten locations sampled at this site the mean biomass was 186.9g per grapnel. Employing the conversion constant of 0.108 this converts to a mean biomass of 186.9 x 0.108 = 20.2 $gDWm^{-2}$. However, at this site there also a number of free-floating or floating-leaved taxa that are not adequately sampled using the grapnel method so an additional allowance is required to cover the contribution by these taxa. This requires applying a biomass standard from published figures (see Table 3.1 in WFD61a) and correcting this to take account of the percentage cover of the various taxa. Thus, for *Lemna minor*, *Lemna trisulca* and *Potamogeton natans* the cover score of 1 is translated to 1% cover. Therefore for these species 1% of their biomass standard is required. For *Nuphar* and *Sparganium emersum* the cover score of 2 equates to 7% cover so for these two species a supplement equivalent to 7% of their biomass standard is required. The total biomass of floating-leaved species at this site is thus estimated to be 69.5 g DWm^{-2}. Taking the data from the grapnel sampling and the supplements based on the biomass standards gives a total APB for this site of 89.7 g DWm^{-2}.

For N_AQUA the Observed metric is simply a count of the number of taxa with an aquatic growth habit, in this case 7.

For MFI the metric is a weighted average of the MFI values given in Table 3.11, where the cover score of each taxon (on a scale of 1-5) provides the weighting factor. Hence the value of MFI for the site is:

(3 x 9.44 + 3 x 9.64 + 1 x 8.80 + 1 x 8.21 + 1 x 6.27 + 2 x 8.42 + 1 x 7.52 + 1 x 5.69 + 1 x 7.24 + 2 x 8.32 + 2 x 8.87) / (3+3+1+1+1+2+1+1+1+2+2) = 8.46

For N_EMERG the Observed metric is simply a count of the number of taxa with an emergent growth habit, in this case 4.

For REL_EMERG the survey reported that the average width of both reed fringes was equivalent to 10.3% of the channel width giving a value for REL_EMERG of 0.103.

Table 3.10. Environmental data for case study site at Lambhill, Glasgow, Forth & Clyde Canal.

	SEPA19
Canal Name	Forth and Clyde
Site Name	Lambhill
NGR	NS5863469702
Site code	
SEPA location code	121436
Channel type	Trapezoidal
Traffic (my)	500
Alkalinity (mg/L $CaCO_3$)	97.2
Altitude (m)	54
% Sand	0
% Silt-clay	50

Table 3.11. Raw macrophyte survey data for case study site at Lambhill, Glasgow, Forth & Clyde Canal, as sampled by SEPA in summer 2006.

Taxa	**biomass constants**	**cover score**	**Grapnel Biomass (g)**	**Inferred biomass ($gDWm^2$)**	**NAQUA**	**MFI**	**N_EMERG**	**Fringe width**
Elodea nuttallii		3			a	9.44		
Glyceria maxima		3				9.64	e	
Lemna minor	20	1		0.2	a	8.80		
Lemna trisulca	40	1		0.4	a	8.21		
Mentha aquatica		1				6.27	e	
Nuphar lutea	600	2		42	a	8.42		
Phalaris arundinacea		1				7.52	e	
Potamogeton natans	239	1		2.39	a	5.69		
Potamogeton trichoides		1			a	7.24		
Sparganium emersum	350	2		24.5	a	8.32		
Typha latifolia		2				8.87	e	
Total submerged			186.9	20.2				
				89.7	7	8.46	4	0.103

3.8.3 Calculating Expected metric values

The expected values for each of the five metrics are calculated using the functions in Table 3.4. Values are summarized in Table 3.12.

Thus, for APB in Scotland

Expected APB= 127.85 * EXP (-0.039 * sqrt (Boat traffic)) = 53.5 $gDWm^{-2}$

Thus for NAQUA, with a trapezoidal/asymetric channel

Expected NAQUA =11.16 * EXP (-0.008 * sqrt (boat traffic)) = 9.33
Applying an 80% correction for sites in Scotland = 7.47

Thus, for MFI,

Expected MFI = 5.124 +3.539 * Ln (log (alkalinity)) = 7.56
Where alkalinity is measured as $CaCO_3$ in mg/L = 7.23

Thus, for N_EMERG in Scotland

Expected N_EMERG with a trapezoidal/asymetric channel = 6

Thus, for REL_EMERG, with a trapezoidal/asymetric channel

Expected REL_EMERG = 0.315 * EXP (-0.00688 * sqrt (boat traffic))
= 0.27

Table 3.12. Summary of Observed and Expected metric values, EQI and EQR values and overall Ecological Potential for case study site at Lambhill, Glasgow, Forth & Clyde Canal.

Macrophyte metrics	Obs	Exp	EQI	EQR	Grp EQR
APB	89.7	53.5	1.68	1.00	0.96
N_AQUA	7	7.47	0.94	0.94	
MFI	8.46	7.56	0.63	0.51	0.51
N_EMERG	4	6	0.67	0.67	0.61
REL_EMERG	0.103	0.27	0.38	0.51	
average of three groups					0.69
Ecological Potential					Good

Obs=Observed; Exp=Expected; EQI=Ecological Quality Index; EQR=Ecological Quality Ratio; Grp EQR = Ecological Quality Ratio for discrete groups of metrics where pale blue refers to in-channel pressures, pale green to water quality and brown to bank-related pressures.

3.8.4 Calculating an EQI for biological metrics

For each metric the EQI is calculated simply by dividing the Observed value by the Expected value, except in the case of MFI. Thus;

EQI APB = 89.7 / 53.5 = 1.68

EQI N_AQUA = 7 / 7.47 = 0.94

EQI MFI = (8.46 – 10) / (7.56 – 10) = 0.63

EQI N_EMERG = 4 / 6 = 0.67

EQI REL_EMERG = 0.103 / 0.27 = 0.38

Values are summarised in Table 3.12.

3.8.5 Converting the EQI to an EQR

Before these metrics can be combined they must be converted into an EQR where the class widths of all metrics are the same. This means adopting a class system where each of the five possible classes of Ecological Potential occupies 0.2 units of the EQ gradient.

For the metrics N_AQUA and N_EMERG no change is needed to the EQI since the boundary between Maximum and Good Ecological Potential falls at 0.8 and all subsequent classes are therefore 0.2 units apart (Table 3.6).

For the metric MFI the class boundaries are at 0.85, 0.70, 0.55 and 0.40, i.e. the class width on the EQI scale is 0.15. This width therefore requires to be 'stretched' to 0.2 units so that the MFI EQI is directly comparable to the N_AQUA and N_EMERG EQI. Thus, we apply the following formula for a piecewise linear transformation:

=IF(EQI $_{MFI}$>=1,1,
IF(EQI $_{MFI}$>=0.85,((EQI $_{MFI}$-0.85)/(1-0.8))*0.2+0.8,
IF(EQI $_{MFI}$>=0.70,((EQI $_{MFI}$-0.70)/(0.85-0.70))*0.2+0.6,
IF(EQI $_{MFI}$>=0.55,((EQI $_{MFI}$-0.55)/(0.70-0.55))*0.2+0.4,
IF(EQI $_{MFI}$>=0.40,((EQI $_{MFI}$-0.40)/(0.55-0.40))*0.2+0.2,
IF(EQI $_{MFI}$<0.40,(EQI $_{MFI}$/0.40)*0.2))))))

Thus in the present case:

IF(EQI $_{MFI}$>=0.55,((EQI $_{MFI}$-0.55)/(0.70-0.55))*0.2+0.4 = 0.51

The same procedure is then applied to the other metrics. Thus, for EQI $_{APB}$ and EQI $_{REL_EMERG}$

=IF(EQI>=1,1,
IF(EQI>=0.75,((EQI-0.75)/(1-0.75))*0.2+0.8,
IF(EQI>=0.50,((EQI-0.50)/(0.75-0.50))*0.2+0.6,
IF(EQI>=0.25,((EQI-0.25)/(0.50-0.25))*0.2+0.4,
IF(EQI<0.25,(EQI/0.25)*0.4))))

Values for each metric in EQR terms are summarised in Table 3.12.

3.8.6 Combining EQRs into a value for the site

The individual EQRs shown in Table 3.13 belong to three basic families – those principally indicative of pressures associated with in-channel disturbance (shaded blue in Table 3.12, i.e. APB and N_AQUA), water quality (MFI), and those more responsive to bank-related disturbance (shaded brown in Table 3.13). The classification of the site is determined by the average of these three groups of metrics.

For the in-channel component the aggregated EQR is determined by taking the weighted average of EQR $_{APB}$ and EQR $_{N_AQUA}$ where EQR $_{N_APB}$ receives half the weight to reflect its inherent higher variability within the population of reference sites. Thus:

In-channel component = (EQR_{APB} x 0.5 +EQR_{N_AQUA}) / 1.5
= (1.00 x 0.5 + 0.94) / 1.5 = 0.96

For the water quality component we simply default to the EQR of the metric MFI (in this case 0.37), although it is recognised that other metrics will directly or indirectly also convey information about water quality.

For the bank-related component the aggregated EQR is determined by taking the weighted average of EQR_{N_EMERG} and EQR_{REL_EMERG} where the latter EQR receives half the weight to reflect its higher variability within the reference sites. Thus:

Bank-related component = (EQR_{N_EMERG} + 0.5 x EQR_{REL_EMERG}) / 1.5
= (0.67 + 0.5 x 0.51) / 1.5 = 0.61

The final EQR for the site is then determined by the average of these three groups of metrics (i.e. the average of 0.96, 0.51 and 0.61) which equals 0.69. This would classify this particular site in the middle of Good Ecological Potential. From the metrics assembled for this site we would conclude that the abundance and diversity of aquatic vegetation is largely what we would expect in an unimpacted site, however, the composition of the vegetation suggests enrichment by nutrients. In terms of the banks the extent of emergent vegetation is rather limited given the low boat traffic at this site, although still just within the range of Good Ecological Potential.

4 HYDROMORPHOLOGICAL ASSESSMENT

4.1 A hydromorphological perspective on Ecological Potential

The process of defining the ecological potential of canals and other AWBs requires assessment of hydromorphology as well as biology and physico-chemistry. Hydromorphology is a supporting variable, i.e. a variable that supports (or directly influences) the biological elements, its role in classification being to discriminate between Maximum and Good Ecological Potential. Where the hydromorphology is less than good it is assumed that this will be reflected in the values for dependent biological quality elements (i.e. a water body cannot fail to achieve its objectives based on hydromorphology alone). For heavily modified and artificial water bodies MEP in terms of hydromorphological elements is defined as:

> *The hydromorphological conditions are consistent with the only impacts on the surface water body being those resulting from the artificial or heavily modified characteristics of the water body once all mitigation measures have been taken to ensure the best approximation to ecological continuum.*

While hydromorphology standards are needed to discriminate between High/Maximum and Good Status/Potential, an understanding of the relationship between hydromorphology and biology in other (lower) classes is also central to defining those measures that are needed to improve the hydromorphology of a water body that is failing to meet its objective in biological terms.

The 'ecological continuum' could be interpreted in terms of longitudinal, lateral and vertical; dimensions. Disruption of the longitudinal continuum is almost a fundamental intention of canal construction, while lining of the channel to reduce water loss is likely to isolate the channel from the groundwater in many cases. However, the lateral connectivity between land and water across the riparian zone is prone to disconnection depending on the type of bank engineering employed on canals. Restoring lateral connectivity wherever possible should therefore be a key objective of mitigation measures.

