Use of Physical Habitat Structure to Assess Stream Suitability for Brown Trout: A Case Study of Three Upland Scottish Streams

By

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Thesis submitted for the degree of Doctor of Philosophy
University of Stirling

October, 2006

In memory of my father

Michael Joseph Neary

1939-2001

Declaration

I herby declare that this thesis has been composed entirely by myself and has not been previously submitted for any other degree or qualification. The work, of which this document is record, has been carried out by myself. The nature and extent of any work carried out by, or in conjunction with others, has been specifically acknowledged by reference.

James Neary

Acknowledgements

I would like to thank my supervisors Dr. Donald Baird, Dr. David Gilvear, and Dr. Trevor Telfer for their instruction, guidance and support and for providing me with the opportunity to take on this challenge. I would also like to recognize and thank the University of Stirling and the Department of Aquaculture for their financial support by providing a half Student Scholarship and Dr. Baird who found the rest of the money.

This work could not have been completed if not for the technical skills and assistance of a number of individuals. These include Emine, Jon, and Sylvia who helped with field work, Betty, Charlie, Jane and Melanie for help with the day-to-day administrative tasks, Billy for help in the lab, Marilyn and Oron for keeping me connected, and Linda Cameron and Lisa Haddrow who went above and beyond to help supply literature. Thank you all very much. As well, I would like to thank to Scottish Water and Andy Gallacher of the Forestry Commission for providing site access. A special thanks to Dr. Colin Bull for assistance with electro-fishing and transportation issues and an extra special thanks to Dr. Cori Critchlow-Watton who helped with literature, as a go-between, and for listening to me rant and rave. Finally I would like to thank my examiners, Dr Nigel Wilby and Dr Chris Gibbins, whose comments greatly improved this study.

Most importantly I'd like to thank my wife Emine. Nothing is possible without your love.

Abstract

In 2000 the European Union introduced the Water Framework Directive, new legislation that regulates the use of surface waters within the European Community. The goal of this legislation is to protect, enhance and restore all surface waters within the Community to Good Surface Water Status. Good-Status is described as having low levels of anthropogenic distortion in its hydro-morphological and physiochemical components as well as possessing biota that would normally be associated with the type-specific aquatic ecosystem. The assessment of ecosystem status is to be defined by comparisons with intact representative reference sites, by using modelling techniques that define reference conditions, a combination of the two, or expert judgement. As undisturbed aquatic ecosystems are rare or non-existent in Europe the base-line data will have to be defined using the latter methodologies.

The aim of this project is to help define reference conditions for lotic systems in Europe based on the physical instream habitat parameters of a resident species. Brown trout (*Salmo trutta*), a ubiquitous and well studies species endemic to Europe, was used as the target organism to develop the assessment protocol. The project focused on the requirements this species has of aspects of its physical habitat; specifically, its usage of depth, velocity, and substrate. An extensive survey of the scientific literature was used to define the requirements trout has for the three physical parameters at four life stages. These are the spawning, nursery, juvenile and adult-resident life stages. These requirements were expressed as tolerance profiles, which defined suitable, usable and not-suitable habitat. The methodology was demonstrated by evaluating the physical habitat available at six reaches in three small streams, March, Burnhouse and Bin Burns, which drain into the Carron Valley Reservoir in central Scotland.

From the perspective of water depth, these streams seem best suited as nursery areas, are less well suited as juvenile habitat, and do not appear to be well matched for adult residents. The assessment of both velocity and substrate indicated that the portion of the study reaches available for use by resident brown trout increased with trout size. The assessment of all three physical habitat parameters at all study reaches found variable portions of the streams suitable for use by spawning trout. When the habitat variables are integrated all stream segments streams seem best suited as nursery and spawning areas. To a lesser extent juvenile trout can use these burns and very little habitat is available for use by adult resident trout.

The tolerance profiles that were created in this study are standardized assessment criteria that when compared with stream survey data can produce an appraisal of habitat availability in any fluvial freshwater system that supports populations of brown trout (*Salmo trutta*). The assessment method can be combined to produce an integrated habitat assessment, using both an index and by the calculation of Froude number, which is a more realistic approach than the assessment of individual habitat parameters as salmonids choose their microhabitat based on multiple factors. This approach allows an investigator to determine the amount and relative portion of useable habitat and to determine the quality of that habitat. Finally, by examining the physical habitat variable that most strongly correlates with the final integrated habitat distribution the individual habitat parameter that is most important to the distribution of physical habitat at a site can be determined. While this technique would certainly benefit from further development it does show potential to aid in physical habitat assessment of trout streams.

Abbreviations

ASPT	Average Score Per Taxon
AusRivAS	Australian River Assessment Scheme
BACI	Before After Control Impact
BEAST	BEnthic Assessment of SedimenT
BI	Benthic Invertebrates
B-IBI	Benthic Index of Biotic Integrity
BMWP	Biological Monitoring Working Party
C	Celcius
CDV	Critical Displacement Velocity
Cm	centimetre
EPA	Environmental Protection Agency
EU	European Union
FORTRAN	FORmula TRANslation/Translator
fs	flow scores
HSC	Habitat Suitability Curves
HSI	Habitat Suitability Index
IBI	Indices of Biotic Integrity
IFIM	Instream Flow Incremental Methodology
LIFE	Lotic-invertebrate Index for Flow Evaluation
km	kilometre
m	meter
MDA	Multiple Discriminate Analysis
ML/day	Megalitres per day
PHABSIM	Physical HAbitat SIMulation

RIVPACS	RIver InVertebrate Prediction And Classification System
Sec	second
SEPA	Scottish Environmental Protection Analysis
TWINSPAN	Two-Way INdicator SPecies ANalysis
WFD	Water Framework Directive

Table of Contents

Abstract i

Abbreviations iii

Table of Contents v

List of Tables ix

List of Figures xiii

1.0 Introduction 1
  1.1 Literature Cited 6

2.0 Literature Review 10
  2.1 Overview 10
  2.2 The Water Framework Directive 11
  2.3 Ecosystem Health and Integrity 16
  2.4 Habitat Requirements and the Niche 28
  2.5 Reference Conditions 30
    2.5.1 Physical site comparisons 34
  2.6 Single Point Impact Assessment 35
    2.6.1 Green's upstream-downstream model 36
    2.6.2 Before-after-control-impact (BACI) 38
  2.7 River Invertebrate Prediction and Classification System (RIVPACS) 43
  2.8 Bioenergetic Models 47
  2.9 Biotic Indices 49
  2.10 Hydraulic Models 54
    2.10.1 One-dimensional hydraulic models (PHABSIM): 54
    2.10.2 Two- and three-dimensional hydraulic models: 63
  2.11 The lotic-invertebrate index for flow evaluation (LIFE) 65
  2.12 Summary 68
  2.13 Literature Cited 71

3.0 Methods 82
  3.1 Conceptual Overview of the Approach 82
  3.2 The Study Site: The Carron Valley 84
  3.3 Brown Trout: Overview 88
    3.3.1 Target species: brown trout (*Salmo trutta*) 88
    3.3.2 Phylogeny 89
    3.3.3 Distribution 89
    3.3.4 Life history 91
  3.4 Definition of Habitat Parameters and Field Methods 93
    3.4.1 Literature summary and tolerance profiles 94
      3.4.1.1 Water velocity 99
      3.4.1.2 Substrate 99
      3.4.1.3 Tolerance profile construction 103
      3.4.2 Stream survey 104

3.4.3 Stream habitat assessment 106
3.5 Fish Survey 107
3.6 Statistical Analysis 107
3.7 Literature Cited 109

4.0 Depth 113
4.1 Introduction 113
4.1.1 Instream depth requirements: overview 113
4.2 Results 115
4.2.1 Tolerance profiles 115
4.2.1.1 Spawning depth 115
4.2.1.2 Nursery depth (fish length ≤ 7 cm) 120
4.2.1.3 Juvenile depth (fish length: > 7 to 20 cm) 125
4.2.1.4 Adult non-spawners (fish length > 20 cm) 134
4.2.1.5 Summary of tolerance profiles 136
4.2.2 Habitat suitability 141
4.2.2.1 Spawning depth 141
4.2.2.2 Nursery depth (fish length ≤ 7 cm) 144
4.2.2.3 Juvenile depth (fish length > 7 to 20 cm) 150
4.2.2.4 Adult non-spawners (fish length > 20 cm) 152
4.2.3 Habitat maps 155
4.2.4 Fish survey 160
4.3 Discussion 162
4.4 Literature Cited 169

5.0 Velocity 176
5.1 Introduction 176
5.1.1 Instream water velocity requirements: overview 176
5.2 Results 179
5.2.1 Tolerance profiles 179
5.2.1.1 Spawning water velocity 179
5.2.1.2 Nursery water velocity (fish length ≤ 7 cm) 186
5.2.1.3 Juvenile water velocity (fish length: >7 to 20 cm) 191
5.3.1.4 Velocity use by non-spawning adults (> 20 cm) 199
5.2.1.5 Summary of tolerance profiles 206
5.3.2 Habitat preference 208
5.3.2.1 Spawning velocity 208
5.3.2.2 Nursery water velocity (fish length ≤ 7 cm) 214
5.3.2.3 Juvenile water velocity (fish length: >7 to 20 cm) 217
5.3.2.4 Adult non-spawners water velocity (> 20 cm) 219
5.3.3 Habitat maps 222
5.4 Discussion 227
5.5 Literature Cited 232

6.0 Substrate 238
6.1 Introduction 238
6.1.1 Streambed Substrate Requirements: Overview 239
6.2 Results 243
6.2.1 Tolerance profiles 243
6.2.1.1 Spawning substrate 243

6.2.1.2 Nursery substrate (fish length ≤ 7 cm) 249
6.2.1.3 Juvenile substrate (fish length: >7 to 20 cm) 252
6.2.1.4 Substrate used by non-spawning adults (> 20 cm) 257
6.2.1.5 Summary of tolerance profiles 260
6.2.2 Habitat assessment 263
6.2.2.1 Spawning substrate 263
6.2.2.2 Nursery (fish length ≤ 7 cm) and juvenile substrate (fish length: >7 to 20cm) 267
6.2.2.3 Adult non-spawners substrate (> 20 cm) 271
6.2.3 Habitat maps 271
6.3 Discussion 277
6.4 Literature Cited 282

7.0 Integration 288
7.1 Introduction 288
7.2 Methods 289
7.2.1 Physical habitat index 289
7.2.2 Froude Number 291
7.3 Results 292
7.3.1 Integrated quality index 292
7.3.1.1 Integrated spawning habitat assessment 292
7.3.1.2 Integrated nursery habitat assessment (fish length ≤ 7 cm) 299
7.3.1.3 Integrated juvenile habitat assessment (fish length > 7 to 20 cm) 302
7.3.1.4 Integrated adult habitat assessment (fish length > 20 cm) 306
7.3.2 Habitat assessment maps 309
7.3.3 Froude number 314
7.3.4 Secondary analysis: correlation (Cramer's phi) 316
7.4 Final Site Characterization 325
7.4.1 March Burn Upstream 325
7.4.2 March Burn Downstream 326
7.4.3 Burnhouse Burn Upstream 327
7.4.4 Burnhouse Burn Downstream 327
7.4.5 Bin Burn Upstream 328
7.4.6 Bin Burn Downstream (Pre-spate) 329
7.4.7 Bin Burn Downstream (Post-spate) 330
7.5 Discussion 331
7.6 Literature Cited 337

8.0 General Discussion 339
8.1 General Overview 339
8.2 Strengths of the Assessment Protocol 340
8.3 Limitations of the Model 341
8.4 Future Studies 344
8.5 Literature Cited 345

A.0 Stream Discharge 346
A.1 Objectives 346
A.2 Methods 346
A.3 Results 347

A.4 Comments 352

List of Tables

Table 2.1: Definitions of ecosystem health outlined by Xu and Tao (2000) 20

Table 2.2: Potential sources of confusion in experiments and means for minimizing their effect (Redrawn from Table 1 pp.191 *in* Hurlbert 1984)................ 32

Table 2.3: Parameters used in assessment of fish communities (Redrawn from Table 2 pp. 22 in Karr 1981)... 53

Table 2.4: Outline of methods used to calculate joint preference functions used to describe a species preference or tolerance of the combination of physical habitat parameters (depth, velocity, substrate, cover) generated by PHABSIM for each stream cell. Summarized from (Bovee 1982)........ 59

Table 3.1: Location and sample date of study sites in the Carron Valley................ 88

Table 4.1: Literature used to define the range of stream depths that can be used by brown trout (Salmo trutta) to spawn. 116

Table 4.2: Literature used to define the range of stream depths that can be used by brown trout (*Salmo trutta*) fry (≤ 7 cm)... 127

Table 4.3: Literature used to define the range of stream depths that can be used by juvenile brown trout (*Salmo trutta*) (>7 – 20 cm)................................ 132

Table 4.4: Literature used to define the range of stream depths that can be used by adult brown trout (*Salmo trutta*) (>7 – 20 cm)...................................... 138

Table 4.5: Results for chi-square test (α = 0.05) comparing the proportions of quadrats that contained depths that were considered suitable, useable, not-usable and dry for brown trout in both the total and wetted streambed at all study reaches. 142

Table 4.6 Results for chi-square test (α = 0.05) comparing the proportions of quadrats that contained depths that were considered suitable, useable, and not-unusable and dry for spawning brown trout for both the total and wetted streambed at Bin Burn (downstream) before and after the spate. 142

Table 4.7: Summary of the proportion of water depth available for the four life stages of brown trout for all study reaches including both total and the wetted portions of the streambed. 143

Table 4.8: Discharge and surface area (total, wetted and dry) at all study sites examined in the Carron Valley. 146

Table 4.9: The total area that has accessible water depths for each study reach for age class. Total accessible area calculated by multiplying total wetted

area by the sum of the proportion of the streambed that was either suitable or useable. 147

Table 4.10 (A-C):Brown trout observed at the downstream sites of March Burn (A), Burnhouse Burn (B) and Bin Burn (C) between September 2002 and June 2003. 161

Table 5.1: Literature used to define the range of velocities that can be used by brown trout (*Salmo trutta*) to spawn. 181

Table 5.2: Literature used to define the range of stream water velocities that can be used by brown trout (*Salmo trutta*) fry (≤ 7 cm). 188

Table 5.3: Literature used to define the range of stream velocities that can be used by juvenile brown trout (*Salmo trutta*) (>7 – 20cm). 195

Table 5.4: Literature used to define the range of stream velocities that can be used by adult brown trout (*Salmo trutta*) (>20cm). 203

Table 5.5: Results for chi-square test comparing the proportions of quadrats that contained water velocities that were considered suitable, useable, not-usable and dry for spawning brown trout for both the total and wetted streambed at all study reaches. 209

Table 5.6 Results for chi-square test comparing the proportions of quadrats that contained water velocities that were considered suitable, useable, not-usable and dry for spawning brown trout for both the total and wetted streambed at Bin Burn (downstream) before and after the spate. 209

Table 5.7: Summary of the proportion of velocities available for the four life stages of brown trout for all study reaches including both total and the wetted portions of the streambed. 210

Table 5.8: The total area that has accessible water velocities for each study reach for age class. Total accessible area calculated by multiplying total wetted area by the sum of the proportion of the streambed that was either suitable or useable. 213

Table 6.1: Literature used to define the range of substrate sizes that can be used by brown trout (*Salmo trutta*) to spawn. 245

Table 6.2: Literature used to define the range of substrate sizes that can be used by brown trout (*Salmo trutta*) fry (≤ 7 cm). 250

Table 6.3: Literature used to define the range of substrate sizes that can be used by juvenile brown trout (*Salmo trutta*) (>7 – 20cm). 254

Table 6.4: Literature used to define the range of substrate sizes that can be used by adult brown trout (*Salmo trutta*) (>20cm). 258

Table 6.5: Results for chi-square test comparing the proportions of quadrats that contained substrate that was considered suitable, useable, and not-useable, (suitable and unsuitable only for spawning) by brown trout at all study reaches and life stages. 264

Table 6.6: Results for chi-square test comparing the proportions of quadrats that contained substrate that was considered suitable, useable, and not-useable (suitable and unsuitable only for spawning) by brown trout at Bin Burn before and after the spate at all life stages. 264

Table 6.7: Summary of the proportion of stream substrate available for the four life stages of brown trout for all study reaches for the total streambed (including wetted and dry portions). 265

Table 6.8: The total area that has acceptable stream-substrate at each study reach for each age class. Total accessible area calculated by multiplying the total streambed area by the sum of the proportion of the streambed that was either suitable or useable. 267

Table 7.1: Possible combination of habitat suitability codes and quality evaluation for sampling quadrats containing habitat useable by brown trout (*Salmo trutta*)... 291

Table 7.2: Results for the chi-square test that compared the proportions of quadrats classified by integrated quality values (very high, high, good, low, satisfactory, and not-useable) between the seven datasets for both the total and wetted streambed. 293

Table 7.3 Results for the chi-square tests that compared the proportions of quadrats classified by integrated quality values (very high, high, good, low, satisfactory, and not-useable) between the Bin Burn datasets (before and after the spate) for both the total and wetted streambed. 293