The definition of MEP hydromorphology should ideally be a two-way process since at MEP biological quality elements must reflect the biology of the closest comparable water body type, given the physical conditions which result from the artificial or heavily modified characteristics of the water body. Clearly MEP hydromorphology cannot therefore be defined in a way that the biology associated with it directly contradicts MEP biology and similarly the MEP biology cannot be associated with a set of hydromorphological conditions that rarely match hydromorphology MEP.

The case of MEP hydromorphology for canals must reflect the inherent difference between a passively meandering lowland river (probably the closest natural analogue to a canal due to its static boundary) and what that water body would be allowed to look like if modified to the minimum level necessary to support navigation (or any other functions provided by canals such as water supply or flood defence). Assessments should therefore consider this minimum level of modification as the target, not a natural river itself. Thus, for example, the hydromorphology of a canal could clearly not be mitigated by removing locks to restore the natural gradient, or by introducing large woody debris to restore physical habitat variability since these would directly contravene the navigation function of canals. Canals also deviate well beyond navigable rivers in terms of

the absolute level of hydromorphological 'impact'. Thus, canals are usually small, straight, constrained channels of uniform depth and width. These features, combined with the age of channels and their mode of construction, preclude the use of many of the restoration techniques commonly advocated for engineered rivers (e.g. RSPB et al., 1994).

An approach to hydromorphological assessment of canals, which complements recent work on the identification of potential mitigation measures is discussed below.

4.2 Hydromorphology and mitigation

Mitigation measures are central to an alternative definition of hydromorphology at MEP for AWBs and HMWBs. Under this definition it is envisaged that desirable hydromorphological attributes are effectively the end product of appropriate mitigation measures. An alternative approach to defining MEP therefore rests upon identifying the specific mitigation techniques, which, if implemented, would, it is assumed, deliver hydromorphology and associated biology that was closest to that of the nearest equivalent natural water body given the physical conditions which result from the artificial or heavily modified characteristics of the water body. Those measures that would be likely to deliver substantial improvements could then be considered a minimal subset that would be needed to achieve GEP and these could be considered a core part of the programme of measures. Where these key mitigations could not be delivered (e.g. on grounds of cost, adverse environmental impact, or technical feasibility) an objective below GEP would be set. Measures that might only deliver a slight additional improvement to the biology associated with GEP could be considered as the basis for MEP. Oldmeadow et al., (2007) review mitigation measures suitable for canals and navigable rivers and shortlist measures needed to achieve MEP or GEP under different scenarios.

While this approach is attractive as a short term pragmatic solution it requires supporting information on potential restoration techniques and their ecological benefits. In general the evidence base linking mitigation action and ecological response is very weak. In WFD61a we reviewed a range of suitable mitigation measures for use in canals and recommended a number of approaches for improving understanding of the links between mitigation-hydromorphology and ecology. We also developed a prototype assessment approach that combined assessment of both hydromorphology and mitigation.

The approach developed below does not consider mitigation techniques. Rather it focuses on a definition of MEP in terms of the morphological forms and processes that would be found in a natural lowland river analogue, minus the impact associated with boat traffic. The list of mitigation measures needed for any particular WB is simply the means to reinstate process-form interactions that are currently missing. What is required from one canal to the next will vary.

4.3 Hydromorphological assessment

4.3.1 A two step approach to assessing hydromorphology on canals

A two step spreadsheet based tool has been produced as part of this project that will provide a standardized hydromorphological assessment of any canal site. The tool is provided separately as file canalHMassessv1.2.xls. The tool provided represents a considerably streamlined version of the tool developed in Phase 1 and it should be possible to complete the assessment of any site within ten minutes given a GIS, aerial photography, or preferably ground level photography of the site. In streamlining the hydromorphological assessment tool the components dealing specifically with mitigation have been removed as these were considered to render the tool too unwieldy. The tool now focuses on observable features; whether or not these are the product of deliberate mitigation is irrelevant.

Step A

Certain contextual information for a canal reach is required in order to define the hydromorphological features that it is reasonable to expect given the permitted use. This information is used to exclude features which fall into the following categories:

i. Overall negative impact on the permitted use (although feasible, the presence of a certain feature, or implementation of the mitigation necessary to create it, would directly impact on the use of a canal for navigation).

ii. Would not be practicable in that particular location (cost is not considered here but practicality takes into account health and safety limitations).

iii. Could not be achieved without 'significant' adverse impacts on the 'wider environment'. The wider environment incorporates natural features or built structures, which may have statutory protection. However, it is also includes aspects such as health and safety and the impact of the necessary works on the permitted use or the integrity of the water body. For example, it is not realistic to stipulate that the offside bank of a channel which runs on an embankment through a suburban area and carries a moderate to high boat traffic should be soft (i.e. not hard engineered) since this carries an unacceptably high risk of bank failure and subsequent flooding. It may be that the offside bank in this situation *is* soft and this will enhance the ecological potential of the reach, however, we could not *expect* it to be soft or to be re-engineered to *be* soft. Nevertheless, replacing the soft bank with a hard engineered bank in this situation could lead to deterioration in Ecological Potential unless compensatory hydromorphological features (e.g. trees on offside bank, low extent of linear moorings) were present or introduced.

In the above context the term 'significant' is taken to mean a lasting negative impact on use (e.g. boat traffic would be clearly restricted relative to the normal interannual variation in traffic at a site) or a prolonged impact on the status of natural features at a site, such that, for example, an SAC would be at risk of

failing to meet Favourable Conservation status if this were the only pressure. It is important to note that cost and technical feasibility are separated from the classification stage as they form part of the measures appraisal and objective setting process, to set an alternative objective or an extended deadline.

In the first stage of the assessment process it is necessary to determine if one or more of a key set of exemptions or 'constraining factors' apply to the reach to be assessed. Information on these will already be available through the process of determining the expected channel type referred to in section 2.4.1. The specific exemptions are listed in Table 4.1. If one or more of the exemptions listed applies to the reach being assessed the procedure then moves to answering the questions in Step B. If none of the exemptions apply the reach must first be assigned to one of nine possible combinations of traffic and channel width, as referred to in Table 4.2. Once the canal has been classified according to this matrix the questions in Step B must be followed.

Table 4.1. Specific exemptions to be considered in assessments of canal hydromorphology (i.e. default no longer applies and can be justified).

The channel is geologically constrained by solid rock
The channel is a designated Commercial Waterway
The channel performs alternative/additional functions (e.g. Flood defence or water supply)
The channel is elevated above the surrounding land on both sides by an embankment
The channel is located in an area dominated by urban, industrial or former industrial landuse
The offside bank is immediately adjacent to a railway or metalled road or there are formal vehicular access arrangements
The canal carries a traffic of >10000 boat movements per year
The canal has official Heritage designation of which the banks form part

Table 4.2. Matrix of boat traffic and channel size, to be used in assessing hydromorphology at sites where exemptions do not apply.

		Channel width (m)		
		<11	.11-15	15>
Boat traffic (my)	<2000	a	b	c
	2000-6000	d	e	f
	6000-10000	g	h	i

Step B

A set of criteria are used for defining the hydromorphological features of the reach. The weight given to each answer reflects an expert opinion on whether the hydromorphological feature in question reflects a minor or major deviation from the hydromorphology that would be expected given the level of use and channel dimensions (Table 4.3). The weight given to the answers depends on the classification of the site according to the matrix in Table 4.2. The following principles apply to the scores:

1. Negative weights apply when the recorded feature indicates a large level of deviation from reference hydromorphology which is likely to translate to a large impact on ecology. The more negative the weight the greater the deviation and its likely impact.
2. Positive scores are given to features that would not be expected under the given level of use (e.g. a soft engineered offside bank where one or more of the exemptions in Table 4.1 would apply) but which would serve to reduce the impact of negative features. Effectively these are 'bonus' marks.
3. Similar scores suggest that several features are interchangeable in the amount of deviation from reference hydromorphological conditions or the ecological impact they create.
4. Questions relating to features that are not relevant under the particular scenario described in Step A (i.e. could not be applied on grounds of technical feasibility, adverse impacts on use or the wider environment, or which would have a neutral effect if applied), are given a zero score.

For each criteria a short rationale and justification of the rules for assigning scores is given below.

Q1. Physical attributes of the offside bank.

In general it is assumed that the offside bank of a canal should not be hard engineered. However, the justification for having a soft offside bank decreases with increasing traffic; decreases with reducing channel width and decreases from rural to urban settings. A soft offside is not justifiable on embankments, or where access constraints exist (e.g. there is a requirement for vehicular access on the offside bank), or

where dependent protected environmental and heritage features occur. In the absence of other solutions there may be a requirement for hard engineering to reduce water loss by leakage. However, given the situations under which leakage is most likely, embankments are covered in the above statements, while on contour canals the towpath is generally on the downslope (i.e. where leakage is possible) and hard engineering of the towpath bank is allowed for.

In the absence of specific exemptions the greater the extent of hard engineering of the offside bank the greater the deviation from reference hydromorphology. When hard engineering utilises 'porous', rather than 'non- porous' materials, impacts are considered less due to the greater habitat support value of such substrates. Thus, porous materials are considered to include riprap, gabion mattresses, sand bags or un-pointed stone or brickwork compared to sheet steel, solid concrete or intact mortared brickwork. Use of hard structures with large interstitial spaces/cracks offers habitat for macroinvertebrates and crayfish and will help dissipate wave energy. It is therefore environmentally superior to hard reflective engineering such as concrete or steel piling.

If the offside bank is not hard engineered but has little or no vegetation cover due to bank erosion caused by boat passages or livestock poaching a negative score is also applied.

The WFD definition of MEP in terms of hydromorphology focuses on the ecological continuum. Consequently, in the case of canals, it is argued that a key element of hydromorphological assessment on canals should relate to the level of lateral connectivity between channel and riparian zone. This is reflected in the scale of the scoring for this question.

Q2. Tree cover on the offside bank

High tree cover can lead to shading of aquatic and emergent vegetation while large inputs of leaf litter may lead to DO stress, especially at high traffic. Conversely, a lack of tree cover leads to loss of habitat diversity at a corridor scale and at high traffic removes a significant source of organic matter for the aquatic ecosystem.

Q3. Physical attributes of the nearside bank

The expectation on all navigable waterways is that the nearside bank will be hard engineered to allow safe mooring etc. Where soft-engineering is in place this may compensate for other impacts, such as poor quality offside habitat, or may substitute for other forms of mitigation (e.g. bed stabilisation). Soft nearside banks are therefore only interpreted as a positive feature. The scores applied mirror those used for assessing the offside bank with the exception that all negative scores are set to zero since a soft nearside bank is treated as a bonus rather than an expectation.

Table 4.3. Canal hydromorphology assessment sheet showing scores for different attributes.

Exemptions apply? YES / NO		Offside bank									Nearside bank							Linear Moorings n/s			Hydrocompliance		Offline features		
		fully HE		% soft			fully soft		Overhanging trees		fully HE		% soft			fully soft		% of length affected			status		None	mainly HE	mainly SE
		piling	stone	1-10	.10-50	50-90	veg	bare	±absent	present	piling	stone	1-10	.10-50	50-90	veg	bare	<10	.10-25	>25	Pass	Fail			
		0	1	1	2	3	4	2	0	1	0	1	1	2	3	4	2	0	-1	-2	0	-2	0	1	2
	width (m)																								
Traffic <2000 my	<11	-4	-3	-3	-2	-1	0	-2	-1	0	0	1	1	2	3	4	2	0	-1	-2	0	-2	0	0	2
	11-15	-4	-3	-3	-2	-1	0	-2	-1	0	0	1	1	2	3	4	2	0	0	-1	0	-1	0	0	1
	15>	-4	-3	-3	-2	-1	0	-2	-1	0	0	1	1	2	3	4	2	0	0	-1	0	-1	0	0	1
Traffic 2000-6000 my	<11	-4	-3	-3	-2	-1	0	-2	-1	0	0	1	1	2	3	4	2	0	-1	-2	0	-2	0	1	2
	11-15	-4	-3	-3	-2	-1	0	-2	-1	0	0	1	1	2	3	4	2	0	0	-1	0	-2	0	1	2
	15>	-4	-3	-3	-2	-1	0	-2	-2	0	0	1	1	2	3	4	2	0	0	-1	0	-2	0	1	2
Traffic 6000-10000 my	<11	-3	-2	-2	-1	0	0	-1	-1	0	0	1	1	2	3	4	2	0	-1	-3	0	-3	0	2	3
	11-15	-3	-2	-2	-1	-1	0	-1	-2	0	0	1	1	2	3	4	2	0	-1	-2	0	-2	0	1	2
	15>	-4	-3	-3	-2	-1	0	-2	-2	0	0	1	1	2	3	4	2	0	-1	-2	0	-2	0	1	2

Exemptions are listed in Table 4.1. The scores associated with the features in a standard assessment reach are totalled. Negative scores indicate <GEP, scores of 0-1 = GEP, scores >1 = Maximum Ecological Potential (see Example in Table 4.4)

Key: HE = Hard engineered; SE = Soft engineered; n/s = nearside bank

Q5. Extent of linear moorings on nearside bank.

Linear moorings effectively reduce the channel width, therefore magnifying boat traffic impacts, eliminate habitat and result in increased disturbance of the bed during manoeuvring. Greatest negative impacts will occur in narrow channels where linear moorings will create the greatest relative reduction in effective channel width.

Q6. What is the hydrocompliance status of the channel?

Regular dredging is required to maintain sufficient draught. This reduces disturbance of the bed and associated turbidity/nutrient loading. Main benefits probably occur at moderate traffic. At high traffic dredging may favour the provision of silt-free areas for larger invertebrates. Dredging is also an effective form of vegetation management at low traffic to offset encroachment by emergent vegetation. Dredging may increase species richness of vegetation, especially over the short to medium term, but also carries a risk of promoting colonisation by invasive species.

This question requires access to assessments of hydrocompliance status which reflects the ratio of typical draught to mid channel depth. Hydrocompliance is assessed by BW in functional lengths of 1km based on a hydrographic survey that is carried out at least once every eight years. The cross-sectional profile is measured every 50m to a resolution of 0.1m and a Minimum Open Channel (MOC) standard appropriate to the typeof navigation is then applied to each profile. That profile is considered to pass if <5% of the MOC is unimpeded. The functional length is considered to achieve hydrocompliance (i.e. pass) if >70% of cross sections meet the minimum standard of <5% of MOC impeded. In the hydromorphological assessment a 'pass' status is expected since a 'fail' status indicates the potential for greater disturbance of the bed through boat passages than is appropriate for that type or level of boat traffic. Specific uncertainties in hydrocompliance assessment may require attention in interpreting this feature.

If hydrocompliance has not been formally assessed on the basis of systematic bathymetric mapping (e.g. on non-BW waterways) it is suggested that an alternative MOC is applied based on minimum open (i.e. unobstructed) channel width and minimum mid-channel depth. As a rough guide the standards in Table 4.4 can be applied and would normally indicate that two boats can pass each other safely, and without contacting the bed. It is considered very unlikely that hydrocompliance would not be formally assessed on designated commercial waterways.

Table 4.4. Suggested Minimum Open Channel standards to achieve hydrocompliance where formal assessments have not been undertaken.

Canal type	Minimum Unobstructed channel width (m)	Minimum mid-channel depth (m)
Narrow Canal	6	1.2
Broad Canal	8	1.5

Q7. Presence of offline features

Off channel moorings, including marinas, remove the need for in-channel moorings and reduce associated disturbance. They also provide refugia for young fish. Ability to isolate boat-related pollution incidents from the main channel may also be advantageous. When effective their value relative to the main channel increases with traffic. In general, hard engineering will reduce the value of such sites, especially at low traffic densities. Sideponds attached to lock flights would not be included within the assessment of offline features for reasons set out in 4.3.2.

4.3.2 Undertaking assessment in practice

The method is designed to provide rapid assessment and could probably be carried out in conjunction with biological sampling, with a negligible impact on the workload. To maximise information content and links between different components of classification it will be highly advantageous if hydromorphological assessment is undertaken at the same sites used for macrophyte survey or invertebrate sampling. To ensure reasonable coverage of a waterbody, each site to assess should be 500m in length and centred on the location of sites used for biological assessment. Thus it is envisaged that there will be up to three hydromorphological assessments per water body.

The timing of hydromorphological assessment is not critical but should ideally be undertaken in spring or summer to coincide with biological sampling and maximum boat traffic. Monitoring is only required once per monitoring cycle. When carrying out assessments it is critical that they are undertaken in sections of open channel and that operational features are avoided (such features should be avoided anyway during biological sampling). Thus, locks, tunnels and bridges and their approaches, wharves, licensed moorings, work yards etc should not appear in reaches to be assessed because they are an integral part of the permitted use. Including such features would be equivalent to assessing hydromorphological impacts of reservoir operation by surveying the dam wall. If the density of such features in a given water body is very high the number or size of lengths to be assessed may have to reduced.

4.3.3 Classifying ecological potential using HM assessments

The sum of the scores achieved by a reach reflects the answers to questions in Step B. The scores to the various questions are totalled. If the total for the site is negative then the site is considered to fail to achieve Good Ecological Potential because there is a predominance of features that indicate significant deviation from the expected hydromorphology (see Table 4.3). It is possible that 'legacy effects' could be invoked to account for a failure to achieve GEP but this can only be implemented retrospectively. When the total score for a site is 0 or 1 it is assumed that the site achieves GEP because there is a balance of positive and negative features or no strongly negative features. To achieve MEP the score must be >1 which indicates a predominance of the hydromorphological features that are expected given the expected level of use and an absence of other impacts. At this point it is assumed that increasing the score further, e.g. by

implementing certain mitigations to alter other hydromorphological features, would deliver only relatively slight ecological gains.

4.4 Example of hydromorphological assessment

Consider the site used as the worked example in Chapters 2 and 3. A map and photograph of the site is reproduced in Figure 4.1 for reference purposes.

Figure 4.1. Aerial image (Google Earth, 2009) and ground-based photograph for site used in worked example, Lambhill, Glasgow, Forth & Clyde Canal.



The first step of assessment requires testing the exemptions in Table 4.1. None of the possible listed exemptions apply to this particular site. In the second step the canal must be classified according to a matrix of traffic and channel size. Traffic at the site in question is 500 movements per year and the channel is 14m wide. Therefore the site belongs in cell b of the matrix in Table 4.2.

As described above, Step B of the assessment covers a set of criteria for assessing the hydromorphology within each reach. The conditions that best describe the site in question must be identified. Table 4.5 contains the responses to these questions for the example site. This table illustrates the score associated with the response given. Thus, it is straightforward to identify the criteria where a different response would give the greatest change in numerical value. From this, those mitigations that might deliver the greatest improvement in ecological status can be inferred. Equally, those modifications expected to cause the greatest change in ecological quality can also be identified.

Referring to Table 4.5 the example site has a fully vegetated soft offside bank. This feature does not attract a positive score since it is a feature we would expect of a canal of this size and level of use and where no exemptions apply. Similarly, trees are clearly present on the offside bank but again are a feature that would be expected of a canal with the stated characteristics. In terms of the nearside (i.e. towpath) bank, reference to the site photograph and local information indicates

that a quarter of the bank is vegetated by a reed fringe. In practice this is a 'bonus' feature because we would nominally expect the nearside bank to be hard engineered. An even larger bonus score would have been achieved if the nearside bank was vegetated for its entire length. The nearside bank does not support linear moorings. In the event that it supported extensive linear moorings a negative score would have been obtained to reflect the potential for increased disturbance caused by boat passages and loss of vegetated riparian zone habitat. A check on the hydrocompliance of this reach via a GIS which links the BW hydrocompliance register to biological sampling sites indicates that this reach fails. This attracts a negative score because it is then considered that the impact of a given level of traffic is increased due to the lack of sufficient draught which will increase disturbance to the bed. There are no offline features in this site (the hydrological connection to Possil Loch lies to the north of the example reach). Totalling the scores for the different attributes gives an overall value of 1. In this case the partially vegetated nearside bank effectively compensates for the hydrocompliance failure. By achieving a score of 0-1 the site would therefore be considered to achieve GEP but not MEP. The latter would require a hydrocompliance pass, and/or a greater extent of reed fringe on the nearside bank, or soft engineered offline features. Both of the latter occur at other locations along this particular waterway.

If we reconsider the example site, but imagine that one of the exemptions in Table 4.1 applied, this site would then achieve a score of 5 and would be classified at MEP, mainly by virtue of the extent of vegetated nearside and offside bank.

Application of this assessment protocol to sites in Scotland and Ireland where biological sampling was undertaken in 2006 indicates that all sites would be classified at either GEP or MEP. This is in line with the generally high quality of these sites based on assessments using invertebrates and macrophytes. The tool could potentially also be applied remotely to sites in England & Wales that were used for the biological assessment reported in Chapters 2 and 3, but since there are approximately 500 such sites such testing was outwith the scope of this project.

Table 4.5. Example site showing completed hydromorphology assessment sheet for worked example site. Relevant attributes and their scores are highlighted.

Exemptions apply?		Offside bank									Nearside bank							Linear Moorings n/s			Hydrocompliance		Offline features		
		fully HE		% soft			fully soft		Overhanging trees		fully HE		% soft			fully soft		% of length affected			status		None	mainly HE	mainly SE
YES		piling	stone	1-10	.10-50	50-90	veg	bare	±absent	present	piling	stone	1-10	.10-50	50-90	veg	bare	<10	.10-25	>25	Pass	Fail			
NO		0	1	1	2	3	4	2	0	1	0	1	1	2	3	4	2	0	-1	-2	0	-2	0	1	2
	width (m)																								
Traffic <2000 my	<11	-4	-3	-3	-2	-1	0	-2	-1	0	0	1	1	2	3	4	2	0	-1	-2	0	-2	0	0	2
	11-15	-4	-3	-3	-2	-1	0	-2	-1	0	0	1	1	2	3	4	2	0	0	-1	0	-1	0	0	1
	15>	-4	-3	-3	-2	-1	0	-2	-1	0	0	1	1	2	3	4	2	0	0	-1	0	-1	0	0	1
Traffic 2000-6000 my	<11	-4	-3	-3	-2	-1	0	-2	-1	0	0	1	1	2	3	4	2	0	-1	-2	0	-2	0	1	2
	11-15	-4	-3	-3	-2	-1	0	-2	-1	0	0	1	1	2	3	4	2	0	0	-1	0	-2	0	1	2
	15>	-4	-3	-3	-2	-1	0	-2	-2	0	0	1	1	2	3	4	2	0	0	-1	0	-2	0	1	2
Traffic 6000-10000 my	<11	-3	-2	-2	-1	0	0	-1	-1	0	0	1	1	2	3	4	2	0	-1	-3	0	-3	0	2	3
	11-15	-3	-2	-2	-1	-1	0	-1	-2	0	0	1	1	2	3	4	2	0	-1	-2	0	-2	0	1	2
	15>	-4	-3	-3	-2	-1	0	-2	-2	0	0	1	1	2	3	4	2	0	-1	-2	0	-2	0	1	2

Classification of Ecological Potential	
>1	Max
0-1	Good
<0	<Good

Sum of highlighted Scores
1

Exemptions are listed in Table 4.1.
Key: HE = Hard engineered; SE = Soft engineered; n/s = nearside bank

5 CLASSIFICATION OF ECOLOGICAL POTENTIAL BASED ON PHYSICO-CHEMICAL VARIABLES

5.1 Introduction

Classification of Ecological status (or Potential) requires the joint consideration of the status of biological and physico-chemical quality elements with overall status being defined by the poorer of these two groups. In the case of artificial and Heavily Modified Water Bodies the specific physico-chemical elements to be considered are defined by the closest natural water body type. For canals this implies transparency, thermal conditions, oxygenation, salinity, nutrient condition and other pollutants. Annex VIII (indicative List of Main Pollutants) includes two sub-elements that are particular relevance to canals;

- 'Materials in suspension', - a pollutant that will directly reflect boat traffic, and
- 'Metals and their compounds', - these may accumulate in canal sediments where there is a history of pollution from industrial point sources.