Table 7.4: Summary of brown trout (*Salmo trutta*) physical habitat assessment based on integrated criteria for both the total and wetted streambeds for each of the four life stages. 294

Table 7.5: The total area that is useable by brown trout for each study reach and each age class. Total accessible area calculated by multiplying total area by proportion of the streambed that was rate as useable (very-high, high, good, or low). 298

Table 7.6: Proportion of pool, run, and riffle habitat in the wetted portions of each study reach based on Froude Number. 314

Table 7.7: Proportion of wetted quadrats with super-critical flow (Froude number > 1). 317

Table 7.8: Cramers's phi coefficients resulting from the comparisons of suitability scores (suitable, useable, unsuitable) calculated for the individual

assessments with the integrated quality score (very-high, high, good, low, not useable) in the total streambed. 319

Table 7.9: Cramers's phi coefficients resulting from the comparisons of suitability scores (suitable, useable, unsuitable) calculated for the individual assessments with the integrated quality score (very-high, high, good, low, not useable) in the wetted streambed. 322

Table A.1: Average discharge and standard deviation measured in March Burn in late 2002 and early 2003. 347

Table A.2: Average discharge and standard deviation measured in Burnhouse Burn in late 2002 and early 2003. 348

Table A.3: Average discharge and standard deviation measured in Bin Burn in late 2002 and early 2003. 348

List of Figures

Figure 2.1: The abundance of "Species X" at the Impact and Control stations, and the difference of the abundances, as functions of time, in three versions of Impact assessment (Redrawn from Figure 1 pp. 931 *in* Stewart-Oaten et al. 1986)........ 40

Figure 3.1: **Location of the sample sites from which water depth, water velocity, and streambed substrate were taken during the late summer of 2002. Ordnance Survey.........86**

Figure 3.2**:** **The native distribution of brown trout (Salmo trutta). Redrawn after Elliott, 1994.........90**

Figure 3.3: **Symbols used to identify the three data types employed in the construction of brown trout microhabitat tolerance profile.........97**

Figure 3.4: Symbols used to identify the substrate data types employed in the construction of brown trout microhabitat tolerance profiles........ 100

Figure 3.5**:** **Modified Wentworth scale after Bain et al. 1985 used for stream substrate particle size classification. Redrawn from Table II in Heggenes and Saltveit 1990.........103**

Figure 4.1: Procedure used to define the range of stream depths that can be used by brown trout (*Salmo trutta*) to spawn........ 121

Figure 4.2: Procedure used to define the range of stream depths that can be used by brown trout (*Salmo trutta*) fry (≤ 7 cm. - Nursery Habitat)........ 126

Figure 4.3: Procedure used to define the range of stream depths that can be used by juvenile brown trout (*Salmo trutta*) (> 7 cm – 20 cm)........ 130

Figure 4.4: Procedure used to define the range of stream depths that can be used by adult brown trout (*Salmo trutta*) (> 20 cm)........ 137

Figure 4.5: Comparison of tolerance profiles constructed for water depths for four size classes of brown trout (*Salmo trutta*), which including spawning, nursery (≤ 7 cm), juvenile (>7 – 20 cm), and adult residents (> 20 cm)..... 140

Figure 4.6 (A-G): Histograms of the proportion of spawning depth at all study sites classified as suitable, useable, not-useable, and dry for both the total and wetted streambed........ 145

Figure 4.7 (A-G): Histograms of the proportion of water depth available to young trout (≤ 7 cm) at all study sites classified as suitable, useable, not-useable, and dry for both the total and wetted streambed. 148

Figure 4.8 (A-G): Histograms of the proportion of water depth available to juvenile trout (> 7 cm to 20 cm) at all study sites classified as suitable, useable, not-useable and dry for both the total and wetted streambed................ 151

Figure 4.9 (A-G): Histograms of the proportion of water depth available to adult trout (> 20 cm) at all study sites classified as suitable, useable, not-suitable and dry for both the total and wetted streambed. 154

Figure 4.10: Water depth available for spawning brown trout (*Salmo trutta*) at the downstream sites of March (Q = 0.0341 m^3/sec) and Burnhouse Burns (Q = 0.0058 m^3/sec) based on assessed depth requirements. 156

Figure 4.11: Water depth available for brown trout (*Salmo trutta*) fry (≤ 7 cm) at the downstream sites in March (Q = 0.0341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) base don assessed depth requirements. 157

Figure 4.12**: Water depth available for juvenile brown trout (*Salmo trutta*) at the downstream sites in March (Q = 0.0341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on assessed depth requirements. 158**

Figure 4.13: Water depths available for non-spawning adult resident brown trout (*Salmo trutta)* at the downstream sites in March (Q = 0.0341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on the depth requirements. 159

Figure 5.1: Procedure used to define the range of stream water velocities that can be used by brown trout (*Salmo trutta*) to spawn. 181

Figure 5.2: Procedure used to define the range of stream water velocities that can be used by brown trout (*Salmo trutta*) fry (length ≤ 7 cm). 187

Figure 5.3: Procedure used to define the range of velocities measured in the water column that can be used by juvenile brown trout (*Salmo trutta*) (> 7 cm – 20 cm).. 193

Figure 5.4: Procedure used to define the range of velocities measured near the substrate that can be used by juvenile brown trout (*Salmo trutta*) (> 7 cm – 20 cm)... 194

Figure 5.5: Procedure used to define the range of near bed mean column velocities that can be used by non-spawning adult brown trout (*Salmo trutta*) (>20 cm)... 201

Figure 5.6: Procedure used to define the range of near bed mean column velocities that can be used by non-spawning adult brown trout (*Salmo trutta*) (>20 cm).. 202

Figure 5.7: Summary of tolerance profiles by age class [including spawning; nursery (≤7 cm); juvenile (>7 –20 cm); and adult non-spawners (> 20 cm)] for measures of A) mean column velocity and B) near bed water velocity used by brown trout (*Salmo trutta*). 207

Figure 5.8 (A-G): Histograms of the proportion of velocity available for spawning trout at all study sites classified as suitable, useable, not-suitable and dry for both the total and wetted streambed. 212

Figure 5.9 (A-G): Histograms of the proportion of measures of velocity available to young trout (≤ 7 cm) at all study sites classified as suitable, useable, not-suitable and dry for both the total and wetted streambed..................... 215

Figure 5.10 (A-G): Histograms of the proportion of measures of velocity available to juvenile trout (> 7 cm to 20 cm) at all study sites classified as suitable, useable, not-usable and dry for both the total and wetted streambed. .. 218

Figure 5.11 (A-G): Histograms of the proportion of measures of velocity available to adult trout (> 20 cm) at all study sites classified as suitable, useable, not-unusable and dry for both the total and wetted streambed. 220

Figure 5.12: Water velocity available for spawning brown trout (*Salmo trutta*) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on velocity requirements.. 223

Figure 5.13: Water velocity available for brown trout fry (*Salmo trutta*; length ≤ 7 cm) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on velocity requirements. 224

Figure 5.14: Water velocity available for juvenile brown trout (*Salmo trutta*; length >7 to 20 cm) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on velocity requirements. 225

Figure 5.15: Water velocity available for adult non-spawning brown trout (*Salmo trutta*; length >20 cm) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on velocity requirements. 226

Figure 6.1: Modified Wentworth scale after Bain et al. 1985 used for stream substrate particle size classification. Redrawn from Table II in Heggenes and Saltveit 1990. 240

Figure 6.2: Procedure used to define the range of substrate particle sizes that can be used by brown trout (*Salmo trutta*) to spawn. 247

Figure 6.3: Procedure used to define the range of substrate particle sizes that can be used by brown trout (*Salmo trutta*) fry (≤ 7 cm)................................. 251

Figure 6.4: Procedure used to define the range of substrate particle sizes that can be used by juvenile brown trout (*Salmo trutta*) (>7 – 20cm). 256

Figure 6.5: Procedure used to define the range of substrate particle sizes that can be used by brown trout (*Salmo trutta*) Adults (>20cm)............................ 259

Figure 6.6: Summary of tolerance profiles by age class [including spawning; nursery/fry (≤7 cm); juvenile (>7 –20 cm); and adult non-spawners (> 20 cm)] for measures of substrate particle sizes used by brown trout (*Salmo trutta*).. 261

Figure 6.7 (A-G): Histogram of the proportion of streambed substrate available for spawning trout at all study sites classified as suitable and not-suitable for the total streambed (dry and wetted portions)...................................... 266

Figure 6.8 (A-G): Histogram of the proportion of streambed substrate available to young trout (≤ 7 cm) at all study sites classified as suitable, useable, and not-suitable for the total streambed (dry and wetted portions). 269

Figure 6.9 (A-G): Histogram of the proportion of streambed substrate available to juvenile trout (> 7 cm to 20 cm) at all study sites classified as suitable, useable, and not-suitable for the total streambed (dry and wetted portions). 270

Figure 6.10 (A-G): Histogram of the proportion of streambed substrate available to adult trout (> 20 cm) at all study sites classified as suitable, useable, and not-suitable for the total streambed (dry and wetted portions). 272

Figure 6.11: Habitat available for spawning brown trout (*Salmo trutta*) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on the substrate requirements... 273

Figure 6.12: Habitat available for brown trout fry (*Salmo trutta*; length ≤ 7cm) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on the substrate requirements. 274

Figure 6.13: Habitat available for juvenile brown trout (*Salmo trutta*; length > 7 to 20 cm) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on the substrate requirements. 275

Figure 6.14: Habitat available for brown trout adults (*Salmo trutta*; length > 20 cm) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on the substrate requirements. 276

Figure 7.1 (A-G): Histograms of the proportion of streambed, both total and wetted, available for spawning brown trout at all study sites classified using the integrated quality criteria (i.e. very high to low and not-useable). 296

Figure 7.2 (A-G): Histograms of the proportion of streambed, both total and wetted, available to young trout (≤ 7 cm) at all study sites classified using the integrated quality criteria (i.e. very high to low and not-useable). 300

Figure 7.3 (A-G): Histograms of the proportion of streambed, both total and wetted, available to juvenile trout (> 7 cm to 20 cm) at all study sites classified using the integrated quality criteria (i.e. very high to low and not-useable). 304

Figure 7.4 (A-G): Histograms of the proportion of streambed, both total and wetted, available to adult trout (> 20 cm) at all study sites classified according to using the integrated quality criteria (i.e. very high to low and not-useable). 307

Figure 7.5: Habitat available for spawning brown trout (*Salmo trutta*) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec). This habitat assessment incorporates depth, velocity and substrate survey data. 310

Figure 7.6: Habitat available for brown trout fry (*Salmo trutta*; length ≤ 7cm) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec). This habitat assessment incorporates depth, velocity and substrate survey data. 311

Table 7.7: Habitat available for juvenile brown trout (*Salmo trutta*; length >7 to 20 cm) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec). This habitat assessment incorporates depth, velocity and substrate survey data. 312

Figure 7.8: Habitat available for adult resident brown trout (*Salmo trutta*; length >20 cm) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec). This habitat assessment incorporates depth, velocity and substrate survey data. 313

Figure 7.9 (A-G): Histograms of the proportion of pool, run, and riffle habitat in each study reach based on criteria developed for Froude number. 315

Figure A.1 Average discharge measured in March Burn-upstream in late 2002 and early 2003. 349

Figure A.2 Average discharge measured in March Burn-downstream in late 2002 and early 2003. 349

Figure A.3 Average discharge measured in Burnhouse Burn-upstream in late 2002 and early 2003. 350

Figure A.4 Average discharge measured in Burnhouse Burn-downstream in late 2002 and early 2003. ... 350

Figure A.5 Average discharge measured in Bin Burn-upstream in late 2002 and early 2003. ... 351

Figure A.6 Average discharge measured in Bin Burn-downstream in late 2002 and early 2003. ... 351

1.0 Introduction

The European Union (EU) Water Framework Directive (WFD) (European-Union 2000) requires the Member States of the EU to assess, monitor, and where necessary, improve the ecological quality of its surface waters. The WFD sets out definitions of various surface water quality classes which describe the biological and physiochemical standards expected of 'high' 'good' and 'moderate' quality streams and rivers for four biological quality elements (phytoplankton, macrophytes and phytobenthos, benthic invertebrates and fish) and a variety of hydromorphological and physiochemical quality elements. The WFD also states that hydromorphological assessment of streams and rivers should form part of the operational monitoring programmes of EU member states (Raven et al. 2002, Davy-Bowker and Furse 2006). The WFD not only requires the Member States to protect, enhance and restore surface waters such as rivers to at least a 'good' classification, it also requires that rivers be characterized by type, based on various physical parameters such as altitude, size, geographic location and geology. Determining the ecological status relies on calibration against type-specific 'reference' conditions that can be derived either directly through establishing a network of sites that are considered 'totally or nearly totally undisturbed' or, if these are not available, indirectly through modelling or expert opinion (Raven et al. 2002, Davy-Bowker and Furse 2006).

Individual organism, such as fish, that reside in river ecosystems rely on the temporally and spatially variable physical, chemical, and biological template that occurs in these environments and can exist in that ecosystem if it possesses the proper suite of physiological, behavioural, and life history traits (Orth 1987, Poff and Ward 1990). This has been described as a multidimensional niche (Hutchinson 1957) of environmental conditions within which individuals or viable population can be sustained. The success of individuals and hence populations of organisms can be limited by single components or combinations of the physical, chemical and biological components that encompass a river ecosystem (Orth 1987, Hardy 1998). There is considerable evidence to suggest that both the quality and quantity of available habitat affect the structure and composition of resident biological communities (Poff and Ward 1990, Calow and Petts 1994) and that changes in the quality and quantity of available habitat can have adverse impacts on aquatic biota (Maddock 1999).

The management of aquatic systems; including habitat, requires an understanding of relations between the structure and function of physical systems, how the animals and plants use these structures as habitat and where these important features are likely to occur within a watershed (Naveh and Liberman 1993). Management of lotic systems is often focused on anthropogenic water use that impacts flow levels and has the potential to impacts commercially and culturally important species such as trout (Van Winkle et al. 1998). A variety of models have been developed to provide a scientifically sound and objective decision framework which links physical habitat changes with biotic components (Stalnaker 1993, Jowett 1998). Most of these methods are based on hydraulic analysis of water supply coupled with empirical observations of habitat quality. The propensity of fish to favour specific ranges of physical variables and the ability of hydraulic variables to predict current speed and water depth have been combined to predict the potential impact of changes in flow rate on fish habitat quality (Stalnaker 1994, Guay et al. 2000). The most popular method in the United States is the instream flow incremental methodology (IFIM) (Bovee 1982) and its physical habitat component (PHABSIM) (Milhouse et al. 1989) and similar methods are becoming widely used internationally (Dunbar et al. 1997, Van Winkle et al. 1998, Booker et al. 2004). These hydraulic models have proved successful in a number of applications (Jowett 1992, Jager et al. 1993, Nehring and Anderson 1993, Railsback et al. 1993). Although useful, they have attracted criticism (Mathur et al. 1985, Scott and Shirvell 1987, Gore and Nestler 1988, Crowder and Diplas 2002). One of these criticisms is that these models require expensive inputs of topographical and hydraulic measures along numerous cross-sections (Bovee 1982, Crowder and Diplas 2002) and the habitat preferences articulated and used in the model are often site specific (Lamouroux and Souchon 2002).

The habitat preference of the target species is expressed as habitat suitability curves (HSC) (Bovee 1982). These curved describe the functional relationship between physical habitat parameters and the occurrence of resident species and are produced from the observational studies of fish habitat utilization (Heggenes et al. 2002). The suitability criteria are often applied broadly as their creation can contribute to the time and expense involved in the use of these hydraulic models, however the transferability of curves developed from site-specific data has been questioned (Shirvell 1986, Gore and Nestler 1988, Hayes and Jowett 1994, Greenberg et al. 1996). There are a number of different types of HSC that have been categorized into four main types. These are: those based on literature or expert opinion, habitat utilization, habitat preference, and conditional preference curves (Waddle 2001). Of these, the literature-based curves have the most potential for transferability as they are based on a combination of sources and are intended to reflect general habitat suitability throughout the entire geographic range of the target species (Armour et al. 1984, Stier and Crance 1985, Waddle 2001).

In order to try and facilitate more transferable habitat criteria, this study is intended to develop generalized suitability criteria based on a broad survey of the literature that examined the microhabitat use of target species, brown trout, for the physical parameters of water depth, velocity and substrate particle size. These suitability criteria will then be field tested in three small Scottish burns to investigate the utility of the suitability criteria and to try and discern differences between the assessed habitat availability between the study reaches. Two reaches were studies on each burn and a before and after survey was conducted at one study reach following a large spate. This resulted in seven sets of data.

Specifically the objectives were:

1) Develop suitability criteria, based on a survey of the available literature, for the physical habitat variables water depth, water velocity, and streambed substrate.
2) Use these criteria to assess the proportion and availability of suitable values of each of these physical habitat variables in six reaches (seven data sets).
3) Use the suitability criteria to determine the quality of the physical habitat available for each of the physical habitat variables in the six reaches (seven datasets).
4) Integrate the three physical habitat variables into a single index value. Determine the availability and quality of the physical habitat at the six study reaches base don the assessment method and make comparisons between the data sets.