Physico-chemical elements are considered to support the biology and Environmental Quality standards need to be proposed that are compatible with this supporting relationship. These standards form targets which programs of measures must be designed to deliver.

The results below are based on an analysis of relationships between macroinvertebrates and ammonia, BOD, and chloride and between macrophytes and SRP, TON and TSS. Other relationships assessed were non-significant and therefore could not be used to suggest standards. At present there is insufficient data to undertake a meaningful analysis of the relationship between the biology and other physico-chemical variables.

5.2 Water quality associated with reference/MEP conditions and Good/Moderate boundary as inferred from invertebrates

Water quality data were derived from a compilation of datasets including the environment agencies dataset collated for the present project and the PSYM database in which water quality data were also mainly derived from Environment Agency data. These data were integrated with data from sites in Scotland and Ireland surveyed in 2006. In the case of the PSYM database invertebrate sampling locations were chosen as far as possible to coincide with Environment Agency water chemistry sampling stations. In the dataset collated for the present project, water chemistry and invertebrate sampling locations were matched as far as possible.

Water quality associated with reference/MEP conditions was defined by modelling the relationship between a number of chemical determinands and the EQI values for each inverterbrate metric independently, and the overall invertebrate EQR. The appropriate class boundaries for each metric were then fitted to the models to derive the corresponding values for each chemical parameter at the class boundary.

Note that in all cases the site database is comparatively small (250-300 sites with matched invertebrate biology and physico-chemistry) so results should be treated with some caution.

5.2.1 Ammonia (as N)

The best model was obtained using the combined invertebrate classification (i.e. as defined by all metrics). A model based on ASPT was slightly less significant, while N_FAM EQI was the only other independent metric to be correlated with ammonia concentrations.

Taking the results from the full classification model the mean ammonia concentration at MEP was 11 ug/L, rising to 84 and 157 ug/L at the Maximum/Good and Good/Moderate boundaries respectively (Table 5.1). Environmental standards for ammonia in rivers are usually based on 90th percentiles but the limited availability of data for canals makes this approach more vulnerable to outliers. However, the mean concentrations of ammonia in canals at the upper class boundaries are compatible with current standards for lowland rivers based on 90th percentiles.

Table 5.1. Indicated mean concentrations of ammonia (µgl^{-1}) at the Maximum/Good and Good/Moderate EP boundaries based on significant relationships between ammonia and different invertebrate metrics

Metric	Ammonia model	r^2	Max	Max/Good	Good/Mod
all	y = -0.3633x + 0.3746	0.064	*11*	*84*	*157*
N_FAM	y = -0.1768x + 0.2908	0.032	114	149	185
ASPT	y = -0.648x + 0.7591	0.058	111	163	215

In the models y = the modelled ammonia concentration and x = the EQI of the relevant metric or overall classification

5.2.2 Biochemical Oxygen Demand

The best model was obtained using the combined invertebrate classification (i.e. as defined by all metrics). All independent metrics EQI values were significantly correlated with BOD, most notably ASPT. The standards derived using the full invertebrate tool suggest that on average BOD of 2.1 mg/L is typical at Maximum Ecological Potential for canals. At the upper class boundaries, BOD standards of 2.6 and 3.0 mg/L BOD would be appropriate to support invertebrate biology at Maximum or Good Ecological Potential respectively (Table 5.2).

BOD standards are normally based on 90th percentiles and take account of phsyiological requirements of fish. The BOD standards for cyprinid dominated lowland rivers of 4 and 5mg/L are the High/Good and Good/Moderate ecological status boundaries respectively appear to be compatible with the results of the present analysis for canals which indicates equivalent standards of 2.6 and 3 mg/L.

Dissolved Oxygen (DO) concentration or % oxygen saturation are often used as the basis of environmental standards for oxygen in rivers. In the case of canals we were uanble to establish a significant relationship between macroinvertebrate metrics and these determinands. It is likely that there is high uncertainty in mean values of DO and % saturation in canals when compared to rivers because of the strong influence of water temperature and time of day on measured values. In general the relevance of DO or % saturation standards in canals compared to

rivers is also questionable given the very sluggish and stable flows and pronounced influence of natural biological processes on oxygen availability which suppress the range of variation in DO and saturation.

Table 5.2. Indicated values for BOD (mgl^{-1}) at the Maximum/Good and Good/Moderate EP boundaries based on significant relationships between BOD and different invertebrate metrics

Metric	BOD model	r^2	Max	Max/Good	Good/Mod
all	y = -2.3508x + 4.4499	0.086	*2.10*	*2.57*	*3.04*
N_FAM	y = -1.2768x + 4.0212	0.054	2.74	3.00	3.26
ASPT	y = -3.5416x + 6.3267	0.060	2.79	3.07	3.28
EPT	y = -0.7941x + 3.4847	0.049	2.69	2.96	3.22
COL+BUG	y = -0.6522x + 3.3802	0.037	2.73	2.95	3.16
MOLL+ODON	y = -0.7219x + 3.5265	0.035	2.80	2.99	3.17

In the models y = the modelled BOD and x = the EQI of the relevant metric or overall classification

5.2.3 Chloride

The best model was obtained using the EPT metric although the combined invertebrate classification (i.e. as defined by all metrics) was almost as good and indicated more precautionary standards. As with BOD all independent metric EQI values were significantly correlated with chloride. The standards derived using the full invertebrate tool suggest that on average chloride concentrations of 44 mg/L is typical at Maximum Ecological Potential for canals in general. At the upper class boundaries, chloride concentrations of 52 and 60 mg/L would be appropriate to support invertebrate biology at Maximum or Good Ecological Potential respectively (Table 5.3). It should be noted that concentrations much lower than these would be required in canal water bodies in the Scottish highlands but there is insufficient data to specify these concentrations. Note that chloride concentrations may also be elevated in certain locations due to normal operation (e.g. back pumping of sea water). Consequently the suggested standards for chloride may not be applicable to some short coastal routes.

Table 5.3. Indicated values for chloride (mg/L) at the Maximum/Good and Good/Moderate EP boundaries based on relationships between chloride and different invertebrate metrics

Metric	Chloride model	r^2	Max	Max/Good	Good/Mod
all	y = -39.502x + 83.288	0.079	*43.8*	*51.7*	*59.6*
N_FAM	y = -17.062x + 72.5	0.033	55.4	58.9	62.3
ASPT	y = -55.191x + 110.57	0.047	55.4	59.8	63.1
EPT	y = -18.804x + 70.613	0.086	51.8	58.2	64.4
COL+BUG	y = -13.239x + 66.993	0.054	53.8	58.3	62.6
MOLL+ODON	y = -14.65x + 69.69	0.047	55.0	58.7	62.4

In the models y = the modelled chloride concentration and x = the EQI of the relevant metric or overall classification

5.3 Water quality associated with reference/MEP conditions and Good/Moderate boundary as inferred from macrophytes

Water quality data was derived from the environment agencies dataset collated for the present project, supplemented, where appropriate by spot sample data. As emphasised previously the spatial and temporal match between water chemistry and biology is poor and the sample size is small. Consequently the outputs of this analysis, although intuitively reasonable, should be interpreted with some caution.

The approach used to identify water quality associated with MEP and GEP for macrophytes was dictated by the limitations of the data. Supporting chemistry was defined based on the global relationship between biological metrics and physico-chemical variables, reading off the values associated with the EQI associated with each particular class boundary. This is the same approach that was used in the LEAFPACS project and is most appropriate when the size of a dataset is relatively small and the application of an upper percentile approach (as recommended by UKTAG (2006)) is therefore very vulnerable to outliers.

Of those water quality parameters for which data were available in the environment agencies database macrophyte metrics were poorly or unrelated to ammonia, dissolved oxygen concentration, BOD and heavy metals. The analysis presented here therefore focuses on those determinands that are likely to be of greatest direct relevance to macrophytes, viz, phosphorus (as SRP), nitrogen (as TON) and total suspended solids (TSS).

5.3.1 Soluble Reactive Phosphorus

Figure 5.1 illustrates the global relationship between SRP and the macrophyte fertility metric (MFI) in a large UK dataset on river vegetation. This relationship forms the evidence base for the use of this metric for assessing fertility pressures in canals. Overlaid on this diagram is the available data for canals. This is vastly inferior in terms of data availability (2900 matched surveys on rivers cf 200 on canals) and it is unsurprising that the relationship is so much weaker, yet it is similar in form. Notably for vegetation with metric values in the range 7-8.5 (i.e. the core vegetation of both canals and lowland, base-rich rivers) the associated SRP values are more than 6 fold higher in rivers than canals (285 cf 45µg l^{-1}). This probably reflects a higher density of point source inputs from STWs in the case of rivers, coupled, in canals, with high uptake rates of nutrients from the water column by macrophytes and high rates of loss of silt-bound nutrients to the sediment under stagnant flow conditions. Ireland is notable in having canals with extremely low SRP concentrations (5-20 µg/L) relative to the vegetation present. Many Irish canals could be considered almost analogous to marl lakes in terms of their catchment chemistry and are distinct from canals in Great Britain in this respect. Calcium concentrations are so high in Irish waterways that phosphorus will be readily lost from the water column by co-precipitation with calcium. This generates very low SRP concentrations in the water column, although the sediment, the main source of phosphorus for most macrophytes, may remain relatively fertile.

Taking the canal-specific model in Figure 5.1 it is possible to predict the SRP concentrations that would be necessary to support the ecology at a particular status for a given alkalinity. Thus the reference site model in section 3.4.4 predicts an MFI score under reference conditions of 8.07 for an alkalinity of 200 mg/L $CaCO_3$. Taking EQR values of 0.85 and 0.70 as the Maximum/Good and

Good/Moderate EP boundaries respectively for this metric, the MFI values and the SRP concentrations associated with these thresholds can be calculated from the model in Figure 5.1. This suggests values of 64 and 106 µg/L SRP at the Maximum/Good and Good/Moderate EP boundaries respectively, for this level of alkalinity. These values are independent of boat traffic. Values for a range of other conditions are given in Table 5.4. It should be noted that these are broadly in line with the values proposed for rivers. Irish canals are excluded from this analysis and it is suggested that the most appropriate standards for these waterways are those for marl lakes. Willby (2005) analysed nutrient-vegetation relationships in marl lakes for the LEAFPACS project and proposed that standards of 10 and 25 µg/L Total Phosphorus were appropriate for the H/G and G/M boundaries in such lakes.

Figure 5.1. Global relationship between annual mean Soluble Reactive Phosphorus (SRP) concentrations and macrophyte fertility index (MFI) comparing rivers and canals.

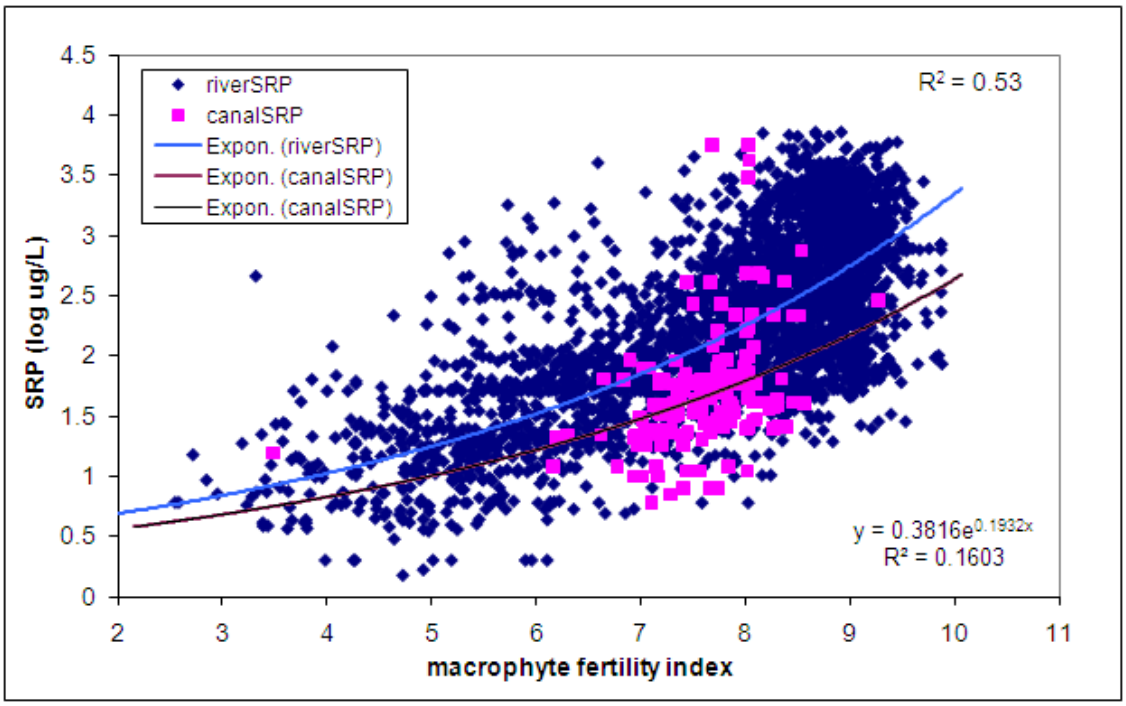


Table 5.4. Indicated values for SRP (µgl^{-1}) and TON (mgl^{-1}) at the Maximum/Good and Good/Moderate Ecological Potential boundaries based on their global relationships with MFI. Note that the SRP values cannot be applied to Irish canals.

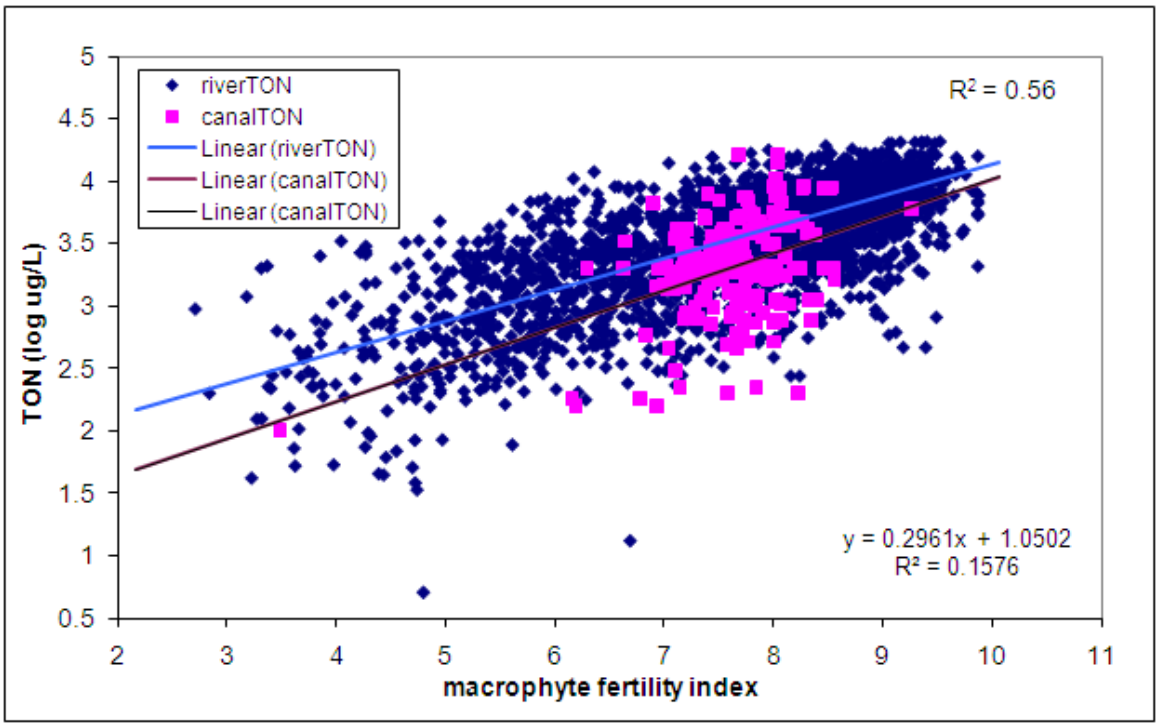
Alkalinity (mg/L)	Soluble Reactive P (ug/L)		Total Oxidised N (mg/L)	
	MEP/GEP	GEP/Mod EP	MEP/GEP	GEP/Mod EP
10	10	22	0.4	1.0
50	29	56	1.3	2.5
100	44	78	2.0	3.2
150	55	94	2.4	3.7
200	64	106	2.8	4.1
250	73	117	3.0	4.4

5.3.2 Total Oxidised Nitrogen (TON)

Figure 5.2 illustrates the form of the global relationship between MFI and annual mean TON in canals following the format of Figure 5.1. In the range of MFI values 7-8.5 (see above) the mean TON concentrations in rivers are three fold higher than those found in canals (6.0 cf 2.0 mg/L). However, Irish canals do not differ from canals in Great Britain with respect to TON concentrations to the same extent way that they do for SRP.

Taking the canal-specific model in Figure 5.2 it is possible to predict the TON concentrations that would be necessary to support the ecology at a particular status for a given alkalinity. Following the approach described above for SRP this suggests that for an alkalinity of 200 mg/L $CaCO_3$ standards of 2.8 and 4.1 mg/L TON at the Maximum/Good and Good/Moderate EP boundaries are appropriate in a canal. These values are independent of boat traffic. Values for a range of other conditions are given in Table 5.4. These standards are comparable with those indicated for macrophytes in rivers by the LEAFPACS project

Figure 5.2. Global relationship between annual mean Total Oxidised Nitrogen (TON) concentrations and macrophyte fertility index (MFI) comparing rivers and canals.



5.3.3 Turbidity and TSS

As illustrated in Figure 5.3 transparency in canals, expressed as turbidity (NTU) or suspended solids (TSS), is highly dependent on boat traffic. Standards therefore need to take this underlying relationship into account given that traffic is a permitted environmental pressure. Transparency decreases until about 3000 my, whereafter the rate of decrease steepens. The relationship plateaus at about 8000 my. It must be emphasised that these relationships are based on spot

measurements of turbidity and TSS collected at the same time as macrophyte surveys. Since surveys were undertaken between 10-16.00 hours in the months June-September which coincides with the period of peak boating activity this implies that the turbidity values are likely to be close to their maximum. Roughly speaking these measures of TSS or NTU are likely to be comparable to the 90th percentile of monthly measurements which is often used to set standards for other determinands (e.g. ammonia or Dissolved Oxygen).

In terms of relationships with metrics APB and N_AQUA are strongly related to measures of transparency in canals. Using the relationships developed in 3.4 it is possible to calculate the values of APB and N_AQUA that would be expected at MEP and at the boundary between Good and Moderate Ecological Potential at different boat traffic (Table 5.5). Using these values it is then possible to interrogate the relationship between these metrics and turbidity (Figures 5.4 and 5.5) to derive the values of transparency measures that would typically needed to support the biology at a given status at a given boat traffic. It is useful to compare predictions for AQUA_B and N_TAXA since one would expect the former to be more sensitive (abundance will decline before species are actually lost).

Figure 5.3. Relationships between boat traffic and spot measured TSS or tubidity (as NTU) at height of annual boat activity in canals (data: N.Willby). Relationships are modelled using locally weighted scatter plot smoothing (LOWESS).