The focus is just the physical habitat parameters required of brown trout; the study is further restricted by concentrating on just the stream dwelling portion of brown trout's life history. The choice of habitat variables is further developed in Chapter 3. Brown trout's habitat needs change as they mature and grow (Heggenes 1988, Heggenes et al. 1999); each stage is therefore addressed separately. The size/age classifications used are: nursery (≤ 7 cm), juvenile (>7 to 20 cm), and adult resident (>20 cm). Size/age classes are variable (Table 4.2 to 4.4) and these categories seemed to typify those used in the literature. These distinctions are based on fish length as changes in habitat use seem to be related to fish size rather than age (Werner and Gilliam 1984, Persson 1991). Chapter 4 will deal with water depth, Chapter 5 will deal with water velocity, and Chapter 6 will deal with the particle size of the substrate. Chapter 7 will include the integration of the three habitat variables and provide a final overview of the amount and quality of habitat available to the four life history classes of brown trout in two study streams. A final discussion of the strengths and weaknesses of the assessment tool will be discussed in Chapter 8.

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2.0 Literature Review

2.1 Overview

The Water Framework Directive (WFD) was introduced by the European Union in 2000 and is designed to regulate the use of surface water in Europe. The ideas incorporated into the WFD borrows heavily from the currently favoured philosophy advocated for environmental protection, which can be loosely described as the ecosystem approach (Karr 1991, Haskell et al. 1992, Jackson and Davis 1994, Polls 1994, Pollard and Huxham 1998, Quigley et al. 1998, Hart et al. 1999). The ecosystem approach has been used to develop management theories referred to as ecosystem health (Miller 1984, Schaeffer et al. 1988, Rapport 1989, Page 1992) and ecosystem integrity (Karr 1991, Karr 1995). This new legislation focuses on managing surface waters in an integrated way thus considering the physical, chemical, and biological component. This chapter provides a synopsis of the WFD and ecosystem health and integrity. It reviews the concept of the niche and ideas concerning the habitat requirements of organisms. Further, an overview of the use of reference condition and physical site comparisons is provided. Lastly, the most common monitoring and habitat assessment techniques in use today are outlined and the advantages and disadvantages of each are summarized. The methods reviewed are as follows: Green's upstream-downstream model, BACI, Bioenergetic models, RIVPACS, Biotic Indices, and Hydraulic Models (including PHABSIM).

2.2 The Water Framework Directive

In the autumn of 2000 the Parliament of the European Union adopted legislation that redefined how aquatic ecosystems are protected and the way in which the use of water is regulated within the Community. The current WFD (European-Union 2000) replaces seven previously implemented Community directives that controlled the use of water resources, and oversaw the protection of the aquatic environment. The accrual of these so-called ‘first wave’ directives, the earliest of which was adopted in 1975, is intended to streamline 25 years of European Union legislation and reinforces the Communities view that water is not simply a commercial product to be exploited but is an essential component of Europe’s natural and cultural heritage; a resource that should be preserved, protected, and improved. The increasing demands on the supply of clean water, the inevitable limitation in the availability of this resource, and the publics’ growing concern for a healthy environment provided the motivation for a revised approach to the governance of Europe’s aquatic environment. This legislation summarizes the European Unions’ approach to water policy in the beginning of the twenty first century and outlines the objectives and methodologies to be used to achieve water quality goals in the foreseeable future.

The primary aim of this legislation was to provide a framework from which inland surface waters, transitional waters, coastal waters and ground waters could be protected. In addition to streamlining legislation, the current directive includes the following elements: the expansion of the extent of water protection to include all waters (surface and groundwater), the co-ordination of water quality objectives, the management of water based on the river basin model, the amalgamation of both quantitative controls and quality standards in management strategies, increasing public participation in the development of water policy and ecosystem management, and the inclusion of provisions that will ensure that the cost of providing water resources are covered by the user.

The seven water policy directives that preceded the WFD, and will be repealed after it has been fully implemented. These are: The Information Exchange Directive (European-Union 1977) , The Surface Water Directive (European-Union 1975), The Dangerous Substances Directive (European-Union 1976), The Fish Water Directive (European-Union 1978), The Drinking Water Directive (European-Union 1979a), The Shellfish Water Directive (European-Union 1979b), and The Ground Water Directive (European-Union 1979c). Under each of these treaties a specific type or class of water was protected. The current legislation not only includes all the categories of water formerly covered by these seven directives but also extends coverage to include all waters, both on the surface and underground, within the Community. The levels of specific coverage (e.g. bathing or drinking water) were not weakened and will be protected to the same extent as outlined in the former treaties. However, the current directive extends coverage to all surface waters and insists that Community members improve and protect the quality of these waters to a minimum enforceable standardized level.

The WFD requires that member states protect, enhance, or restore all surface waters at a level defined as 'Good Surface Water Status'. This status is composed of two elements: Good Ecological Potential and Good Surface Water Chemical Status. Good Ecological Potential involves determining the composition, abundance and, in some cases, age structure of aquatic flora and fauna, referred to as the ‘biological element’. In addition, it will be necessary to assess the quality of hydromorphological, chemical, and physiochemical elements that support the biological elements within these ecosystems. The quality-status of any particular water body is determined by a comparison of relevant water quality parameters with references sites that have been determined to be of ‘high status’. Typically a high status water body will have no, or very minor, anthropogenic alterations to its hydromorphological and physiochemical quality elements. In addition, the biota present will be those normally associated with each type-specific aquatic ecosystem under undisturbed conditions. Good status is assigned when the biological elements show low levels of distortion resulting from human activity and deviate only slightly from those normally associated with the surface water body type. Good Surface Water Chemical Status is defined as compliance with all standards established for chemical substances at the European level. In order to meet the requirements of Good Surface Water Status both the ecological and chemical elements must be of ‘good’ quality-status.

Groundwater quality is viewed differently to surface water. The presumption is that groundwater should not be polluted at all. Thus, chemical status requirements are not established. Ground waters are to be monitored so that any increase in levels of contaminants can be detected and reversed. As well, the importance of the quantity of groundwater is recognized and will be protected under the current directive. Only that portion of the annual recharge that is not needed to sustain connected terrestrial ecosystems, such as wetlands, may be removed.

The reference conditions for surface waters can be established by using one of two techniques. System-A divides Europe into 25 broadly defined ecoregions. Within each of these ecoregions all surface water is identified and categorized as one of six types. These include rivers, lakes, transitional waters, coastal waters, or as artificial or heavily modified surface water bodies. Surface waters are then further divided by physical, geological, or chemical descriptors such as altitude, size of water body or catchment, and salinity. For example, rivers within each ecoregions are further classified by altitude, size and geology of catchment area. Alternatively, surface waters may be classified using System-B. Using this approach, member states can classify aquatic ecosystems using physical or chemical factors that are thought to dictate or influence the structure and composition of the resident biological populations. They are then further classified by obligatory and optional factors. Obligatory factors include locator variables (e.g. altitude, latitude, and longitude), geological type, and size of catchment. Optional factors include physical or chemical descriptors that may be useful in distinguishing between different types of surface water bodies. A member state may classify lotic systems using obligatory factors that include, altitude, latitude, longitude, geology and size of catchment, along with optional factors such as distance from source, energy of flow, substratum composition or air temperature range. If System-B is used the same level of differentiation must be achieved as would be seen under System-A.

Once surface water body types are characterized, type-specific hydromorphological and physiochemical quality elements, expected of the system under high ecological status, will be defined for each water body type. In addition, high-status type-specific biological reference conditions, again based on structural parameters of aquatic flora and fauna, will be established. Type-specific reference conditions may be determined using a number of mechanisms. These include the following: 1) A system of spatially based reference sites, 2) reference conditions established using either predictive or hindcast models that employ historical, palaeological and other available data, 3) a combination of both spatially based and modeling methodologies, or 4) the use of expert judgement. If a spatially based approach is employed, a network of reference sites for each surface water body type will be established. The network will incorporate enough high-status sites that sufficient confidence can be established as to the value of the reference conditions. For surface water bodies that are artificial or heavily modified, reference conditions will be considered to be those that meet a system's maximum ecological potential.

The identification of type specific surface water body types will be conducted in the context of river basins. Within each ecoregion individual river basins are to be identified and assigned to specific River Basin Districts. River Basin Districts will be analyzed for their hydromorphological, physiochemical and biological characteristics and assed for the impact of human activity on surface and groundwater, while an economic analysis of water use will be conducted. For each River Basin District a management plan will be prepared which will contain information regarding the location, boundaries and type of surface water bodies present, identify reference conditions, summarize significant anthropogenic impacts on these water bodies, include an economic analysis of water use, and identify the ways in which the objective of good-status surface and groundwater will be achieved.

The implementation of the WFD is to be conducted under the scrutiny of the public and is to involve the participation of all interested parties. In particular, this is to include the production, review and updating of River Basin Management Plans. This is to ensure that a balance is reached between interested groups within society and to aid in the enforcement of the legislation. The directive also includes provisions that allow for member states to charge for the cost of abstraction, distribution and the collection and treatment of wastewaters. The motivation for cost-recovery is to reinforce the notion that water and aquatic ecosystems are to be conserved and protected. Full details of the requirements of the WFD are outlined in the European Communities Official Journal (European-Union 2000) and are summarized in other Commission publications (European-Communities 2002).

2.3 Ecosystem Health and Integrity

The ideology embodied in the WFD appears to borrow heavily from the currently favoured philosophy advocated for environmental protection, which can be loosely described as the ecosystem approach (Karr 1991, Haskell et al. 1992, Jackson and Davis 1994, Polls 1994, Pollard and Huxham 1998, Quigley et al. 1998, Hart et al. 1999) The European Union has incorporated components such as management conducted within river basins, public participation and accountability, and the protection and preservation of water resources. This reflects ideas such as the recognition of biotic complexity, inclusion of humans within ecosystems and biological sustainability, which are key aspects of ecosystem based management (Marmorek et al. 1992). Although a specific definition of the ecosystem approach is elusive, this management protocol generally includes a holistic perspective that embraces multiple levels of biological, managerial and political structure, the integration of existing and historical data, is ecologically sensitive and forward looking, and is ethically correct (Christie et al. 1986, Cash 1995).

More specifically, ecosystem based environmental management contains a number of key tenets. The first of these is that humans and human activities are recognized as important and influential components of ecosystem structure and function. The advocates of this approach reject the historically held notion that human beings are separate or somehow removed from natural systems (Cash 1995). With this recognition comes the realization that the well being of our society is often heavily dependent on the resources that are provided by these systems (Loeb 1994). The detrimental effects of the misuse of these resources, resulting in the loss of environmental services and injurious consequences to other species that share these environs, suggests not only a stake in the well-being of these systems but that we should provide a stewardship role on behalf of other non-human users. Thus, the interests in our society will vary from those concerned with the extraction of resources to those who advocate the protection of ecosystems in an unaltered state. The ecosystem approach recognizes the multiplicity of interest in the state or condition of natural systems and promotes the inclusion of all interested stakeholders in the creation of environmental policy (Rapport et al. 1985b, Martinka 1992, Steedman 1994, Cash 1995, Karr 1995).

Concern regarding the status of natural systems illustrates society's wish to maintain ecosystem at a desired state. This concern is reflected by the Good Status Surface Water designation outlined in the WFD. The desired condition of an ecosystem may vary from one stakeholder to the next but it provides environmental managers with goals from which a management framework can be constructed. The development of such a framework will include an assessment and monitoring program designed to gauge an ecosystem's condition. Ecosystem based management with properly designed monitoring programs contains a number of important features. Included is a holistic approach that attempts to incorporating the complexity inherent in the abiotic and biotic components of ecosystems as well as the intricacies of the many-layered political and managerial structure of our societies. Data collected from the monitoring program will be utilized along with historical information to help identify elements at a location that are responsible for a reduction in environmental quality. This information can provide insight into potential areas of concern and give direction for the development of future management objectives and the long-term maintenance of natural systems (Regier 1992).

The concept of a desired state for natural systems speaks strongly to advocates of an environmental management framework known as Ecosystem Health. In the simplest terms, this concept makes an analogy between the state of an ecosystem and the condition or status of a mammalian body (Miller 1984, Schaeffer et al. 1988, Rapport 1989, Page 1992). Health in a mammalian body exists when an organism is functioning properly and any derivation from that optimal state is considered a negative or unhealthy change. An organism uses homeostatic processes that respond to internal and external influences so that a state of optimal function is maintained. This is made possible because organ systems are integrated and dependent. The failure of one system can lead to the failure of other systems and ultimately the death of the organism. The functioning of these internal systems varies little between individuals so by monitoring vital signs of an organism, such as the pulse rate, temperature, or responsiveness, the condition of an organism can be determined (Calow 1976). Like a mammalian body, a healthy or normally functioning ecosystem is the desired state or target condition sought by ecosystem managers. Differing ecosystems show similar responses to ecosystem stress. These responses may include changes in nutrient cycling, species diversity and composition, and disease incidence (Rapport et al. 1985a). It is thought that by monitoring key structural or functional components of a natural system, deviation from the desired state can be assessed (Schaeffer et al. 1988). This concept holds obvious appeal in that the notion of health can be readily understood and appreciated by policy makers, scientists, industry representatives and members of the public alike. The advantage to environmental managers is that a conceptual endpoint of ecosystem condition can be established and agreed upon by multiple stakeholders with disparate views; from this point a management framework can be constructed (Regier 1992).

The use of the concept of ecosystem health in environmental management has been discussed in the literature for the last fifteen to twenty years (Costanza et al. 1992, Woodley 1993, Cash 1995). The concept that ecosystems resemble organisms or possess characteristics usually attributed to organisms is even older. The earliest record of this type of thinking comes from the Scottish geologist and physician, James Hutton, who viewed the earth as a superorganism capable of self-management (Hutton 1788). Calow (1992) has interpreted the superorganism concept as ecosystems being similar to a machine in which all components fit and work together to maintain a predetermined optimal endpoint. Elements of organism-based characteristics are found in Clements theory of succession (Clements 1936). Clements viewed the development of ecosystems as a tightly regulated predictable progression through a series of stages, each dominated by specific taxa, eventually reaching a climax state. This would be similar to the development of an animal from an embryo through to maturity. A current but extreme view of ecosystem health is that healthy human bodies and biotic communities are self-adjusting and exist in optimum equilibrium maintained by feedback mechanisms (Ferguson 1994).

Although discussed frequently, a clear and concise definition of this concept has been difficult to formulate and has stimulated much debate in the scientific literature (Rapport et al. 1985a, Chapman 1992, Ferguson 1994, Shrader-Frechette 1994, Steedman 1994, Karr 1995). A recent review by Xu and Tao (2000) begins by quoting the differing definition of ecosystem health used by seven authors who are either actively involved in developing this concept or have recently published commentary regarding its use in ecology or environmental management. Examples of these are included in Table 2.1. Costanza (1992) has identified several contexts in which the term has been used. 1) Health as homeostasis, or the ability of a system to maintain itself within a range of "normal variation", 2) Health as the absence of disease, 3) Health as diversity or complexity, 4) Health as stability or resilience, which refers to a systems ability to resist perturbation or to recover quickly from perturbation, 5) Health

Table 2.1: Definitions of ecosystem health outlined by Xu and Tao (2000)

Author	**Definition**
Karr (1986)	A biological system, whether individual or ecological, can be considered healthy when its inherent potential is realized, its condition is stable, its capacity for self-repair when perturbed is preserved, and minimal external support for management is needed
Page (1992)	A harmonious relationship among the parts of the body and between the body and the outside world; and the concept of homeostasis, taught in medical schools as a normative concept, has a direct parallel with the notion of stability in ecosystems
Ulanowicz (1992)	A healthy ecosystem is one whose trajectory towards the climax is relatively unimpeded and whose configuration is homeostatic to influences that would displace it back to earlier succession stages.

as vigour or scope for growth, which measure a systems activity and resilience, and 6) health as a balance between components.

Discussions concerning Ecosystem Health regularly involve ecologists and environmental managers, although workers in other fields have contributed to its development (Davis 1983, Dwivedi and Sankar 1991, Nielsen 1992, Beasley 1993). The multidisciplinary nature of this concept was reflected in a workshop, which took place in the early 1990's, and was attended by ecologists, philosophers, economists, and social thinkers (Costanza et al. 1992). Those who attended this gathering addressed some of the theoretical and applied issues relating to the use of this concept in science and environmental management, including the ambiguity surrounding its definition. From these discussions, an operational definition emerged. Haskell and his colleagues (1992) reports this to be as follows: "An ecological system is healthy and free from 'distress syndrome' if it is stable and sustainable – that is, if it is active and maintains its organization and autonomy over time and is resilient to stress". The definition is to be applied to all complex systems and contains four key tenets for inclusion when applied to ecosystems: 1) sustainability, 2) activity, 3) organization and 4) resilience. This definition is meant to serve as a starting point for research and discussion and does not purport to be either final or wholly inclusive.