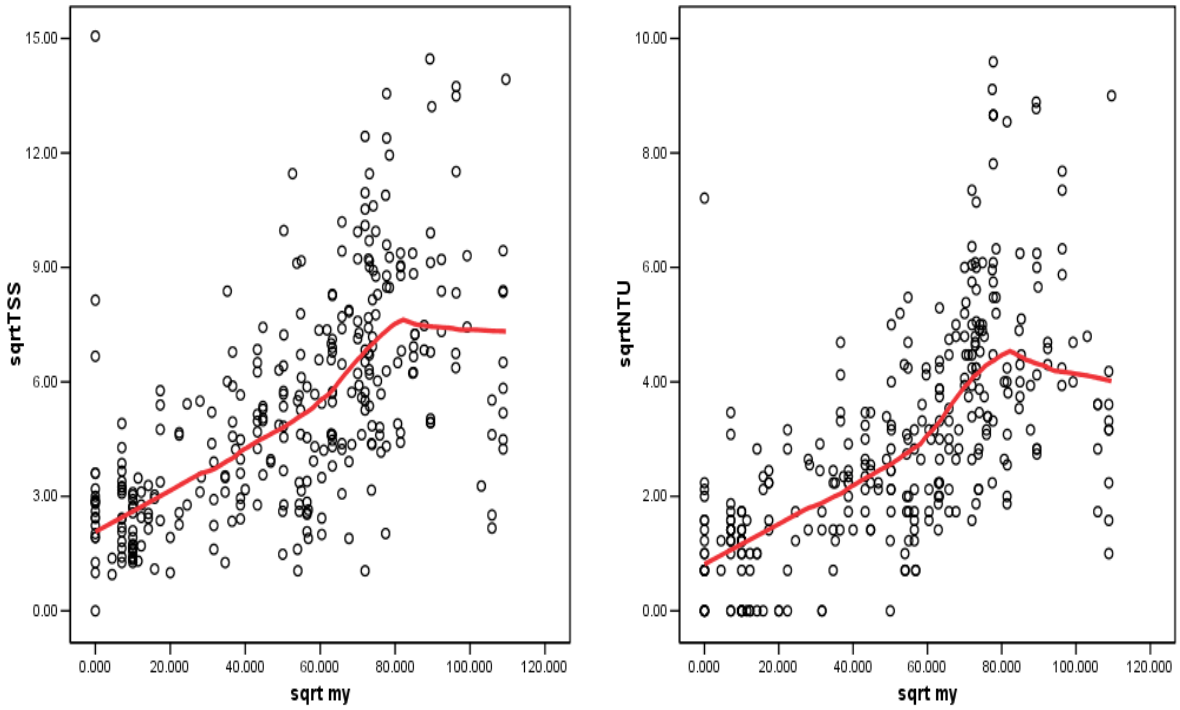
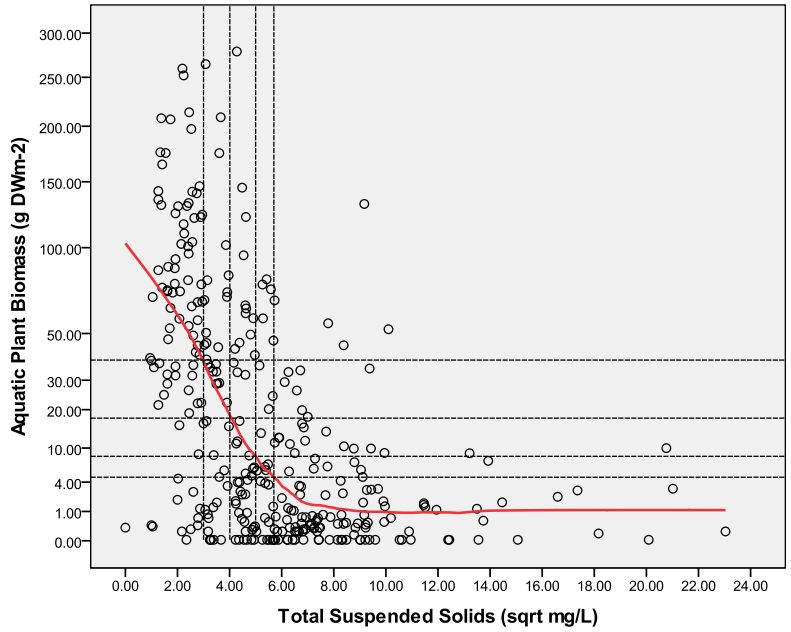


Figure 5.4. Relationships between APB and measures of transparency in canals. The relationship is fitted by LOWESS. The dashed lines are used to indicate the TSS or turbidity associated with the values of APB that indicate the G/M thresholds at four contrasting boat traffic densities given in Table 5.6.



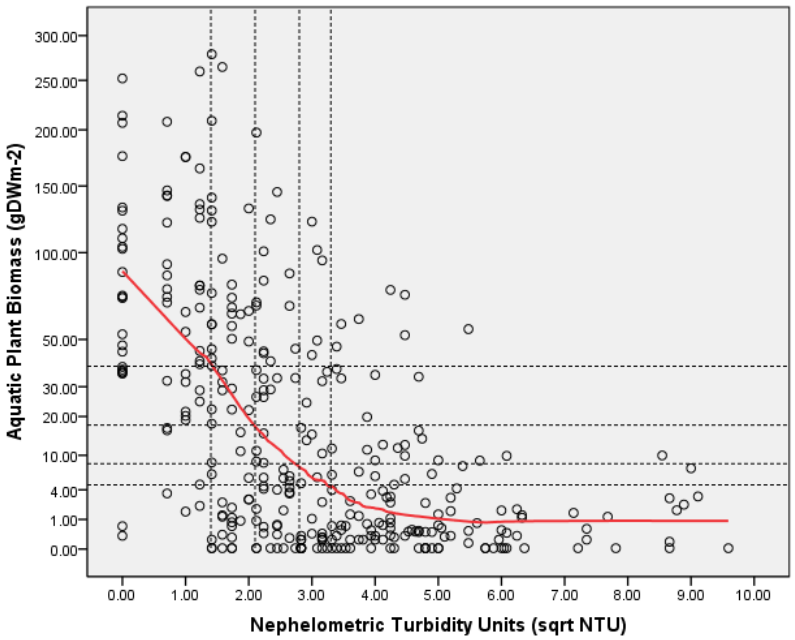


Figure 5.5 Relationships between N_AQUA and measures of transparency in canals. The relationship is fitted by LOWESS. The dashed lines are used to indicate the TSS or turbidity associated with the values of N_AQUA that indicate the G/M thresholds at four contrasting boat traffic densities given in Table 5.6.

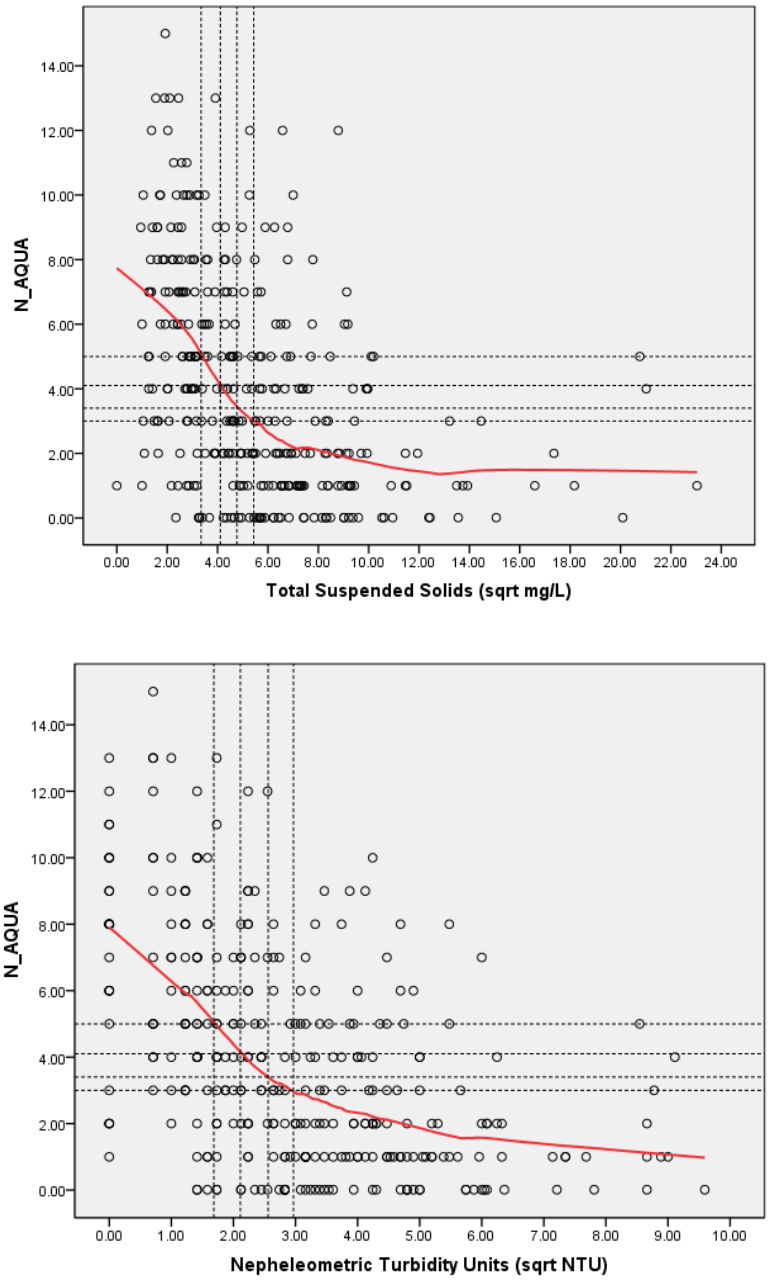


Table 5.5. Aquatic plant biomass and species richness expected at Maximum Ecological Potential and GEP/MEP boundary in canals at varying boat traffic densities.

Traffic (my)	Aquatic Biomass (APB)		Aquatic species no. (N_AQUA)	
	MEP	G/M*	MEP	G/M**
1000	75.1	37.5	8.3	5.0
3000	35.0	17.5	6.8	4.1
6000	16.5	8.3	5.7	3.4
9000	9.3	4.7	5.0	3.0

*based on EQR = 0.5; ** based on EQR = 0.6

Table 5.6. TSS (mg/L) and turbidity (as NTU) associated with threshold values of AQUA_N and AQUA_B at the G/M boundary at four contrasting boat densities.

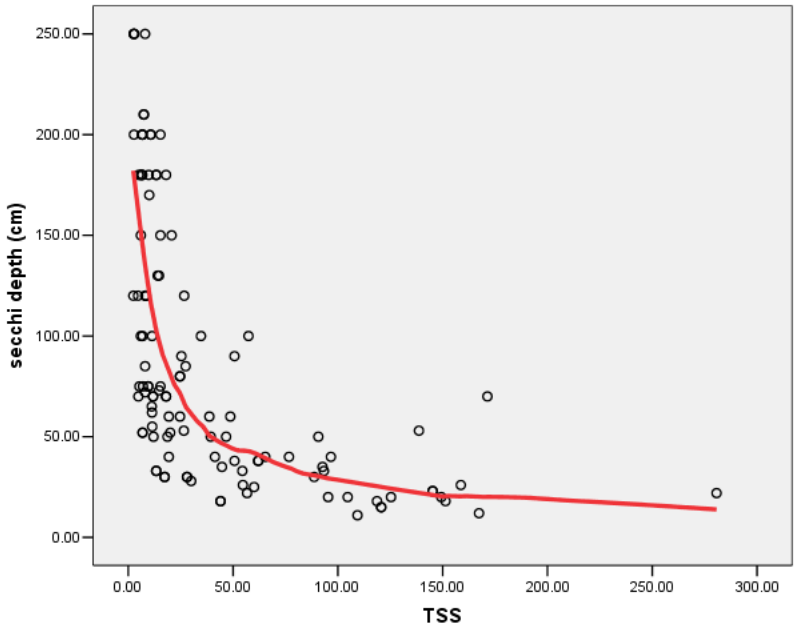
Traffic	APB		N_AQUA	
(my)	TSS	NTU	TSS	NTU
1000	9	2	11	3
3000	16	4	17	4
6000	25	8	23	7
9000	32	11	29	9

Based on this approach Table 5.6 summarises the values of TSS and turbidity that would be expected at the boundary of Good and Moderate Ecological Potential (i.e. the values that would be expected to support the biology at this boundary). Similar standards apply in the case of biomass and species richness. It should be noted (and is apparent from Figures 5.4 and 5.5) that there is considerable unexplained variation in the relationship between macrophyte metrics and turbidity or suspended solids. Consequently the values given in Table 5.6 are only indicative and have significant uncertainty associated with them. On the basis that the unexplained variation is smaller in the case of TSS v biology relationships the values for TSS in Table 5.6 are to be preferred over NTU as the TSS values can be stated with greater certainty.

Because previous research has been concerned partly with the relationship between traffic, turbidity and suspended solids, simpler and quicker measurements of transparency, such as secchi depth have often not been made.

However, this data was collected as part of the original PSYM study and it is recommended that secchi disc measurements are used in the field as a simple and direct measure of transparency. Using the relationship shown in Figure 5.6 it is possible to ascertain that a TSS value of 32 mgl^{-1}, which forms the value expected to support macrophyte biomass at the G/M boundary at 9000 my, would typically be associated with a secchi depth of c.60cm. Attributes which increase the risk that TSS values exceed 32 mg/L (e.g. linear moorings, narrow channel, limited draught, unvegetated soft margins) will therefore increase the risk that macrophytes fail to achieve Good Ecological Potential.