Frequently coupled with the term health is the concept of ecosystem integrity. Although used jointly, ecosystem integrity is generally thought of as being a separate but related concept. This distinction was made most clearly by Karr (1991, 1995) who defined integrity in terms of conditions "at sites with little or no influence from human actions; the organisms living there are products of the evolutionary and biogeographical processes influencing that site". Karr further developed this idea by stating that " integrity refers to the capacity to support and maintain a balanced, integrated, adaptive biological system having the full range of elements and processes expected in the natural habitat of a region" (Karr 1995). This contrasts with health, which includes the influence of human activity and attempts to manage or maintain ecosystems at a predefined desired state such as cultivated areas or parks. This distinction seems to have been incorporated into the requirements of the WFD. Integrity equates with the High-Status designation, which permits no or minor anthropogenic alterations to the biological, physiochemical and hydromorphological quality element. Ecosystem health resembles Good-Status in that biological elements may show levels of distortion resulting from human activity but may deviate only slightly from normative conditions. The use of integrity and its differing meaning from health are a useful way to distinguish between discussion that include human influence in ecosystems and those that do not. The concept of integrity has been reasonably well accepted (Steedman 1994, Cash 1995, Scrimgeour and Wicklum 1996, Meyer 1997). There has been some resistance to the use of this term largely because some investigators feel that no system is completely untainted by humans (Wicklum and Davies 1995). However, conceptually ecosystems integrity is a condition that can be readily appreciated, although perhaps difficult to identify. However, it is a state that has existed, can be used as an endpoint when gauging human influence and may be a target to be sought by environmental managers in the future.

In contrast, the concept of ecosystem health has been much more controversial and is far from being universally accepted. One of the first investigators to express doubt about the utility of this approach in either environmental management or ecology was

Peter Calow (1992). This was followed by well-articulated critical reviews of the concept by Suter (1993), Wicklum and Davies (1995), and Scrimgeour and Wicklum (1996). The primary concern is with the use of the health analogy and its attempt to compare human health with a discernable ecosystem state. As previously described there is a great deal of variability in the interpretation and applications of this analogy (see: Constanza 1992). Calow (1992) recognized this and classified the variable use of the concept into weak and strong forms. The weak form of the analogy is the interpretation that health is a normal state and illness or unhealthy ecosystems are a departure from normal conditions. The strong form of the analogy takes a much more strict tone in that it more closely identifies the health of an ecosystem with that of an organism. In this usage health defines an optimal or favourable condition for the function of a whole organism that is actively defended by homeostatic processes. Ideally this optimal state should be generalized between individuals so that specific health criteria can be objectively defined. Not surprisingly, much of the criticism of the ecosystem concept focuses on interpretations based on the strong form of the analogy.

This strong form seems to imply that ecosystems not only resemble organisms but are in fact entities structured at a higher level of organization that are tightly integrated; something like a super-organism (Miller 1984, Schaeffer et al. 1988, Rapport 1989, Calow 1992, Suter 1993). Advocates of ecosystem health deny this association; however, critics insist that if the analogy is appropriate then some characteristics of ecosystems must resemble that of a human or mammalian body (Calow 1992, Suter 1993, Wicklum and Davies 1995). A healthy ecosystem suggests that the system has an optimal state and that this condition can be defended and maintained. This is the idea of sustainability and resilience mentioned by Haskell and others (Karr et al. 1986, Haskell et al. 1992, Page 1992). If ecosystems have defendable optimal states then they must have equivalent mechanisms to defend this state such as the integrated organ systems and homeostatic feedback mechanisms found in mammals. Also, if ecosystems react to stress in a similar and predictable way then ecosystems of a specific type should have consistent structures, regular and predictable development, and distinct identities. Many ecologists would argue that these attributes cannot be attributed to ecosystems (Suter 1993, Cash 1995, Wicklum and Davies 1995, Scrimgeour and Wicklum 1996).

To begin, in order to maintain an optimal state an ecosystem must be able to sustain itself in some sort of dynamic or stable equilibria. The short temporal and relatively small spatial scales that ecological studies have traditionally been conducted would suggest that this is the case (Golley 1993). However, ongoing long-term studies and palaeoecology investigations reveal that ecosystems are in a constant state of change and the appearance of stability may be an artefact of our investigative techniques and personal perspectives (DeAngelis and Waterhouse 1987). Data from long term studies show a continuum of temporal variability in the dynamics of population and community with no clear demarcations of equilibrium states (Connell and Sousa 1983). Thus, the scale, both temporally and spatially, at which change should be measured is difficult to define (Scrimgeour and Wicklum 1996).

If stable states do occur and a system can be described as being resistant and resilient to environmental stress then, if the health analogy holds, some mechanism must be in place to maintain an ecosystem's equilibrium. Unlike humans or other mammals, ecosystems do not have active mechanisms like neural networks or hormonal systems that actively respond to stimuli and make active adjustments accordingly. They may seem to maintain equilibria at some spatial and temporal scale but the mechanisms involved are passively organized (Calow 1976). Additionally, active feedback mechanisms would require a tighter level or integration that is present in natural communities. A system may be interconnected structurally, for example in terms symbiotic or predator prey relationships, but the removal of a population of a specific species will not necessarily result in the demise of an ecosystem. This differs from organisms where the removal of a vital organ like the heart will result in death. An ecosystem will change with the removal of species, particularity keystone species, but it will continue to exist. Spatially, ecosystems do not have clearly defined boundaries such as the skin or cortex of an organism. As well, consistent structures are difficult to find. It is possible to categorize an ecosystem into broad types such as boreal forests, lakes, or rivers. However, the structure and function within specific systems will vary from example to example. Finally, mammalian systems will develop from an embryo to a mature individual through a series of predictable and consistent stages. At one time it was thought that biotic assemblages, particularly plant communities, developed in a predictable succession of one dominate community followed by another until a climax community was established (Clements 1936). However, most modern ecologists, beginning with Gleason (1939), reject this notion (Whittaker 1957, Engelberg and Boyarsky 1979, Simberloff 1980, McIntosh 1985, Botkin 1990).

An additional line of argument against the ecosystem health concept centres on evolutionary principles. The development of an individual organism is controlled by genotypic programming that has resulted from evolutionary pressures (i.e. natural selection). The attributes of individual organisms and the resulting characteristics of populations of these organisms have been programmed into individuals and can be passed on to offspring through that organism's genetic material. There is no evidence to suggest that natural selection operates at the community or ecosystem level of organization. Thus, there is no mechanism for these systems to develop tightly integrated systems, feedback mechanisms, or consistent structures (Krebs and Davies 1978, Stearns 1980, Sibley and Calow 1986, Cockburn 1991, Calow 1992, Ferguson 1994).

As discussed most advocates of ecosystem health reject the idea that ecosystems are organisms or superorganisms (Scrimgeour and Wicklum 1996). Even the weak form of the analogy has attracted some criticism. This form of the analogy still implies that ecosystems exist in a desirable or healthy state and that this state can be identified. This state would have to be defined subjectively because, as already discussed, ecosystems may not exist in a state of equilibrium. Critics would argue that science could not be used to prove or disprove that a system is either healthy or unhealthy as these are subjectively defined (Wicklum and Davies 1995). However, if the ecosystem approach is used and the ecosystem health concept is employed in the development of a framework for environmental management those human derived definitions of optimal states would still be considered to be valid (Karr 1995, Scrimgeour and Wicklum 1996). Rapport (1989) argues, "Judgements on ecosystem health also involve taking into account more than strictly ecological functions (e.g. consideration of the human uses and amenities desired from the system)". The use of science and scientific techniques may be employed in the assessment of the ecosystem to determine if it meets criteria defined subjectively by environmental managers. As well, the consequences of decisions and causes of ecosystem degradation may be used by traditional and newly developed methodologies based on the scientific method.

Others argue that the use of the concept of health may give a false impression of the nature of ecological systems and lead to bad management decisions. Morover, subjective definitions of ecosystem health may excuse exploitation of natural resources and lead to further environmental degradation. However, the ecosystem approach that is embodied in the concept of ecosystem health and integrity clearly advocates the use of scientific methodology and current understanding of ecosystem function in the management of these systems. Clearly defined management goals do not preclude the use of common sense or of an ethical treatment and appreciation of environmental resources and natural systems (Steedman 1994).

Some ecologists have criticized the concept of ecosystem health and integrity as having limited utility in a strictly scientific or purely ecological context. However, they have found a great deal of acceptance as a tool in environmental management (Karr et al. 1986, Rapport 1989, Schaeffer 1991, Costanza et al. 1992, Suter 1993, Ferguson 1994, Karr 1995). The concept of health has been the subject of numerous papers in the scientific literature, spawned a journal (Journal of Aquatic Ecosystem Health) and has been the focus of a number of workshops. Ecosystem health and integrity has also been incorporated into environmental legislation in a number of countries, including the Water Pollution Control Act (U.S.A.), the U.S. Clean Water Act, Canada's National Park Act, and the Great Lakes Water Quality Agreement. It seems clear that this concept is well established and may find some utility with environmental managers, thus deserves our consideration.

2.4 Habitat Requirements and the Niche

The term habitat is widely used, not only in ecology but elsewhere. It is generally understood to mean simply the place where an organism lives. Thus, an organism's habitat requirements are the abiotic features of the environment necessary for the persistence of individuals or populations. Habitat requirements can be distinguished from other aspects of the environment that are less critical, insofar as changes in their availability will have little or no effect on an organisms abundance or ability to persist in the environment. Habitat may also refer to the place occupied by an entire community. Habitat in this case consists mostly of physical and abiotic components of the environment utilized by a population or community of organism (Rosenfeld 2003). Further, Odum (1971) discusses examples where the biotic community may also be considered components of habitat extending the definition to include both the biotic and abiotic components that are utilized in the physical space occupied by an organism.

The ecological niche, on the other hand, is a more inclusive term that includes not only the physical space occupied by an organism, but also it's functional role in the community (Odum 1971). The naturalist Joseph Grinnell coined the term niche in 1917 in his paper 'The niche relationship of the California Thrasher'. Grinnell used the word niche in reference to a species 'ultimate distributional unit' within which each species is held by its structural and instinctive limitations. He also put forth the idea that no two species in the same general area can occupy for any length of time the same ecological niche (Ginnell 1917). Thus, Grinnell thought of the niche mostly in terms of microhabitat, or what we would now call the spatial niche. It wasn't until 1927 that Charles Elton gave the first working definition of the niche concept. Elton was one of the first to begin using the term niche in the sense of the functional status of the organism within a community. Since Elton placed the emphasis on energy relations, his version of the concept might be considered the trophic niche (Elton 1927). In 1957 G.E. Hutchinson suggested that the niche could be visualized as a multidimensional space or 'hypervolume' within which the environment permits an individual or species to survive indefinitely. Hutchinson's niche, which is though of as the multidimensional or hypervolume niche, is amenable to measurement and mathematical manipulation. Hutchinson also made a distinction between the fundamental niche or the maximum 'abstractly inhibited hypervolume' when the species is not constrained by competition with the realized niche, a smaller hypervolume occupied under biotic constraints (Hutchinson 1957). In instances where a species has a complex life history, for instance, larvae, juveniles, and adults, that occupy different habitats, then different life history stages can be defined by different habitat requirements and corresponding niches (Hutchinson 1957, Odum 1971, Moyle 1997, Rosenfeld 2003).

Habitat selection occurs when an organism avoids a particular habitat or uses a habitat in greater proportion than it's availability in the environment. Habitat selection can be demonstrated, for example, if fish occur at higher densities in particular habitats or if fish occur at higher frequencies in particular microhabitats to the relative frequency of that microhabitat in the environment (e.g. fish occupy specific depths more frequently than those depth are found in the environment). Observations of habitat selection can be used to infer habitat preference but preference is more clearly determined in habitat choice experiments in which factors that can influence habitat selection such as predation risk, competition, and habitat availability are controlled. Habitat selection in natural environments represents the habitat preference under the conditions present (realized niche). True habitat preference is the habitat that would be selected under ideal conditions (i.e. in the absence of predation, completion: Hutchinson's fundamental niche (Rosenfeld 2003).

2.5 Reference Conditions

In the simplest terms, the control in an experimental investigation is the experimental factor left untreated from which differences in response within the treatment factors can be discerned and, if the experiment was properly designed, insight into the functioning of the system of interest can be drawn. Typically, a control must undergo the same procedures as the treatment to ensure that the results (procedural treatment), and the conclusions drawn from those results, are not an artefact of the experiment itself. To illustrate this point, Underwood (1997) gives a fictitious example in which mice are injected with a drug that is designed to influence heart rate. Handling the mice, injecting the drug, and measuring the heart rate are all stressful events for the mouse and will affect the animal's heart rate. If the same handling and injection procedures are not carried out with the control animals using a placebo it will be difficult or impossible to make inferences regarding the cause of the animals change in heart rate (i.e. are they due to the drug, the handling or both). Ecological experiments often have a further complication. Because there is so much inherent variability in natural systems distinctions between factors and inferences regarding causation becomes increasingly difficult; if a difference in response can be detected, it may be either due to the treatment applied, an artefact of the experimental design or natural variation within the system. To help clarify this situation a third type of factor may be introduced into the experimental design. In this type of ecological investigation a site or sites are left untreated and un-manipulated in any way so that a comparison with the control treatments can be made. If different responses arise between the control and pristine sites then the results observed may be an artefact of the experiment.

Hurlbert (1984) expands on this idea by including other forms of control into the experimental design in order to minimize what he calls ‘sources of confusion’. This includes randomization controls, replication controls, and interspersion controls. These aspects of experimentation, which Hurlbert describes as ‘obligatory design features’ are put in place to try and eliminate bias and unintended effects that may creep into an experiment. The source of experimental confusion includes temporal change, procedural effects, experimenter generated variability and bias, variability among experimental units, and non-demonic intrusions. A summary of the sources of confusion and the features of experimental design that can be used to reduce or eliminate them are presented in Table 2.2 (redrawn from Table 1, Hurlbert 1994); a review of these ideas are examined by Hulbert (1984) and Underwood (1997).

Control, in the experimental context, can also refer to the conditions under which an experiment is conducted. This type of control may manifest as the homogeneity of experimental units, uniformity and consistency in which treatment procedures are applied, or the regulation of the environment within which experiments are conducted. An example using the mice investigation would involve an investigator ‘controlling’ an experiment by using genetically similar stock, conduct experiments in identical environments (e.g. temperature, feeding regimes), and by injecting the same volume of treatment drug into the same tissues or blood vessel on each organism. Of course the ability to control environmental conditions, which is possible to varying degrees in a laboratory setting, becomes almost impossible when investigating a system as complex

Table 2.2: Potential sources of confusion in experiments and means for minimizing their effect (Redrawn from Table 1 pp.191 *in* Hurlbert 1984).

Source of Confusion	Features of experimental design that reduce or eliminate confusion
Temporal Change	Control treatments
Procedure effects	Control treatments
Experimenter Bias	Randomized assignment of experimental units to treatments Randomization in conduct of other procedures “Blind “ procedures*
Experimented generated variability (random error)	Replication of treatments
Initial or inherent variability among experimental units	Replication of treatments Interspersion of treatments Concomitant observations
Non-demonic intrusions§	Replication of treatments Interspersion of treatments
Demonic intrusions	Eternal vigilance, exorcism, human sacrifices, etc.

* Usually employed only where measurement involves a large subjective element
§ Non-demonic intrusion is defined as the impingement of chance events on an experiment in progress

and variable as an ecosystem. Not only does an investigator have little or no control over environmental conditions, but there may not be sites available that are un-impacted or pristine so that un-treated controls could be established. As well, difficulties may arise in establishing sites with a high enough degree of homogeneity so that procedural controls may be conducted. This lack of homogeneity or ability to control naturally occurring variability will then affect an investigators ability to implement other design features such as replication, randomization or interspersion.

The Water Framework Directive envisions the establishment of sites or type specific biological reference conditions within each of the established ecoregions. It is to these sites or reference conditions that comparisons can be made, and from which the quality status of a water body will be determined. This type of reference condition would be classified as the ‘un-treatment’ control defined by Hurlbert and Underwood. It is important to point out that the Water Framework Directive is being established in order to develop monitoring and assessment protocols not to outline procedures for use in experimental manipulations. However, comparisons will be made between sites or specific conditions; if this is to be done objectively, statistical analysis must be used. Experimental manipulation will most likely not be conducted in the course of routine monitoring but the requirements for the control or reference conditions that are established and the mechanisms used to make those comparisons must conform to the protocols that have been developed for traditional scientific experimentation. A number of questions immediately arise when these types of comparisons are considered: 1) Do pristine conditions actually exist? 2) Do different sites or ecosystems resemble each other structurally and functionally so that valid comparisons can be made? 3) Can comparisons be made so that Hulbert's ‘obligatory design features’ may be included in a monitoring and assessment protocol? 4) If pristine sites cannot reasonably be found, how can reference conditions be established?

Pristine, defined in a dictionary, is as follows: 1) of or involving the earliest period, state, etc.; original. 2) Pure, uncorrupted. 3) Fresh, clean, and unspoiled. (Anonymous 1999). When applying this definition to natural systems one can easily imagine a Garden of Eden style ecosystem functioning in a balanced and harmonic manner in which man lives within the constraints of the natural order and has little or no impact on the structure or function of these systems. Currently, this idea of man’s relationship with the environment is generally considered a myth. Even in regions which have only recently been developed and exploited by modern industrial and agricultural practices, such as North America, there exists evidence that the aboriginal populations manipulated ecosystems and landscapes for hunting and agricultural purposes (Day

1953, Foreman and Russell 1983, Denevan 1992). In areas with a longer history of environmental exploitation and manipulation, such as Europe, locating areas free of even minor human manipulations is challenging. Petts (1989) describes the manipulation of rivers in Flanders, Germany, France, Italy, and England, though the construction of weirs used for water power beginning as early as 1250 CE. In Europe, at the end of the twentieth century, all major river systems have been altered in some manner and even in remote alpine regions an average of only 10 % of river length can be considered semi-natural (Ward et al. 1999). Difficulties arise outside of even heavily developed regions such as Europe. In remote areas that have been exposed to limited human development, global environmental change, such as elevated levels of CO_2 in the atmosphere (Emmerson et al. 2005) and the presence of synthetic chemical compounds in the polar regions (MacDonald et al. 2000) would suggest that truly pristine ecosystems no longer exist.

2.5.1 Physical site comparisons

Despite the patterns discussed above, there is still a need to compare sites or environmental conditions that have been degraded by human activities with those that are relatively un-impacted. Examples of investigations and commentary that utilize more functionally defined pristine reference sites are commonly found in the literature. These include work on an alpine stream in Italy (Ward et al. 1999), streams in redwood forests in California (Welsh and Ollivier 1998), discussions on the assessment of river health (Townsend and Riley 1999), and investigations of hydrologic processes (Allan et al. 1993). Conceptually, pristine sites used in these types of investigations are similar to what Karr defines as ecosystem integrity [see above: Karr (1991, 1995)]. Included in Karr's definition of integrity are three important components: 1) that the biota within these systems span a variety of spatial and temporal scales, 2) that the these systems include an array of kinds of things (elements of biodiversity) including the processes that generate and maintain them, and 3) that these systems are embedded in a dynamic evolution and biogeographical context.

This definition seems to allow for slight perturbations resulting from human activities and outlines the essential components that must be considered if an ecosystem is considered to possess ecological integrity. An ecosystem has ecological integrity if an ecosystem's structure is present and is functioning as would be expected based on the evolutionary and geochemical context in which it is placed. If an ecosystem possesses these characteristics then it can be used to gauge the status of similar systems. We have already determined that in many regions, particularly Europe, these sites probably don't exist. However, the Water Framework Directive allows us to determine the structural and functional character of an ecosystem using other techniques such as predictive or hindcast models which are based on historical, paleoecological or expert judgement. Can these systems be used to determine what an ecosystem had once or should look like?

2.6 Single Point Impact Assessment

The problem of attributing causation is well illustrated in the literature, particularly in the publications surrounding a seemingly simple impact assessment methodology that was proposed by R.H. Green in 1979. Green had compiled a manual that outlined sampling methodologies and statistical designs that could be used in ecosystem level investigations (Green 1979). This text was generally well received (see comments by Hurlbert 1984, and Steward-Oaten et al., 1996); however one design, which involved the investigation of point-source discharges into rivers, was criticized by Hurlbert in his paper on pseudoreplication (Hurlbert 1984). This initial criticism resulted in a flurry of papers that developed and expanded this idea that eventually led to two differing statistical approaches designed to assess the impact of a point source discharge.

2.6.1 Green's upstream-downstream model

Green suggested that the affects of an outfall could be investigated by taking samples of a representative biotic indicator, such as the abundance of a particular species of invertebrate, at sites upstream and downstream of the point source discharge at periods both before and after a new instillation began to discharge effluent. The nearby upstream site, which would presumable have a similar abundance of the species under study and be exposed to a similar regimen of biotic and abiotic variability, would act as a control. Any observed differences in abundance at the impacted site after the discharge was initiated, outside of the naturally occurring variation, could be attributed to the obvious anthropogenic source. The statistical treatment of the data would include a two-way analysis of variance (ANOVA) (McGhee 1985) and a significant ‘areas-by-times’ interaction between the two factors was thought to demonstrate causality.

Hurlbert disagreed. Hurlbert’s criticisms (again, outlined in the 1984 article on pseudoreplication) focused on the inability to employ or include his ‘obligatory design features’ into experimentation or sampling protocols in rivers. This is a reoccurring problem in research and monitoring programs in lotic ecosystems. Rivers, particularly larger systems, are unreplicated. Because of the lack of replicate systems, treatments cannot be interspersed or assigned randomly and any number of samples taken at the two sites are not replicate samples but, in fact, pseudoreplicates. As well, because of the fluvial nature of these systems, events or occurrences at one site can influence the structure or function at others (Vannote et al. 1980). This generally occurs from upstream to downstream, however, there are instances of downstream process impacting sites further upstream (McDowall 1998). These site-to-site relationships, of course, invalidate any assumptions regarding independent samples, a key requirement when conducting an ANOVA.

Hurlbert stresses that replication is unnecessary if it can be demonstrated that the individual experimental units being used in any investigation are identical. Not only do they begin an experiment the same but that they remain the same and behave the same throughout the course of the experiment. The only deviation permitted is the effect of the treatment being imposed on the system. Hurlbert recognizes and accepts the unique nature and the tremendous amount of variability between and within ecosystems (including site-to-site variation in rivers). He stresses that for this sort of analysis to be valid the differences between upstream and downstream location must remain constant over time if no wastes were being discharged or if there was no effect. In other words, it is not necessary for the sites to resemble each other precisely but it is important that they respond to the natural biotic and abiotic influences on their structure and function in the same way. Hurlbert insists that this assumption is unreasonable and that the use of ANOVA in this instance is unacceptable. Because inferential statistics cannot be used any observed changes in species abundance at the downstream site after the onset of discharge cannot be attributed to the outfall directly (i.e. we cannot determine causality); however, we can demonstrate that the sites differ.

2.6.2 Before-after-control-impact (BACI)

Not unexpectedly, Hurlbert was challenged in his assessment of Green's Before-After-Control-Impact (BACI) design. This challenge was published in 1986 by Stewart-Oaten, Murdoch, and Parker (Stewart-Oaten et al. 1986). These authors agreed that Green's original design was not 'optimal' but believed with some modifications it was still possible to use inferential statistics to determine if an impact had occurred from a point source discharge. These authors disagreed fundamentally with how the analysis was perceived and offered a solution to Hurlbert's 'insurmountable' problems with the design. To begin, Steward-Oaten and his colleagues contend that impact assessment problems are generally concerned with the effect of a specific project in a specific area. They are not concerned with making generalizations about classes of impacts (e.g. pulp mills, agriculture, municipal outfalls) on types of ecosystems (e.g. rivers, boreal forests, estuaries). Because the intent is to examine the influence of a particular cause on a particular ecosystem, replication (and randomization) on a broader scale is unnecessary. The variability, which is of interest, occurs at the local level and making statistically based inferences regarding natural and anthropogenic variation is perfectly valid.

The authors continue by describing how an impact can be perceived in the environment and how anthropogenic variation can be discriminated from natural cycles and random fluctuations. Based on Green's original design, abundance is estimated at sites up and downstream of a point source discharge both before and after the commencement of effluent release. The factor of interest is the difference between the abundance at the impact and control sites both before and after discharge begins. The simplest view is that abundance levels would remain constant over time (and the difference between them); with the expectation that the difference in species abundance between the control and impacted sites would decrease if the effluent had a negative impact on the system (Figure 2.1A). This 'naïve' view was further developed to account for regular or systemic variation in species abundance through natural seasonality or regular patterns cause by the effluents themselves (Figure 2.1B). Of course, random variations from chance events (e.g. spates, births, deaths, movements of individual or groups and so forth) will produce a species abundance curve that is not regular and is, in fact, quite erratic (Figure 2.1C). This more realistic scenario will produce a variable species abundance estimate with time. However, a noticeable decrease in overall abundance will still indicate a negative influence from the effluent; a difference that can be discriminated with the use of a simple comparative statistical test such as a t-test.

Steward-Oaten and his collaborators stress that the impact will have a regular and ongoing influence on the biota downstream of the outfall and that an estimate of the difference between the systemic patterns, illustrates in Figure 2.1B, in species abundance is required for the assessment. They also explain that the actual abundance observed is just one of many abundances possible or that could be realized depending on the interaction and outcomes of random influences [i.e. if systemic factors were held constant and the scenario could be replayed there are a number of possible jagged lines (actual populations) that would vary around the regular systemic fluctuations]. They remind us that samples taken at either site at a single point in time will estimate the mean and variability around the actual population abundance (Figure 2.1C) and that this is a reasonable estimate of the mean abundance that would be observed in the absence of random factors. Apparently, Green's error was to use the variation among these samples to estimate the degree to which the actual abundance might fluctuate about the

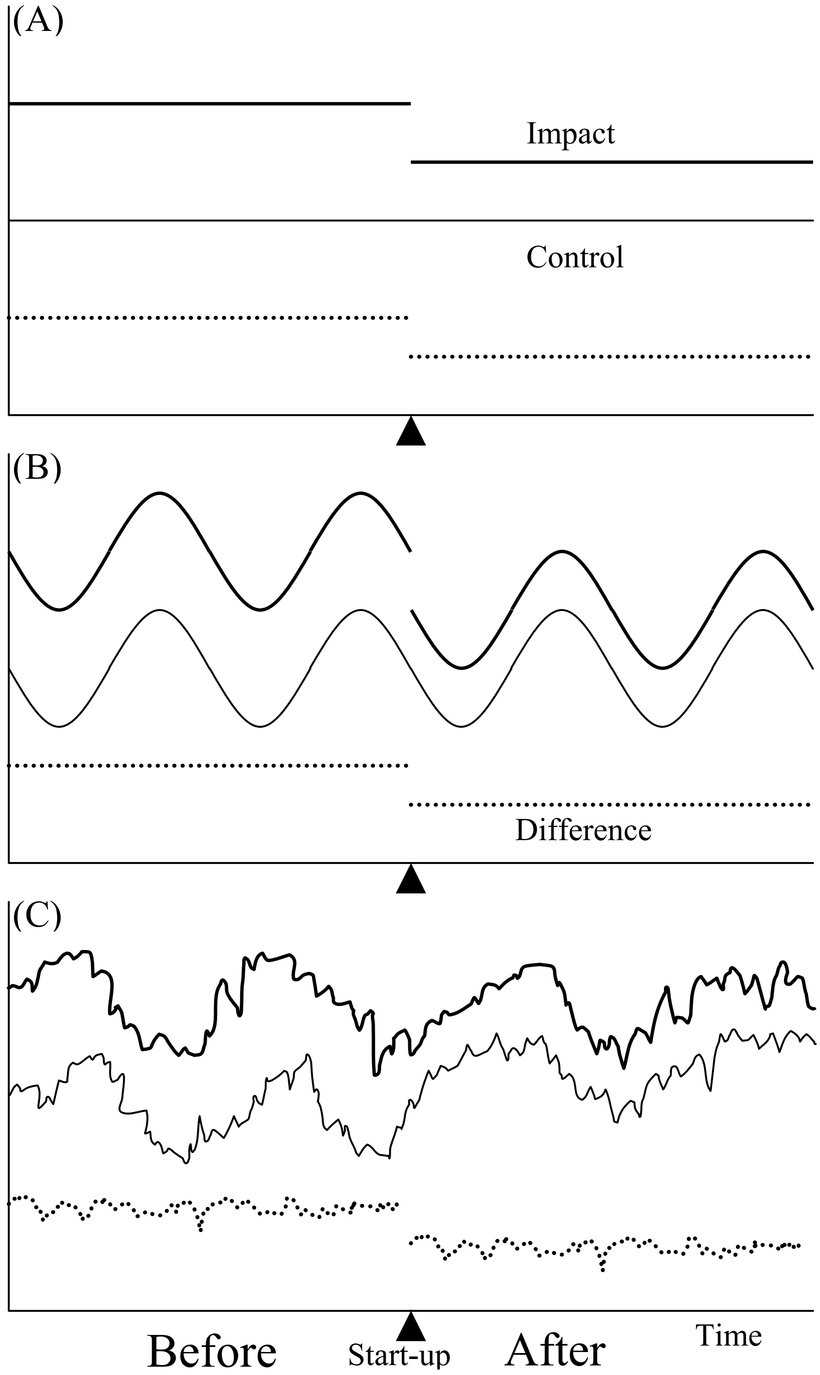

Figure 2.1: The abundance of "Species X" at the Impact and Control stations, and the difference of the abundances, as functions of time, in three versions of Impact assessment. (A) In the most naïve view each station's abundance is constant except for a drop in the Impact station's abundance when the discharge begins. (B) In a more plausible but still naïve view, the abundance fluctuates (e.g. seasonally), but the difference still remains constant except when the discharge begins. (C) In a more realistic view, the abundances fluctuate partly in synchrony and partly separately: the former fluctuations disappear in the differences but the latter remain, and effects of the discharge must be distinguished from them. (Redrawn from Figure 1 pp. 931 *in* Stewart-Oaten et al. 1986).

abundance that would be expected if random factors were removed (abundance seen in Figure 2.1B). However, the data collected can provide information regarding the combined random and systemic variation only and is unable to isolate the effects of the discharge (i.e. behaviour of the systemic factors which include the effects of the effluent) from the other naturally occurring factors that would generate regular variation in species abundance.

Their solution was to sample at a number of points in time at both sites simultaneously, before and after the discharge began. This would not only provide an estimate of the variation due to random and systemic factors at a single point in time but would demonstrate how the mean abundance fluctuated over a period of time. If we assume that random deviations about the systemic mean evened out over time (long run average deviation close to zero) and that the systemic factors responsible for the regular variation do not to change (i.e. no changes in amplitude, frequency, or long term mean abundance), then estimates about all sources of variation can be made and the influence of the point source discharge can be ascertained. The authors have discussed a numbers of problems with the design and mentioned some solutions, as well as suggesting that, in some instances, this model may not be appropriate. Further authors have discussed advances with this design (Stewart-Oaten et al. 1992), or criticisms and alternatives (Underwood 1992, 1994). In addition, there are examples available that illustrate how this method can and is being used by scientists and environmental manager [examples include (Guidetti 2001, Rybczyk et al. 2002, Bro et al. 2004)].

What this discussion and the work initiated and pursued by Green, Hurlbert and Stewart-Oaten and colleagues illustrates that a number of key factors must be considered when designing environmental monitoring and assessment protocols. Green's purpose was to provide common sense and easily accessible solutions to often complex environmental problems: a necessity for effective and efficient environmental protection. Hurlbert illustrates the need for clarity, understanding, and thoughtfulness in experimental design, whether used in basic or applied research, so that ecosystem processes can be clearly understood, the correct mechanisms of environmental degradation can be ascertained, and remediation or prevention can be initiated. Stewart-Oaten and his colleagues were able to demonstrate that a change in perspective could provide solutions to 'insurmountable' problems in ecosystem studies. They stressed that the proper scale for impact assessment work was, in many cases, smaller than that which would be required for more traditional ecosystem level investigation (i.e. the site specific or local scale was appropriate and a more general inclusive approach was unnecessary or undesirable). They also stressed the necessity for a clear understanding of the parameters being estimated and how they should be used, as well as illustrating the importance of discriminating between natural and anthropogenic influences on ecosystem variables. Finally, they proposed an improved methodology that could be used to determine the influence of an anthropogenic disturbance and illustrated its value with data from an actual study [see power plant example pg 932 *in* (Stewart-Oaten et al. 1986)].

Unfortunately, the requirement that the affected site be sampled before the impact commences seriously limits it's utility for most environmental impact assessment work. More often than not, a point source discharge has been operating for a number of years before any attempt is made to determine the consequences of the discharge to the receiving waters. As well, the usual constraints of finding appropriate control sites coupled with the frequent occurrence of multiple point source and non-point source discharges and other influences (e.g. discontinuities such as confluences with major tributaries), particularly in the lower reaches of large rivers, further complicates any attempt to disentangle anthropogenic effects from natural variation. As well, logistical constraints with regard to the necessity to sample control and impact sites simultaneously, coupled with difficulties in meeting the statistical pre-requirements (i.e. assumptions regarding the natural variation in the target species) further eroded the utility of this methodology.

2.7 River Invertebrate Prediction and Classification System (RIVPACS)

Another technique used for site assessment is the River Invertebrate Prediction and Classification System (RIVPACS). The key architects of this technique included J.F. Wright, D. Moss, P.D. Armitage, and M.T. Furse and their goal was to classify river invertebrate communities by physical and chemical parameters and determine environmental status based on the macroinvertebrate communities found at a given site. The basic premise of this monitoring and assessment tool was that under specific physical (flow, substrate, slope etc.) and chemical (pH, Chloride, alkalinity etc.) conditions a given macroinvertebrate community could be expected. Conceptually, under a particular set of environmental conditions at pristine or un-impacted sites, a diversity of invertebrates will be tolerant of the ambient environmental conditions and should be able to survive and exploit the available resources so that stable populations of those organisms can be maintained over time. If the expected community was not found, usually indicated by a decrease in species diversity, then the site could be considered degraded (Furse et al. 1984, Wright et al. 1984).