Figure 5.6. Relationship between TSS and secchi depth in canals (PSYM dataset)



5.3.4 Chlorophyll

WFD 61a (Willby et al., 2008) reviewed the data on chlorophyll concentrations in canals and the relevant literature for large rivers and shallow lakes. Although chlorophyll is strictly an attribute of a biological quality element (i.e. phytoplankton) it was suggested that, for convenience, in the case of canals chlorophyll would be best viewed as part of a set of physico-chemical quality elements because the best that could be hoped for from the available data was a set of standards.

Willby et al. (2008) illustrated a significant positive relationship between boat traffic and chlorophyll. On this basis a simple fixed threshold of chlorophyll concentration at the upper class boundaries of ecological potential for all canals could not be justified. Different standards would also be necessary to reflect underlying baseline differences in the fertility of different canals, as reflected for example in their alkalinity. On this basis, the standards in Table 5.7 were

proposed. For these purposes Irish canals are assigned to the moderate alkalinity category due to their propensity for co-precipitation of phosphorus which leads to very low water column fertility. Both the Scottish Highlands canals (Caledonian and Crinan) would be assigned to the lowest alkalinity band.

Table 5.7. Proposed upper limits of annual mean chlorophyll (ug/L) in canals needed to achieve Good Ecological Potential. Values for HG and GM boundaries for the nearest equivalent lake types are also given.

Alkalinity/Traffic	Low-Moderate (<3000 my)	Moderate-High (>3000 my)	UKTAG lake standards	
	Mod-High plant cover	Low-Moderate plant cover	HG	GM
High (> 1meq/L)	15	30	10-13	22-29
Moderate (0.2-1meq/L)	10	20	8-11	15-19
Low (<0.2 meq/L)	5	N/A	4-6	7-11

5.4 Specific pollutants

The WFD specifies nine families of specific pollutants, including various persistent organo-toxins. Of particular relevance to canals are the family metals and their compounds. The list of specific pollutants includes copper, iron and zinc which are likely to be common constituents of industrial discharges, as well as of diffuse pollution in urban areas. Other, potentially more hazardous metals such as cadmium, mercury, lead and nickel are treated as priority substances and used to define surface water chemical status.

For all specific pollutants UKTAG has advised environmental quality standards. These are based on the long term (annual mean) and short-term (Maximum Admissible Concentration). The long-term standards will be used for classification while failure of the Maximum Admissible Concentration will trigger additional investigation and monitoring. The approach to use of specific pollutants in classification is that standards should protect against adverse effects on biological quality elements. Consequently, if a standard is not exceeded it is assumed that a water body can potentially achieve high or good ecological status or potential, with respect to that standard. In terms of canals it is advised that the recommended standards should be transferred and that compliance with these standards should be a necessity for a canal water body to achieve GEP.

5.5 Summary of thresholds for selected physico-chemical parameters at High/Good and Good/Moderate boundaries

Water quality boundaries based on macroinvertebrates and macrophytes are summarised in Table 5.8. The environment agencies value is given first, followed by the PSYM database value. Comparisons are included with the standards proposed for lakes and rivers in the UKTAG Environmental Standards report (UKTAG, 2006).

Note that because of small sample sizes for the canal data there is a considerable degree of uncertainty about these values. The values for lakes and rivers are typically based on the 90th percentile of the values associated with a given class in a given lake or river type where the population ranges from hundreds to thousands of data values. Clearly this approach is vulnerable to outliers when applied to much small datasets such as exist for canals. The values considered within such analyses are typically some upper or lower percentile of the measurements obtained at a site based on monthly sampling over several years.

Table 5.8. Summary of water quality parameters at Maximum/Good and Good/Moderate Ecological Potential boundaries

	CANALS		UKTAG (RIVERS)	
Determinand	Max/Good	Good/ Moderate	High/Good	Good/Moderate
Ammonia mg/L[1]	0.08	0.16	0.3 [1,6]	0.6 [1,6]
BOD mg/L[1]	2.6	3.0	4 [6,7]	5 [6,7]
Chloride mg/L [1]	52	60	na	na
Suspended solids mg/L [2,3,4]	1-20	9-32	na	na
SRP µg/L [2,5]	64	106	50 [8,9]	120 [8,9]
TON mg/L [2,5]	2.8	4.1	na	na

1 based on macroinvertebrates

2 based on macrophytes

3. summer maxima

4. dependent on traffic

5. dependent on alkalinity, values presented are for 200 mg/L CaCO3

6. based on 90th percentile

7. based on fish

8. 90th percentile of annual mean of population of Good or High status sites

9. based on diatoms

6 COMBINING CLASSIFICATIONS INTO AN OVERALL ASSESSMENT OF ECOLOGICAL POTENTIAL

6.1 Introduction

Classification of the ecological status of water bodies requires the integration of information from different groups of quality elements. The different groups of quality elements and their components considered in the present exercise are reviewed in Table 6.1 below. The levels of confidence in the outputs of the classification tool are also indicated.

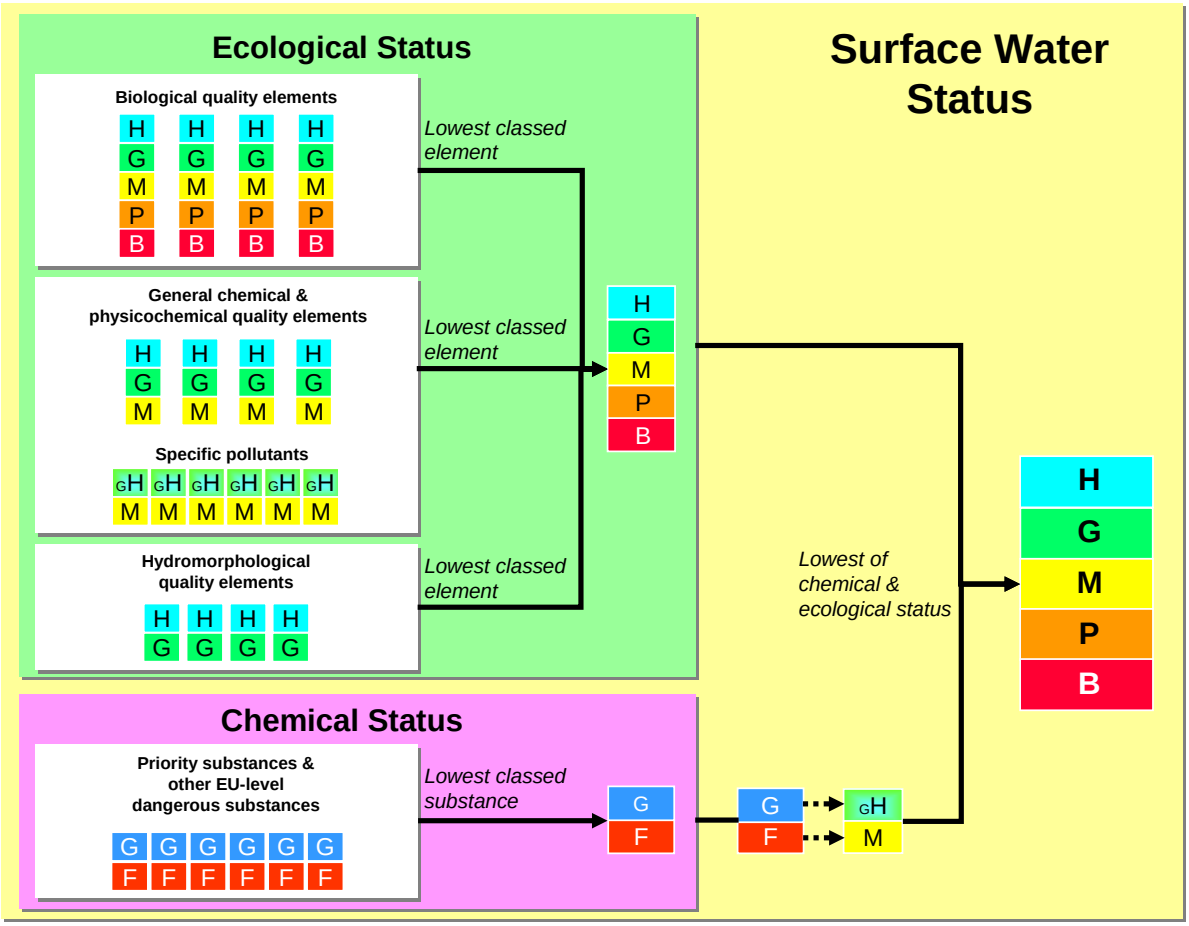
Table 6.1. Status of quality elements considered in classification of ecological potential of canal water bodies.

Quality element	Sub-element	Status	Confidence in outputs
A. Biology	1. Macrophytes and phytobenthos	Prototype	High
	2. Macroinvertebrates	Prototype	High
	3. Phytoplankton	Standards	Medium
B. Physico-chemistry	5. BOD	Standards	Medium
	6. Ammonia	Standards	Medium
	7. Phosphorus	Standards	High
	8. Nitrogen	Standards	High
	9. Suspended Solids	Standards	High
	10. Chloride	Standards	High
C.Hydro-morphology	N/A	Prototype	Medium
D. Invasive non-native species	N/A		High

The WFD invokes the 'one-out-all-out' rule and is thus explicit on how information from different quality elements should be combined to achieve an ecological classification of a water body. Essentially, the components of the various quality elements can be viewed as the lowest tier (although the metrics which are used to classify each component could be viewed as an even lower tier). The component with the lowest class dictates the status of the quality element. The quality element with the lowest status then dictates the ecological potential of the water body. The overall surface water status is defined by the lower of ecological or chemical status in which chemical status is defined relative to the standards established for a series of hazardous substances. These include a number of substances that may be of particular relevance to canals due to historic industrial pollution (e.g. lead, mercury and cadmium). This is illustrated in Figure 6.1.

The 'one out all out approach' has received significant criticism in recent years on the basis that it is too precautionary and should be replaced by some form of assessment based on combined weight of evidence across multiple quality elements. It is therefore possible that the prescribed approach will be reviewed and revised.

Figure 6.1. The basis for classification of surface water status of water bodies through the one out all out approach. Taken from the draft UK Classification Scheme for Surface Waters (UKTAG, 2007)



6.2 Invasive species

Chapter 7 of WFD61a (Willby et al., 2008) provided a detailed assessment of the role of invasive non-native species (INNS) in the ecological classification of canals. In general, the thinking in relation to invasive species has not changed significantly in the interim and the salient points from WFD61a are therefore reproduced below.

UKTAG have recognised the need to incorporate invasive species into the ecological classification of water bodies. The metrics proposed in the present report for macroinvertebrates, macrophytes and physico-chemistry do not take specific account of INNS, although they may indirectly reflect their effects, if for example, species richness of macrophytes is reduced through competitive exclusion. In this sense metrics designed for other quality elements might be expected to register the impacts of INNS when these impacts are sufficiently severe. However, some INNS carry a sufficiently high risk of impact based on experience at other sites that their *potential* for impact should form the basis for classification, rather than there being a need for these impacts to be demonstrated first before they translate to changes in ecological classification. This was the philosophy adopted by UKTAG (2005) in their guidance on risk assessment.

The Alien Species Group (ASG) have provided recommendations to UKTAG on how INNS should be considered as part of the process of risk assessment and classification of water bodies for the WFD (UKTAG, 2004). Canals or other artificial water bodies were not explicitly considered in this guidance. WFD61a therefore listed several considerations in relation to the inclusion of INNS in the classification of canals.