The determination of river type and the expected invertebrate community involved an exhaustive data collection period that initially involved 268 sites throughout Britain in which 28 different parameters were measured (Wright et al. 1984). The sampling program has evolved over the years and now includes 614 reference sites in Great Britain (Wright et al. 1997) and has expanded to include sites in Northern Ireland (Wright et al. 2000). The data generated was analyzed using a number of multivariate statistical techniques. To begin, the invertebrates sampled were identified to species (where possible) and a BMWP score was calculated. A BMWP score is a classification system developed by the Biological Monitoring Working Party: a collaborative group who were charged with establishing a system that could be used to determine the biological condition of British rivers (National-Water-Council 1981). The resulting scoring system is based on samples of a river's invertebrate community. A score is assigned to each invertebrate taxon (at the family level) based on that group's tolerance to pollution. The BMWP score is determined by summing the individual scores of all families present at a given site. Alternately, this diversity index may be expressed as an ASPT score (Average Score per Taxon), which is calculated by dividing the BMWP score by the total number of scoring taxa (Furse et al. 1984).

The BMWP or ASPT score (i.e. the invertebrate community) from the good quality reference sites, along with the environmental data (physical and chemical parameters) collected at each of these locations, was then used to classify the reference sites into groups. This was done using a FORTRAN program called TWINSPAN (two-way indicator species analysis) (Hill 1979), which utilizes a type of statistical analysis called Detrended Correspondence Analysis (Hill and Gauch 1980). The data were then analyzed using Multiple Discriminate Analysis (MDA), a technique that exposes the physical parameters most responsible for characterizing the groupings identified by TWINSPAN. The environmental variables that distinguish a particular group of reference sites will be associated with a specific invertebrate assemblage found at those locations. Thus, a specific biological community will be expected to be present under the physical and chemical conditions identified. By applying the classification scheme identified by the multivariate analysis of the reference sites to potentially impacted locations the expected invertebrate assemblage can be predicted based on the observed environmental characteristic. If this assemblage is not found then that site may be considered degraded or polluted (Furse et al. 1984).

This technique has proven to be quite robust and has proven to be very successful at predicting the expected community structure and correctly classifying sites (Armitage et al. 1987). RIVPACS has been implemented as a monitoring tool in Britain by (EPA in England and Wales and by SEPA in Scotland) and similar multivariate approaches have been developed and implemented in other parts of the world. These include AusRivAS (Australian River Assessment Scheme) developed in Australia (Parsons and Norris 1996) and the Benthic Assessment of Sediment (BEAST) developed for use in the Great Lakes in North America (Reynoldson et al. 1995). These techniques have become popular because they employ measures of physical, chemical and biological parameters of an ecosystem, thus they provide information about all aspects of an ecosystem. As well, they are sensitive to a variety of types of pollutants, do not require physical reference sites within the vicinity of the impacted site, are predictive, and provide an endpoint that environmental managers can work towards.

Predictably, RIVPACS has not been universally accepted. The most frequently cited criticism is that this model is empirically derived by correlation, the causal mechanisms behind the relationships observed are not known and cannot be applied generally (Calow 1992, Cash 1995), hence, the need to develop a new model for each proposed application or differing geographical region. This creates serious limitations for the implementation of this model within the context of the European Community. If these techniques are to be applied universally, a new model will need to be developed for each ecologically distinct region, which, in some instances, may necessitate the development of multiple models within a single politically distinct jurisdiction. This may prove restrictive for some EU members if the technical expertise or financial means are not available.

Secondly, the requirements of the Water Framework Directive stipulate that the determination of environmental degradation is to be made from comparisons of conditions that can be regarded as 'high-status'. The use of the best available sites in Britain does not meet this requirement. At the turn of the 20th century, forest cover had been reduced to 1% of its original level. Even today, only 12% of Britain is forested (National-Statistics 2004), most of which is in the form of forestry schemes that are composed of homogonous plantations of non-native species (Oosthoek 2001) . In addition to the conversion of land from native forest to agriculture or urban areas, a long history of watercourse alteration, such as flow regulation and water extraction has altered the physical and chemical characteristics of many if not all watercourses within Britain (Petts, 1989). Because unaltered physical and chemical conditions probably do not exist anywhere in Britain, the use of correlative techniques such as the multivariate analysis employed by RIVPACS and the prediction of an expected invertebrate assemblage or any other biological component at a particular site under ‘high-status’ or anthropogenically unaltered conditions is not possible. Thus, the determination of ecosystem status, as defined by the European Union in the Water Framework Directive is not possible.

2.8 Bioenergetic Models

Another direction in habitat assessment involves the development and application of individual based bioenergetic models (Hardy 1998). Numerous researchers have suggested that a bioenergetics approach will better link habitat models and biological mechanisms (Guensch et al. 2001). Bioenergetic models predict individual microhabitat choice based on the energetic cost and benefits of using different habitats and have been shown to accurately predict microhabitat choice by a variety of drift feeding fish (Rosenfeld 2003). These models work by describing habitat use and fish development through the quantification of the balance between energy gains (i.e. feeding) and energy loss (i.e. swimming, digestion, food capture, growth, reproduction, urine and faeces). Bioenergetic models are comprised of a suite of metabolic equations that quantify functional relationships between physical variables such as water temperature, digestion, as well as metabolic, kinetic, and growth processes in fish based on energy as a common unit (Hayes et al. 2000). Such an energetic approach can be physically based by including the environmental factors that influence fish survival and growth (Booker et al. 2004).

Bioenergetic models have been successful in part because the biomechanics of drift-foraging are relatively easily modelled, with swimming costs a simple function of velocity at the focal point and energy intake a function of size of the foraging window, reactive distance, and energy concentration of invertebrates in the drift (Rosenfeld 2003). Rosenfeld and Boss showed that a simple bioenergetic model for drift-feeding cutthroat trout generated average predictions of net energy intake that were consistent with observed patters of growth in both pool and riffle habitats, and Nislow et al (2000) demonstrated that bioenergetic models provided a reasonable prediction of Atlantic salmon growth rate potential in different habitats (Rosenfeld 2003). Some of the strengths of a bioenergetic approach are that models are based on prey variables that can be used to investigate the ultimate causal mechanisms controlling growth and habitat selection (Hayes et al. 2000); these models are inherently linked to growth and biomass production, they are theoretically more transferable among systems and are connected intrinsically to the food web dynamics of the system. In addition, continued advancements in computing power and the potential for spatial data collection will make such spatially explicit modelling efforts more widely feasible (Guensch et al. 2001).

The main limitations of bioenergetic models relate to uncertainty in the efficiency with which fish harvest drifting invertebrates, and how foraging behaviour and stream hydraulics influence swimming costs. Further there are some limitations to bioenergetic models in relation to uncertainty surrounding the time and effort necessary, in terms of sampling and evaluation of drift samples, to generate reasonable estimates of prey abundance. Any requirements for significant temporal and spatial replication could limit application for general management purposes. Certainly, bioenergetic modelling has the potential to provide insights into the mechanisms underlying patterns of habitat use and production by stream fishes, but whether it proves useful for management or remains primarily a research tool remains to be seen (Rosenfeld 2003).

2.9 Biotic Indices

RIVPACS incorporates a technique of data management that utilizes indexing, a procedure which enables an investigator or environmental manager to consolidate a great deal of ecological information into a condensed and readily digestible form. These techniques developed in order to determine the status or condition of an ecosystem based on the response of an assemblage or community of organisms to various forms of pollution. There have been a variety of biological indexing systems developed and applied to aquatic systems in various places around the world at various points in time. The earliest of these was developed in Europe by Kolwitz and Marsson [1908, 1909 as cited Karr (1991)] at the turn of the twentieth century and incorporated the concept of saprobity, or degree of pollution. This index was developed for rivers based on resident organisms' sensitivity to organic pollution (sewage), which would often manifest as decreases in dissolved oxygen levels (Cairns and Pratt 1993). This methodology incorporates the sensitivities of lotic organisms ranging from bacteria to fish and is still being used in Europe [Biologically Effective Organic Loading Plan in Germany and the Quality Index in the Netherlands (Metcalfe-Smith 1994)].

In North America, diversity indices were more frequently employed. This style of index combined measures of species diversity (i.e. number of species at a site), evenness (degree of uniformity in the distribution of individuals among species), and abundance (the total number of organisms present at a site) as a measure of ecosystems condition. It's thought that undisturbed ecosystems will show a higher species diversity and abundance (greater species richness) and more even distribution of species when compared with degraded sites. A variety of species diversity indices have been developed in the last three to four decades but the most commonly utilized is the Shannon Diversity Index (Cash 1995).

The advantage of a readily understood parameter that could be used by managers to assess an ecosystems state was undermined by some practical and conceptual difficulties with these techniques. Criticisms of Saprobic indices include difficulties with taxonomic resolution, the necessity for intensive sampling regimes, non-transferable saprobic values and species lists, non-quantitative assessments of species tolerances, and the inability to make statements regarding community level responses because of the species specific nature of the technique. Diversity indices were thought to be more useful because they did not rely on subjectively determined pollution tolerances, the diversity values calculated were independent of sample size and thus, demanded less intensive sampling regimes, and the values calculated were believed (wrongly) to be readily amenable to statistical analysis. However, their shortcomings included the inability to provide species level information, there tendency to be misused by managers, and the existence other statistical techniques that incorporate more biological information in a more ecologically relevant form (Green 1979, Cash 1995). Further, Hurlbert suggest that the meaning of 'diversity' is rather elusive and offers alternative species-composition-parameters which might be more meaningful (Hurlbert 1980).

RIVPACS makes use of a third class of index that also relies on measures of species diversity. However, with biotic indices, the organisms of interest are a subset of all taxon available, of which, the pollution tolerances of included species, genera, or family’s are known. As previously mentioned the BMWP scores generated for

RIVPACS, are based on scores assigned to invertebrates based on their pollution tolerances. Typically, organisms expected to be found in a given ecosystem type are ranked based on their tolerances to pollution and assigned numbers from high to low (intolerant to tolerant). A site will be surveyed and a census of the organisms present will be conducted. The predetermined tolerance-based numbers will then be assigned to the organisms or groups of organisms present and summed. This final value or numerical summary is then compared to an index that ranks ecosystem condition, often based on arbitrarily defined categories, from high (pristine) to low (lifeless). Other examples of biotic indices include the Trent Biotic Index and Belgian Biotic Index (Metcalfe 1989, Resh and Jackson 1993).

This type of index focuses on a subset of the available taxa, and requires diversity measures not abundance. As well, samples need only be taken one to three times per year and, if invertebrates are used, can be completed quickly and easily with simple equipment. Therefore, the requisite sampling regimes are simpler, more cost effective and less time consuming then saprobic or diversity indices, thus, more attractive to environmental managers. However, these methods have been criticized on the grounds they do not consider habitat differences in their assessments (i.e. they do not include measures of environmental variables). Of course, this does not apply to the use of BMWP scores when used in the context of RIVPACS. Furthermore, pollution tolerances are often determined subjectively rather than being based on experimentally determined quantitative measures (Metcalfe-Smith 1994). Moreover, the scores generated from these techniques, like those produced from other types of indices, do not lend themselves well to statistical analysis (Norris and Georges 1993).

A further category of this class of environmental assessment is the Indices of Biotic Integrity (IBI). There are two types of IBI, those that examine fish communities (Karr 1991, Dionne and Karr 1992) and those that focus on macroinvertebrates (Benthic Index of Biotic Integrity – B-IBI) (Kerans and Karr 1994). Fish were used for a number of reasons. Like other aquatic organisms they are sensitive to a wide range of environmental perturbations, unlike invertebrates or diatoms a great deal of information is know about the life history of most fish. Fish communities are often include representative organisms from many trophic levels and feeding groups (omnivores, herbivores, insectivores, planktivores, piscivores) and include food of both aquatic and terrestrial origin. Fish are easy to identify, can be used to examine acute and chronic stress, are generally present in most freshwaters, and have appeal to the general public (Karr 1981). The score that is developed is based on up to 13 different fish community parameters (or matrices). These include measures of taxanomic richness, proportion of certain selected taxa and the proportion of habitat and/or trophic specific guilds of species, genera or families of sensitive species, tolerant species, individual condition, and abundance (basic parameters used by Karr are outlined in Table 2.3). These parameters are initially measured at control or reference sites and then a comparison is made with similar measures at a test or impacted site. This is done by assigning a (-), (0), or (+) to each metric which are given the values 1, 3 and 5 respectively. These values for all matrices are then summed and compared to an index of 9 classes that ranges from ≤ 23 to 60 (very poor to excellent).

The advantage to IBI or the B-IBI is that they are sensitive to a range of different types of perturbations as well as to cumulative effects. They provide ecologically relevant information in terms of direct measures of resource condition (i.e. status of fish or

Table 2.3: Parameters used in assessment of fish communities (Redrawn from Table 2 pp. 22 in Karr 1981).

Species Composition and Richness	
	Number of species
	Presence of intolerant species
	Species richness and composition of Darters
	Species richness and composition of Suckers
	Species richness and composition of Sunfish (except Green Sunfish)
	Proportion of Green Sunfish
	Proportion of hybrid individuals
Ecological Factors	
	Number of individuals in sample
	Proportion of omnivores in sample
	Proportion of Insectivores in sample
	Proportion of top carnivores
	Proportion with disease, tumours, fin damage, and other anomalies

invertebrate communities), and involve relatively easy calculations. There are number of disadvantages. In some instances not all matrices respond to stress in a predictable way (Hoefs and Boyle 1992). The resulting score does not relate directly to any observable phenomenon, or to any theoretical or empirical synthesis (Steedman and Regier 1990). Because the index created is adapted and calibrated to local conditions comparison with sites from different locations is not possible.(Regier 1992) This methodology has no predictive properties and the final cumulative score can often mask important information seen in the scores generated by the component measures (e.g. uniformly mid range scores will give the same final score as will several high scores combined with several low scores). Despite these limitations the use of indices has been adopted in many jurisdiction largely in North America (Steedman 1988, Oberdorff and Hughes 1992, Whittier and Rankin 1992).

2.10 Hydraulic Models

2.10.1 One-dimensional hydraulic models (PHABSIM):

The assessment of physical habitat availability in rivers is often used as a method for determining the impacts of management, such as flow regulation or river restoration on communities and species of interest (Booker et al. 2004). Fish and other aquatic species have been shown to prefer specific ranges of physical habitat variables within their environment. Hydraulic models have been designed to predict changes in physical variables within a river segment with changes in flow rate. Combined, the habitat preferences of resident species and hydraulic models have been used to predict changes in habitat quality within a reach with changes in flow rate (Guay et al. 2000). The most widely publicized methodology employing this approach is the instream flow incremental methodology (IFIM) and in particular one of its major components, the physical habitat simulation system (PHABSIM) (Bovee 1982, Leclerc et al. 1995, Hardy 1998).

PHABSIM was developed under the guidance of the US Fish and Wildlife Service. It is a computer-based model that utilizes field measurements of channel shape and physical habitat features as well as knowledge of target species habitat requirements. The model produces simulations of the quality and quantity of habitat available to a target species based on existing conditions and potential habitat resulting from proposed water developments (Maddock 1999). PHABSIM employs strategically placed transects which are used to describe the longitudinal distribution of different habitat types within the streams. Measurements of physical microhabitat parameters, such as depth, velocity, substrate type, and cover are made at intervals along each transect to describe the lateral distributions and gradation of these parameters. The point on each transect where a measurement is made is called a vertical (the measurement is perpendicular to the plane defined by the water surface). Each vertical marks the edge of a stream 'cell', [sometimes called a tile (Guay et al. 2000)] the length of which is established by the investigator in the field. Each stream cell is unique and characterized by a surface area (defined by distances between transects and verticals); a substrate type, a cover type and an average depth and velocity. The depth and velocity within a cell is a function of the streamflow (Bovee 1982).

These physical variables are used as inputs to a hydraulic model that predicts water depth and current speed in any given cell for a specified flow rate. The anticipated quality of a cell as a fish habitat is defined by an index that integrates the preference of fish for the substrate diameter, the water depth, and the current speed in that tile. This results in a map describing the habitat quality index assigned to each tile at a given flow rate. Change in flow rate will change the wetted area of the reach and modifying the number of tiles modelled. As well, the habitat quality index value of each tile will change as the water depth and current speed is altered by changes in flow. Changes in the number of tiles and their habitat quality index then allow for the assessment of the impact of flow rate modification on fish habitat quality and quantity of a river (Guay et al. 2000).

The habitat suitability index (HSI) is the most commonly used index of habitat quality. This index is based on preference curves that represents the degree of preference displayed by the target species over the complete range of current speed, water depth, and substrate diameter found in a river or reach. Preference for a specified range of current speed, water depth, or substrate diameter can be calculated as the ratio of percent utilisation (percent of fish observed that used this range of variable) to percent availability (percentage of the surface area of the river characterised by this range of variables) of these environmental conditions. Preference indices range from 0 (poor habitat) to 1 (best habitat). Integration of the surface area of all tiles weighted by their HSI provides the weighted usable area (WUA) (expressed as a percentage of the total surface area or as square meters of habitat per 1000 m of river) for a river or reach at a given flow (Leclerc et al. 1995, Guay et al. 2000).

Preference curves used as the basis of the HSI are produced variously from observational studies of fish habitat utilization, literature surveys, and expert opinion (Waddle 2001, Heggenes et al. 2002). The data used in the construction of preference curves is often expensive or difficult to obtain so once developed, a curve may be applied broadly. In fact, universally transferable habitat criteria is a desirable goal for users of this method (Bovee 1982). The types of preference curves constructed for use by PHABSIM have been broadly categorized into four broad classes. These include the following:

Category I: Expert opinion or literature curves. These are typically derived from a consensus of experts' accumulated knowledge of habitat use by a species life stage or by evaluating habitat use by information found in the professional literature. Information derived from the literature includes general statements about fish habitat and/or may include variable amounts of field data. Category one curves can be the result of a combination of sources. An individual curve may include information from literature only or literature information combined with field data. This data may be smoothed or modified using professional judgement. These curves are intended to reflect general habitat suitability throughout the entire geographic range of the target species but can be constructed for more specific regions (Armour et al. 1984, Stier and Crance 1985, Waddle 2001).

Category II: Habitat Utilization Curves. These are curves that are based on frequency analysis of field data, which, in some instances, is smoothed or fitted using mathematical techniques. The data employed is collected over a broad range of flows and reflect the conditions that were being experienced by the fish at the time of sampling. These curves do not necessarily describe the conditions that are preferred by the target species, as the full range of habitat may not be available in the stream being studied. Utilization curves are ideally constructed from streams with high habitat diversity and are generally more transferable than curves from streams with low habitat diversity (Armour et al. 1984, Stier and Crance 1985, Waddle 2001).

Category III: Habitat preference curves. These curves are termed preference curves because they attempt to correct for habitat availability bias by factoring out the influence of limited habitat choice. Both habitat utilization data and habitat availability data is collected for these curves. The utilization and availability data is collected simultaneously and should reflect the relative amount of different habitat types in the same proportions in which they exist in the study area. This approach is an attempt to increase the transferability of the curves to stream (or conditions) that differs from those where the curves were originally developed (Armour et al. 1984, Stier and Crance 1985, Waddle 2001).

Category IV curves: Conditional preference curves. These curves describe the habitat requirements of target species as a function of the interaction among variables. For example, fish depth utilization may depend on the presence or abundance of cover, or velocity utilization may depend on the presence or absence of cover. (Armour et al. 1984, Stier and Crance 1985)

Alternately the curves can be distinguished by the formats or functions they describe. The hydraulic component of habitat simulation models like PHABSIM uses hydraulic models to determine the relative amounts of the different habitat conditions in the channel at a particular discharge. Within each cell or tile in a hydraulic model like PHABSIM there will be a discrete combination of depth, velocity, substrate and cover. Any specific combination of these habitat features occurs at only one discharge and in order to evaluate the utility of a cell it is necessary to approximate a function, which quantifies the species preference or tolerance of that combination. Bovee (1982) describes this as a combined or joint preference function and describes four methods to approximate this function within the PHABSIM methodology. These are; binary criteria, preference curves, multivariate suitability functions, and multivariate functions in association with preference curves (Bovee 1982). A basic summary of these four methods is outlined in Table 2.4.

The PHABSIM model has several limitations with respect to both the physical and biological models (Leclerc et al. 1995). Transferability of preference criteria is a goal

Table 2.4: Outline of methods used to calculate joint preference functions used to describe a species preference or tolerance of the combination of physical habitat parameters (depth, velocity, substrate, cover) generated by PHABSIM for each stream cell. Summarized from (Bovee 1982)

Joint Preference Function	Description	Advantages	Disadvantages
Binary Criteria	Suitability of habitat defined within boundaries for each species. If a habitat parameter (HP) in the sample area is within boundary then suitable (HP =1): if the parameter does not meet criteria, unsuitable (HP = 0). HP for depth, velocity, substrate, and cover multiplied together; if product = 0, area unsuitable.	Does not imply selective behaviour of fish within conditions specified (no statistical rules). Can be developed where no data on fish are available. Professional judgement can be applied.	Target species often observed using narrow bands of conditions but can often tolerate a broader range (i.e. no distinction between optimal, sub-optimal, and tolerable conditions).
Preference curves	Habitat suitability described as a curve; peak optimal (1), tails unsuitable (0). Tails represent unsuitability not a cut-off. Determined empirically from frequency histograms (field data), literature sources, professional judgement.	Can be constructed in the absence of hard data. Professional judgement can be incorporated. Complex mathematical functions used with relative ease.	Represent relative probabilities (ratios of probabilities), multiplication of preference factors implies independence among the variables. If developed from fish capture data, bias introduced by the physical condition available at the time the data were collected.
Multivariate suitability functions	P[N/E]: N is the probability of finding one or more fish and E is a given set of environmental conditions. P[E/F]: Probability of observing a combination of stream attributes given the presence of fish.	Environmentally independent; if defined properly then transferable to other systems. Good measure of usability when function is integrated with environmental conditions of stream. Data are collected where fish are found. Easier than P[N/E]: less bias due to interference with fish (higher quality). Smaller portion (relative to P[N/E]) of population can be sampled.	Requires intensive field sampling of all representative reaches of stream. Implies that the entire population has been sampled. Limited utility outside area from which it was derived. Does not distinguish tolerances from preferences. Area sampled must be in relative proportion to their occurrence in stream.

Table 2.4: Continued

Joint Preference Function	Description	Advantages	Disadvantages
Multivariate suitability functions (continued)	S = P[E/F]/P[E]: S is the joint suitability function describing the suitability and P[E/F] is the probability of finding a certain combination of environmental conditions given the presence of fish. P[E] is a probability function describing the relative abundance of various combinations of the environmental attributes available.	Essentially environmentally independent. S is biomass independent as the total biomass of the stream from which S is developed and does not enter subsequent calculations. Inclusion of interactions among variables and the removal of bias caused by physical habitat availability. Mathematical fitting of data rather than subjective curve construction.	The stream from which the data is generated must be at carrying capacity and may be dependent on the presence of sympatric species S provides a relative suitability of environmental condition. Limited utility as indexes of this type should have a maximum value of 1 at optimal habitat. Substantial data requirements can be limiting in some instances. Difficult to interject professional judgement and the complex mathematical functions are difficult to simulate in the model (particularly for cover and substrate).
Multivariate functions in association with preference curves	JPF = f(v,d) x f(s) or JPF = f(v,d) x f(c) JPF = joint preference factor f(v,d) = joint suitability function for depth and velocity f(s) = preference curve for substrate f(c) = preference curve for cover	Fitting data to a joint suitability function requires the function to be continuous and described by an exponential polynomial equation. Substrate and cover may not meet this requirement. Combined use of joint suitability functions and preference curves allows simple continuous variables (depth and velocity) to be described as joint suitability function and complex variables (substrate and cover) as preference curves.	Interactions between depth and velocity accounted for in a combined computation but independence between these two variables and substrate and cover are assumed.

of the model however the utility of the preference curves outside of the reach or region they were developed has been questioned (Shirvell 1986, Gore and Nestler 1988, Hayes and Jowett 1994, Greenberg et al. 1996, Maki-Petays et al. 1997). Most attempts to validate this approach have been conducted using comparisons between WUA and fish density or standing crop. While some studies confirmed the existence of a relationship between WUA and fish density, others found no such relationship (Guay et al. 2000, Stewart et al. 2005). The potential biomass (carrying capacity) of species in a community is only partially set by habitat availability; factors including forage abundance, predation and competition can reduce a particular species biomass below what would be expected based on habitat availability alone. Thus, changes in the amount and quality of habitat accessible to target species may not necessarily result in corresponding changes in biomass. As well, these models tend to focus on select species or specific life stages, thus, have limited scope (Stewart et al. 2005). Further, hydraulic models like PHABSIM are based on tile sizes with areas as large as $10m^2$, thus, the scale of the habitat descriptors used may not have relevance to fish which commonly occupy spaces smaller than $1m^2$. Finally, one-dimensional models calculated downstream changes in velocity and water-surface elevation only, therefore, they have limited utility in systems with significant lateral flow (Stewart et al. 2005). The specifics of the strengths and weaknesses of one-dimensional models can be found elsewhere (Hardy 1998, Maddock 1999, Guay et al. 2000, Rosenfeld 2003) fortunately, some of the limitations of these models may be overcome by utilizing more recently developed two- and three-dimensional models (Leclerc et al. 1995).

2.10.2 Two- and three-dimensional hydraulic models:

Like one-dimensional models, two-dimensional hydraulic models are comprised of two parts: a hydraulic model and a biological model. Hydraulic models use the topography of the stream channel in combination with hydraulic parameters to calculate the depth and velocity that would occur at a set of points in the stream channel for a given discharge. Unlike one-dimensional models, two-dimensional models incorporate a longitudinal axis (upstream-downstream) and a transverse axis (left bank-right bank). These axes are used to define x-y spatial locations (two-dimensions) in which an average current speed and depth are predicted for any specified discharge. These flow predictions are used in conjunction with the topography of the streambed and surveys of the substrate grain size. The model can then be used to describe the habitat available, in terms of depth, current velocity, and substrate at various flow rates (Guay et al. 2000, Panfil and Jacobson 2005).

The biological component of the methodology utilized a habitat classification scheme that outlines the physical habitat requirements of the resident species of interest. Habitat requirements can be defined by stream surveys of target organisms in which the physical conditions in a series of specified locations (e.g. 1 m^2 quadrats) are recorded along with occupancy (i.e. presence or absence of organism at specific locations). This allows an investigator or manager to quantify the variation in the propensity of the resident organism to prefer specific conditions amongst the range of physical characteristics found in the reach. The biological model assumes that when an organism more intensely uses ranges of substrate size, current speed, or water depth, that this range represents a habitat of higher quality. This information is then used to develop an index of habitat quality based on the affiliation or preference the target organism demonstrates for the substrate, current velocity and depth found at that location (Guay et al. 2000).

The results from the hydraulic model at a given flow rate and the biological component of the methodology are then combined. Current speed, water depth, and substrate composition predicted by the hydrodynamic model for a quadrat (tile, cell) are used as inputs for the biological model that assigns an index of habitat quality to that quadrat. Estimation of an index of habitat quality for all tiles modelled produces a map of the spatial heterogeneity of expected fish habitat quality in a river for a given flow rate. This exercise can be repeated to produce predictions of habitat quality and distribution of those habitats for different flow rates (Guay et al. 2000).

The advantage of two-dimensional models over one-dimensional models such as PHABSIM is that the downstream and lateral flow calculations allow for the quantification and reproduction of flow complexity which is more representative of that naturally found in streams (Stewart et al. 2005). Further; two-dimensional models do not necessarily require the measurement of velocities throughout the entire reach of interest; a feature that can be utilized in situations where an accurate description of the channel geometry at an appropriate spatial resolution is obtainable (Hardy 1998). This is a clear advantage over one-dimensional hydraulic simulations where extensive velocity collections and calibrations are typically required during modelling. However, two-dimensional models maintain some of the limitations of modelling systems such as PHABSIM as they determine vertically averaged flow velocities and assume static bed geometry. This means that these models do not depict vertical velocity gradients and are more appropriate for modelling discharges below that required for incipient stream bed motion (i.e. changes in streambed morphology with time) (Panfil and Jacobson 2005).

Three-dimensional hydraulic models have an advantage over two-dimensional models as they do predict vertical velocity gradients. However, like two-dimensional models they do not predict substrate dynamics (Hardy 1998). The ability to make accurate forecasts of the changes in both the sediment size distribution and resulting changes in meso-scale habitat characteristics has important implications in assessing instream flow and habitat restoration efforts. Further, two-and three-dimensional models recognize that it is not just the suitability of a particular location that should be considered; the range of habitat types or physical conditions that are adjacent to this point is also important. Although three-dimensional models have more potential utility for predicting physical habitat suitability they may be restricted in use as these models have high computational demands and require very powerful computers such as super-computers or high end work stations. However desktop personal computers are increasing in computational power and may quickly be able to meet the demands of these models (Hardy 1998). Spatially explicit flow models (two- and three-dimensional) are necessary to describe the spatial and temporal heterogeneity in a river system; not only to model the physical features of the habitat; but also to permit a better understanding of the processes that can be limited to fish existence (Stewart et al. 2005).

2.11 The lotic-invertebrate index for flow evaluation (LIFE)

The Lotic-invertebrate Index for Flow Evaluation (LIFE) method is based on the observation that many freshwater benthic invertebrates (BI) have precise requirements for particular current velocities or flow ranges (Hynes 1970, Statzner et al. 1988) and that alterations in BI community structure may occur as a direct consequence of varying flow patterns, or indirectly through associated habitat change (Petts and Maddock 1994). This methodology is based on the calculation of an index value that links the various taxon of British BI's with a specific flow regime (Extence et al. 1999).

More specifically the method involved the calculation of flow scores (*fs*), which are based on a matrix constructed from pre-defined flow group associations and abundance categories. Commonly observed British freshwater BI species have been allocated into one of six flow groups [i.e. taxa associated with rapid flows (I) to taxa associated with drying or drought impacted sites (VI)]. The abundance of the species sampled are divided into categories ranging from counts of 1-9 (group A) to >10,000 (group E). From this matrix, *fs*-scores are calculated for each species observed, summed and divided by the total number of taxa observed resulting in a LIFE index score. In general higher flows yield higher LIFE scores. These associations can also be made at the family level of taxonomic resolution although working at the family level may result in a loss of precision (Extence et al. 1999).

The associations between flow and the invertebrate community are explored using a computer program that elucidates relationships between the flow parameters that are best correlated with community structure (as measured by LIFE scores). The flow parameters calculated are numerous and include parameters such as percentile flow, mean flow, maximum flow, and minimum flow. These parameters can be calculated over varying periods of time, called 'flow-duration' (e.g. 90, 120, 150 days, or a full year), and in different season or 'flow periods' (e.g. April-September, March-October, etc). The strength of the relationship between the index score and any individual flow parameters varies with site and flow parameter utilized and the practitioner can exploit the strongest relationship for management purposes (Extence et al. 1999).

This methodology was tested on a number of rivers in England and the results indicate that LIFE is robust (i.e. works at various levels of resolution) and is very effective in encapsulating ecological response to changing flow patterns in a range of river types. The data collection step is simple as the methodology uses sampling techniques which are well established and routinely conducted. The LIFE method can be used to summarize the multiple effects of flow on invertebrate populations, much as biotic indices have historically been used to integrate water quality effects. This positive response occurs despite the fact that the flow data used in the LIFE method may not necessarily be the flows to which benthic macroinvertebrates are normally exposed because of the complex interactions that exist between river hydraulics, habitat morphology and habitat composition (Extence et al. 1999).

The preliminary work also illustrates that the baseline index values are inextricable linked with the geographical location of the biological sampling site. Index scores show a progressive downstream decline as current velocities diminish and associated habitat features change and values enumerated at individual sites will be further influences by the quantity and quality of instream habitat available for invertebrate colonization. However, the computer model used creates a large surplus of usable statistics, those flow variables showing the best relationships with the invertebrate fauna are proposed as being of primary importance in determining community structure in particular river systems. Under certain condition, particularly drought, the relationship between the index value and flow breaks down and LIFE scores become independent of flows. This observation can guide a manager towards ecological thresholds in flow below which significant ecological damage will occur and can be used as a guide for minimum flow recommendations (Extence et al. 1999).

2.12 Summary

The Water Frame Directive outlines the basic principles that must be used to ensure the protection of aquatic systems. The WFD ensures that member states restore all surface waters to a level defined as good status (European-Union 2000). Good status incorporates the idea that the biological elements of an ecosystem can show slight distortion resulting from human activity and deviate only slightly from those normally associated with the surface water body type. The incorporation of the human element into management protocols speaks to advocates of an environmental management concept known as the ecosystem approach (Karr 1991, Haskell et al. 1992). This protocol also advocates other concepts that have found there way into the WFD such as a holistic perspective and the maintenance of an ecosystem at a desired state. Good-status can be identified in a number of way: the use of physical sites; reference conditions establish using predictive or hindcast models, a combination of physical sites and modelling techniques, or expert judgement. In order to help define ecosystem status, it is helpful to possess information about the biotic and abiotic requirements of resident organisms so that the degree of departure from these requirements can be assessed. The complete range of habitat requirements of an organism can be defined through observational and experimental studies and can be thought of as a species fundamental (Hutchinson's) niche; however, the habitat it actually selects or prefers under natural conditions is thought of as the realized niche (Rosenfeld 2003).