1. As a result of their connectivity redistribution of INNS through the canal network has often been very rapid. Consequently species that may still be in the process of invading natural water bodies may already be largely naturalised in canals. There is an extensive literature reporting the spread of invasive water plants, molluscs and amphipods via the canal system (e.g., Boycott, 1936; Hynes, 1955; Simpson, 1984).

2. Hydromorphological disturbance is well known to promote recruitment of INNS, especially if coupled with fluctuating resource supply (Willby, 2007). Navigable waterways are inherently disturbed environments. Certain aspects of their management, such as dredging, may also inadvertently promote the growth or spread of invasive species.

3. Whether by accident or design boats can assist the dispersal of invasive species within canals and have also introduced, via ballast water, species to the downstream end of rivers that have subsequently spread into canals.

4. Some native water plant species with ornamental value have been widely translocated outside their native range and frequently exhibit invasive characteristics when present in low traffic canals.

5. There are important national differences that need to be considered when dealing with INNS, especially in the context of a tool that is intended to cover two different ecoregions. Thus, there appear to be several species that are either in the early stages of invading freshwaters in Ireland (e.g. *Elodea nuttallii*) or which are more strongly invasive there than in Great Britain (e.g. *Lagarosiphon major*).

6. Some invasive species have beneficial attributes in canals. Thus some species such as *Elodea nuttallii* will persist at high cover in polluted urban situations where few if any native species would survive. In this case an invasive species probably preserves some of the core components of macrophyte-dependent ecosystem function that would otherwise be lost. In heavily trafficked canals invasive amphipod species, such as *Corophium curvispinum*, are a major food source for large roach and their presence in high densities effectively ‘shores up’ an ecosystem from which large-bodied, plant- associated invertebrates have been lost (Willby, 1994).

Point 1 illustrates that canals may constitute a particular risk to other water bodies through facilitating the spread of invasive species. Points 1-3 indicate that to some extent INNS could be considered an intrinsic attribute of canals that reflects their connectivity and navigation function.

To reflect as far as possible the ASG recommendations the following strategy was proposed for integrating INNS into the ecological classification of canals.

1. When the following *high risk* species are present, regardless of population size or apparent impact (e.g. on metric values of other BQEs), a site cannot be considered to achieve MEP. This approach recognizes the potential risks

associated with invasive species both to the ecology of the canal water body itself and to sites elsewhere in the catchment.

Azolla filiculoides	Water Fern
Crassula helmsii	New Zealand Pygmyweed, Australian Stonecrop
Hydrocotyle ranunculoides	Floating Pennywort
Myriophyllum aquaticum	Parrot's Feather
Pacifastacus leniusculus	Signal Crayfish
Procambarus clarkii	Red Swamp Crayfish
Eriocheir sinensis	Chinese Mitten Crab
Dreissena polymorpha	Zebra Mussel
Dikerogammarus villosus	Killer Shrimp
Stizostedion lucioperca	Zander

2. When the following species are present in a canal in areas/countries (e.g. Ireland) from which they are otherwise largely absent, a site cannot achieve MEP

Elodea nuttallii	Nuttall's Pondweed
Lagarosiphon major	Curly Water Thyme

3. Unless mentioned under 2 above, when the following species are present they are regarded as either naturalized or 'low impact' for the purposes of integrating INNS into ecological classifications. If the species are proving invasive it is assumed that this will be reflected through the effects of dominance on different biological metric values or through distortion of values for physico-chemical variables. Essentially the *presence* alone of these species cannot be regarded as grounds for failing to achieve MEP. It is important to appreciate that several of the species listed below have become exceptionally widespread on canals. Since containment or erradication are no longer an option, it is difficult to formulate a management response to the presence of these species, and as such, it is difficult to therefore see how their presence *a priori* should reduce ecological quality.

Acorus calamus
Aponogeton distachyos
Egeria densa
Elodea canadensis
Elodea nuttallii
Lagarosiphon major
Lemna minuta
Lysichiton americanum
Mimulus guttatus
Vallisneria spiralis

Crangonyx pseudogracilis
Corophium curvispinum

Cyprinus carpio
Carassius carassius

4. The following species are native at a very small number of canal sites but are almost exclusively translocated outside their native range and may then prove invasive. They should be considered as in the case of 3. This removes the need to determine whether such species may be native or introduced to a site which in certain cases would be problematic.

   Hydrocharis morsus-ranae
   Nymphoides peltata
   Stratiotes aloides

5. If the presence of a high risk species is suspected, e.g. due to low EQR values for various metrics that cannot be readily explained in other ways, the necessary operational monitoring should be carried out.

6. The lists of species in 1, 2 and 3 above should be reviewed in advance of each round of river basin planning to take account of changes in the distribution or invasiveness of species, or colonization by new INNS that may pose a risk. In this context, individual countries should be free to add to their own lists of high risk species.

7. Where a site fails to achieve MEP due to the presence of high risk INNS the decision on whether the site then achieves GEP should depend on evidence of impact, as reflected in metric EQR values relative to the G/M boundary. Thus, one should consider if INNS maybe responsible for lowering the metric values of BQEs, or if, for example, their impact is mitigated by management or boat traffic effects. In this instance INNS may contribute to a site failing to achieve GEP but their presence cannot be used separately to over ride the biological classification in the same way it is applied when downgrading a site from MEP to GEP.

8. The view presented here and agreed by the steering group is that canals should be classified as worse than MEP on the basis of the *presence* of high impact INNS. UKTAG has recommended a requirement that such species should be considered as 'established' before any down grading of ecological status can be justified. A species is considered 'established' when it has been present and reproducing for a number of years and is not confined to a small proportion of the water body. The confidence in the classification will then depend on the confidence that the species is established. In our view the requirement for establishment is inconsistent with the risk of impact. If a species has a high risk of impact it should not be allowed to establish before it can influence the classification of a water body. The confidence in the classification is then contingent on the assessment of risk associated with the species (which is a product of analysis of external evidence) not on the confidence that the sampling methods used will detect and assess its population size adequately. We would also emphasise that the species listed in section 1 above are either absent from or, with the possible exceptions of Zebra Mussel and Signal Crayfish, have a very restricted distribution on canals in Britain and Ireland. Since it is fairly unlikely that sites in which these species occur would be classified as being at MEP due to their impacts on other BQEs (or hydromorphology) the downgrading of canal sites from MEP purely on the basis of presence of high risk INNS is liable to affect only a small number of canal sites.

9. The approach discussed in 8, based on presence rather than establishment, is somewhat more demanding than the current UKTAG recommendations. The general position with AWBs and HMWBs is that a *less* demanding

approach is adopted to take account of their altered hydromorphology. However, canals are interconnected to a greater or lesser degree to natural water bodies and the influence of this unusually high level of connectivity needs to be recognised if surface water bodies in the wider landscape are to be protected from the effects of high risk invasive species. To compensate for this more precautionary approach the high risk species listed in sections 1 and 2 above is substantially shorter than the corresponding lists currently provided by UKTAG.

10. The overall principle in relation to integration of INNS into ecological classification is analogous to the treatment of hydromorphology. Thus, water bodies cannot be considered to achieve MEP if they support high risk invasive species, or have experienced slight modifications to their hydromorphology. Beyond this, it is assumed that reductions in Ecological Potential below GEP require direct or indirect impacts of INNS or hydromorphology on metrics used to assess biology or physico-chemistry.

6.3 Data required for full classification of ecological potential

Classification of ecological potential require compliant data collected at replicate locations within a water body at a designated frequency. The requisite information is summarised in Table 6.2. Sites for macroinvertebrates, macrophytes and hydromorphology should ideally be coincident and it will be advantageous if samples for measurement of chlorophyll and physico-chemistry are also obtained from one of these locations.

Table 6.2. Number, distribution and frequency of sampling for different quality elements in a typical sized canal water body (e.g. 20km in length).

Element/sub-element	Sites in water body	Distribution	Frequency	Seasonality
Macroinvertebrates	3-5	equidistant over length	3 years	spring
Macrophytes	3-5	equidistant over length	3-6 years	summer
Phytoplankton	1	downstream end	3 years	(bi)monthly
Physico-chemistry	1	downstream end	3 years	(bi)monthly
Hydromorphology	3-5	equidistant over length	6 years	Spring-summer

6.4 Worked example of classification of ecological potential

The expectation for each quality element and its various components is that the absolute value or EQR to be used in classification is the mean value from samples or surveys taken either (i) from several sites in a water body at a given point in time or (ii) repeated samples from a single point in water body or (iii) a combination of both (i) and (ii).

Consider the worked example site on the Forth & Clyde Canal, already referred to in sections 2.9, 3.8 and 4.4. The overall EQR values for macrophytes (0.69) and macroinvertebrates (0.76) provide the EQRs in Table 6.3. In chapter 4 the worked example indicated that hydro-morphology at this site would be considered at GEP

but not MEP. A set of mean values are also provided for the various physico-chemical quality elements that were monitored. Currently these are simply classified into classes relative to the standards proposed in Chapter 5. It is questionable whether classification to a higher resolution (i.e. EQR) can be justified given the data available on which to base standards.

In this example, some elements (phytoplankton, BOD, nitrogen and chloride) have values consistent with Maximum Ecological Potential. However, the lowest class of Ecological Potential for which data is available is Good. Consequently, based on the one-out all-out approach the Ecological Potential for the water body would also be Good. In this example the class for hydromorphology, which is a supporting element, is also good which lends support to the approach used to classify hydromorphology. A class based on hydro-morphology of less than Good might suggest that the approach to hydromorphological assessment was too stringent. The list of macroinvertebrates and macrophytes recorded at this site contains no high risk INNS so there would be no case for downgrading the site below MEP specifically in relation to INNS.

In the example presented here the data is taken from a single site in the waterbody and for the purposes of illsutration the water body as a whole is classified using the information from a single site. In this example the weight of evidence based on all the information presented is that the site should be classified as being at Good Ecological Potential. Hence, the conclusion based on the one-out, all-out approach is found to be reasonable. However, if the data presented here was unchanged but additional information became available for one of the specific pollutants which classified that pollutant at Moderate Ecological Potential this new lower value would dictate the overall Ecological Potential for the site under the one-out all out approach, even though the weight of evidence would clearly still support a classification of Good Ecological Potential.

Table 6.3. Worked example showing mechanism for combining information from different quality elements to achieve an overall classification of ecological potential for a canal water body

Quality element	Sub-element	normalised EQR	Value	Class	Class for QE	CoC	Ecological Potential for WB
A. Biology	1. Macrophytes	0.69		Good[1]			
	2. Macroinvertebrates	0.76		Good[1]	Good	?	
	3. Phytoplankton		9 ug/L	Max[2]			
B. Physico-chemistry	4.BOD		2.3 mg/L	Max[2]			
	5.Ammonia		0.12 mg/L	Good[2]		?	
	6.Phosphorus		70 ug/L	Good[2]	Good		Good
	7.Nitrogen		0.86 mg/L	Max[2]			
	8.Suspended solids		8 mg/L	Good[2]			
C.Specific pollutants	Chlorine		33.8 mg/L	Max[2]			
	Copper		n/d				
	Iron		n/d				
	Zinc		n/d				
Supporting elements							
C.Hydro-morphology					Good		
D. Alien species					Max		

1. Relative to EQR boundaries for tool
2. Relative to proposed standards

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