Site to site comparison are difficult for many reasons including the lack of High-Status sites for comparison. This necessitates the need for alternate processes such as the upstream downstream comparison outlined by Green (1979): a methodology that was further developed into the BACI protocol (Stewart-Oaten et al. 1986). These upstream-downstream techniques tend to focus on changes in community structure whereas the development of bioenergetic models focuses on being able to predict microhabitat selection based on the energetic cost and benefits of using different habitats (Hardy 1998). The modelling approach has also been used for the assessment of ecosystem quality at the community level. RIVPACS was developed for use in Britain, and incorporates invertebrate community structure and environmental variables to determine if a site has been degraded. It does this by comparing the observed invertebrate community against an expected invertebrate community determined by the model and does not need direct comparisons with another location (Furse et al. 1984, Wright et al. 1984). Biotic indices also employ communities and assemblages of organisms to determine environmental quality. These methods incorporate the sensitivity of organisms, ranging from bacteria to fish, and often employ measures such as species diversity, evenness, and abundance. Indices are useful because they employ biotic responses to the environment and can condense a great deal of information into a readily accessible form (Metcalfe-Smith 1994, Cash 1995). Hydraulic models, such as the PHABSIM approach, combine the habitat preference of resident species with hydraulic models. This method has been designed to predict changes in habitat quality within a reach with changes in flow rate (Bovee 1982, Hardy 1998). Finally, In Britain, the LIFE index is being developed. This method is based on the calculation of an index value that links various taxon of benthic invertebrates with specific flow regimes. The method has great potential as it marries a measured ecological response in a predictable way to a dynamics aspect of an ecosystem (Extence et al. 1999).

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3.0 Methods

3.1 Conceptual Overview of the Approach

With respect to the WFD, there is a need to establish a base line or a set of reference conditions from which comparisons can be made so that the state of the ecosystem can be determined. Some monitoring protocols previously developed and reviewed in the last chapter rely on comparisons of the structural components of ecosystems. For example, RIVPACS (Wright et al. 1984) compares biotic indicators such as macroinvertebrate communities with an expected assemblage. Unfortunately, it is often difficult to determine if any perceived differences in these structural components are a result of the anthropogenic impact suspected in the area or natural variation. An enhancement may be to investigate the requirements that resident organisms have of their environment and determine if a location can support the organisms that they are expected to be able to sustain.

Organisms have evolved over time and have developed a variety of mechanisms that allow them to survive and contend with the physical, chemical, and biological components of their environment (Roff 1992). Their inability to do this or changes in their environment will reduce their ability to survive or reproduce. For example, brown trout (*Salmo trutta*) have evolved to live in cool or cold aquatic environments. The range of water temperatures that will support growth in this species is 4 to 19.5^{o}C. The lower limit for survival is 0^{o}C and the upper limit is between 25 and 30^{o}C (Elliott 1994). Temperatures in warmer equatorial waters will not be able to support brown trout nor will areas with elevated temperatures downstream of outfalls emitting thermal pollution.

Through the species tolerance to the physical, chemical and/or biological conditions we can determine what is appropriate for the species survival and reproduction.

This is the basis for the monitoring protocol that will be developed in the remained of this thesis. The maximum, minimum, and preferred range of physical habitat parameters (water depth, velocity and streambed substrate) that can be sustained by brown trout will be researched and described. This range of tolerances will be the reference conditions for current velocity for this species. By comparing what is present in an ecosystem with what is tolerated by the species we can determine the suitability of the habitat. This thesis is a preliminary investigation designed to illustrate how this approach may function. The target species used in this study is brown trout (S*almo trutta*), and I'll focus on three of the four principle physical habitat parameters that are important to salmonid abundance and distribution. The three parameters of interest will be water depth, current velocity, and substrate. As there is a great deal of literature available for the lotic life stages of brown trout, I'll concentrate on the stream dwelling portion of the trout life cycle. The method will be developed for the four life stages of this species: spawning, nursery, juvenile, and adult residents (Section 3.3).

The first step in the development of the assessment approach involves identifying the tolerance a species has for environmental conditions. These will be defined through a survey of the literature on physical habitat use. It is important to survey as much literature as possible in order to capture the full range of habitat that can be used by the target species: or at least the full range of physical habitat that the species has been observed using. The methods for defining the preferred habitat are detailed in the sections that follow (Section 3.4) but in brief involve summarizing measures of central tendency, standard deviation, as well as habitat suitability that is quantified using habitat preference indices.

These ranges of suitability will be called tolerance profiles. The tolerance profiles are then compared with survey data (measures of depth, water velocity, etc) taken from a stream of interest or concern. In this instance data was collected in a grid system (quadrats) allowing for the reconstruction of habitat suitability and comparisons with use. Within the study reach, areas that are suitable, useable, and not-suitable can be identified for each of the quadrats within the grid. This information will be summarized using histograms and tables. The grid-by-grid classification can also be used to create suitability maps which detail areas that are suitable, useable, or unsuitable for the target species. From these graphs the location of suitable habitat can be identified, and the proportion of total useable habitat and the grade (suitable or useable) can be determined. By completing this analysis for all life stages we can determine the suitability of stream reaches for each life stage. As this will be done for individual habitat parameters the investigator can identify potential problem areas, for example, a shortage of suitable spawning gravels or limited velocity refugia for fry.

3.2 The Study Site: The Carron Valley

The Carron Valley (Figure 3.2) is situated in the Campsie Hills (Ordnance Survey Grid Reference NS 68/78), has a catchment area of 38.7 km^2 and is utilized predominantly for hill farming and commercial forestry. The valley is dominated by a reservoir. The water surface area covers 4.1 km^2 with a mean depth of 8.3 m and a maximum depth of approximately 12 m. The Carron Valley was dammed in 1939 to meet the demands of the petrochemical and manufacturing industries at Grangemouth; and now supplies drinking water to Grangemouth, Falkirk, Kilintilloch, Cumbernaud, Lennoxtown and Milton of Campsie. The primary use of the reservoir is as a public water supply, provides 125 Ml/day of drinking water daily. It has also been used for the last 50 years as a brown trout fishery (Deverill 2000).

The reservoir catchment is made up of a complex of streams originating in the Campsie Fells on its southern shore. A total of 17 streams discharge into the reservoir ranging in stream order from first to fourth. A total of 11 streams discharge into the southern shoreline and of these the March Burn, Burnhouse Burn and the Carron River itself are the most significant, particularly in terms of spawning populations of trout. The catchments drained by March and Burnhouse Burns are of similar size (2.2 km^2 and 3.0 km^2 respectively), while the River Carron catchment is considerably larger (9.8 km^2). Much of the Carron Valley is being managed as a forestry plantation and is under the control of Forestry Enterprise. The total area encompassed by forest is approximately 26.0 km^2 of which the dominant species is Sitka spruce (*Picea sitchensis*). Additional species include Scots pine (*Pinus sylvestris*), European larch (*Larix deciduas*) and Douglas fir (*Pseudotsuga menziesii*). Buffer strips of mixed broadleaves have been planted adjacent to streams running through the plantations. Approximately 8.0 km^2 have been clear-felled in the upper March Burn catchment (1999/2000). At the date of this study no logging had occurred in the Burnhouse Burn catchment (Taylor 2000). Bin Burn was undergoing active logging in the upstream portion of the catchment that was ongoing during the study. This logging was primarily focused on clearing Meikle Bin [lower left hill in Figure 3.1 Ordnance Survey (1981)].

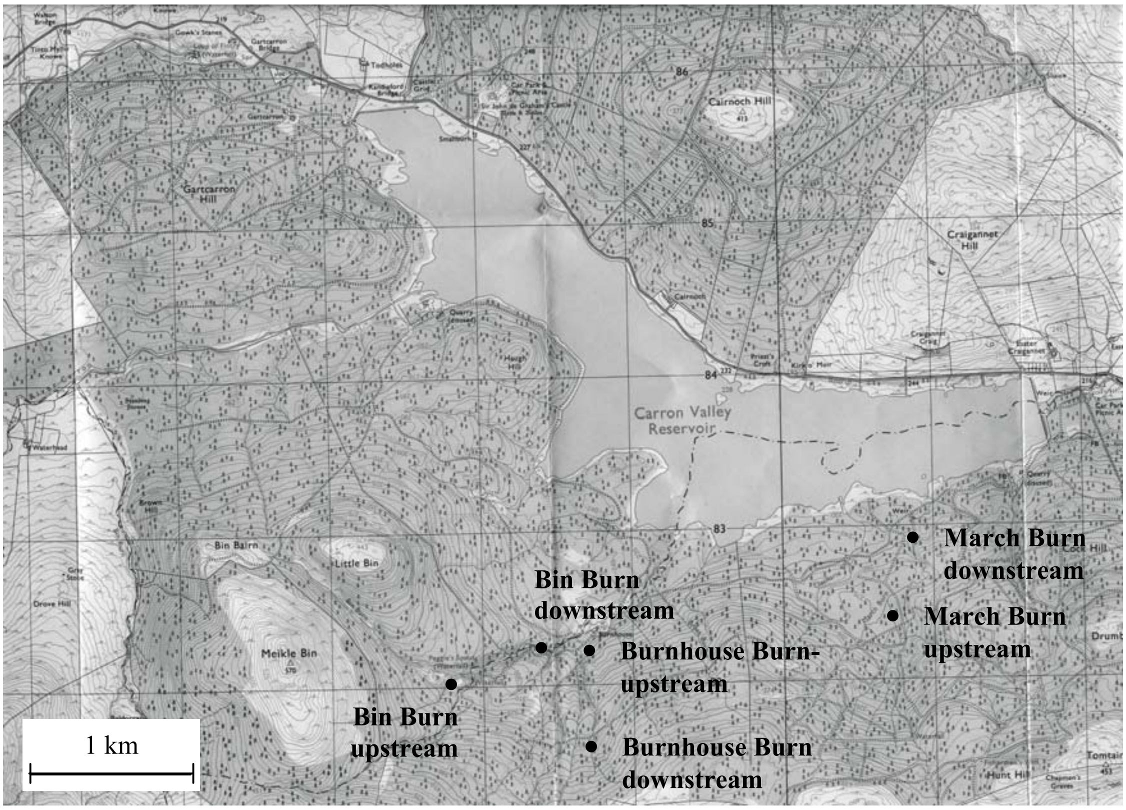

Figure 3.1: **Location of the sample sites from which water depth, water velocity, and streambed substrate were taken during the late summer of 2002. Ordnance Survey.**

Study reaches were chosen in March Burn, Burnhouse Burn and Bin Burn to test the applicability of the monitoring and assessment protocol being developed. The streams investigated are of relatively similar size and are in relative close proximity to each other and the research facilities at the University of Stirling. The discharge and surface area of the study sites are outlined in Table 4.8; Section 4.3.2.1, Chapter 4. Two sites were chosen on each stream, a smaller reach towards the upstream and a larger reach towards the downstream portion of each reach. Spatial and hierarchical analysis is important because it can to provide a broader understanding of how processes and conditions at one spatial scale control and affect the processes and conditions at a smaller scale (Frissell et al. 1986, Imhof et al. 1996). The locations of the study reaches are outlined in Table 3.1**.**

Brown trout, the target species being examined (Section 3.3) reside in the Carron Reservoir and are known to use the smaller streams running into the reservoir as spawning habitat (Taylor 2000); thus, the full spectrum of fish age classes and appropriate habitat are likely to be present. March Burn has an old gauging station downstream of the study sites, which seemed to have restricts access to this stream for spawning adults (personal observation). This site was chosen despite this limitation as the criteria used for assessment were based on literature observations not direct field measures. Further the presence or absence of fish in this reach could provide insight into the gauging stations true effectiveness as a barrier to migration (i.e. impact on the stream).

While the habitat survey was underway an unusually intense rainstorm occurred triggering a mudslide upstream of the logging operations in the Bin Burn watershed. A flood of debris, which included recently cut logs and other materials washed downstream and formed a dam immediately downstream of the study area. The subsequent high waters, dam removal, stream rehabilitation, and watercourse alteration that followed produced physical habitat conditions that appeared different to what had existed prior to the event. This event provided an opportunity to investigate changes in the physical habitat parameters before and after the event. Therefore, a post-spate survey was conducted.

Table 3.1: Location and sample date of study sites in the Carron Valley.

Site	Location*	Date Sampled
March Burn Upstream	NS70650/82393	22 07 2002
March Burn Downstream	NS70879/82972	15 to 18 07 2002
Burnhouse Burn Upstream	NS68507/81552	27 08 2002
Burnhouse Burn Downstream	NS68668/82160	28 08 to 02 09 2002
Bin Burn Upstream	NS67817/81898	14 to 15 08 2002
Bin Burn Downstream Pre-spate	NS68499/82253	24 to 26 07 2002
Bin Burn Downstream Post-spate	NS68499/82253	05 to 12 08 2002

* - Ordnance Survey National Grid Reference System

3.3 Brown Trout: Overview

3.3.1 Target species: brown trout (*Salmo trutta*)

Brown Trout (*Salmo trutta*) was chosen to illustrate how this monitoring protocol may function. There are a number of reasons for this choice. First, brown trout is ubiquitous. The species has a worldwide distribution but its native range is pan-European (Elliott 1994). Another important characteristic of *S. trutta* is that this species is a top-level level predator (Varley 1967, Imhof et al. 1996). Choosing a species near to the top of the food chain with a complicated life-cycle (Section 3.3.4) is important because it will have broad demands from the ecosystem in terms of using a variety of habitat types at different stages of its life cycle, broadening the scope of the assessment (Elliott 1994, Klemetsen et al. 2003). Brown trout along with Atlantic salmon (*Salmo salar*) and Arctic charr (*Salvelinus alpinus*) are among the best-studied fish species in the world (Klemetsen et al. 2003). Included in these studies are recent books (Elliott

1994) and literature reviews (Haury et al. 1999, Armstrong et al. 2003, Klemetsen et al. 2003). This wealth of knowledge has the clear advantages to less well studies species in that much of the habitat requirements of the target species have been observed and recorded: therefore, easier to define. Clearly, basing an assessment protocol on the habitat requirements of a less well studies species would delay the development and implementation of the protocol as the habitat requirements of the target species would need to be clarified.

3.3.2 Phylogeny

The salmonid subfamily *Salmoninae* comprises about 30 species of fish in seven genera of which *Salmo, Salvelinus*, and *Oncorhyncus* are the best studied (Klemetsen et al. 2003). The genus *Salmo* contains two species: brown trout (*Salmo trutta*) and the Atlantic salmon (*Salmo salar*). Linnaeus originally classified brown trout into river trout (*S. trutta*), stream trout (*S. eriox*), and sea-trout (*S. Fario*) and at one time ten species of trout were recognized in the British Isles. However, today most workers recognize only one polytypic species, *S. trutta*. I will refer to all life cycle types of *Salmo trutta* as brown trout or trout for the remainder of this thesis.

3.3.3 Distribution

The brown trout was originally a European species (Figure 3.2) native to the Western Palaearctic ecozone (Schultz 1995). Its northern range limits are Iceland, northern Scandinavia and Russia. Western limits are simply defined by the European coastline and the southern limits by the northern coastline of the Mediterranean Sea as well as the islands of Corsica, Sardinia and Sicily, and the Atlas mountains of North Africa. The eastern limits are more difficult to define but are probably the Ural Mountains and

Figure 3.2: **The native distribution of brown trout (Salmo trutta). Redrawn after Elliott, 1994.**

Caspian Sea (Klemetsen et al. 2003). The southern range extends as far as the upper reaches or the Orontes River in Lebanon (Elliott 1994). Brown trout have been successfully introduced into at least 24 countries outside Europe. These include early introductions (1852-1889) into eastern Russia, Tasmania, New Zealand, USA, Sri Lanka, Canada, Australia and Kashmir. Since then they have also be introduced to countries in Africa and South America (Elliott 1994).

3.3.4 Life history

Brown trout is typically a stream spawning species. Spawners return to natal streams with a high degree of accuracy and they generally move upstream or from lake or reservoirs into tributaries in the autumn or early winter (Raleigh et al. 1986). The earliest spawning tends to occur in areas with the highest latitude and altitude because of lower temperatures and longer egg incubation periods (Klemetsen et al. 2003). Brown trout females dig their nests in the bottom substratum (Haury et al. 1999) and larger females will often spawn on coarser gravel and bury their eggs deeper than small ones (Klemetsen et al. 2003). Each female deposits her eggs in a series of nests, which when aggregated form a redd (Armstrong et al. 2003). Several competing males often court one female, but one large male usually fertilizes the majority of the eggs. However, smaller subordinate males, often called sneakers or precocious males (Elliott 1994), also contribute to the fertilization of the eggs (Garcia-Vazquez et al. 2001). A female will spawn actively over a few days and does not defend her redd after the spawning period is finished. (Klemetsen et al. 2003).

The eggs incubate within the gravel from one to several months [the length of time being influenced by temperature (Armstrong et al. 2003) and hatch in the subsequent spring (Klemetsen et al. 2003). There is a terminology that is frequently used to describe trout as they age. When eggs hatch the young fish is called an alevin. Alvein have a yolk sac attached and feed entirely on yolk and live within the nest gravels. Fry is a transition stage when the trout emerge from the gravel, start to feed and disperse. Parr are older trout with the yolk sac fully absorbed and are feeding entirely independently. Smoltification is the process fish undergo prior to seaward migration: these fish are referred to as smolts. The final stage is mature males and females. The population is often separated into year-classes and each year-class is named after the year in which eggs hatch, not the year in which they are laid. The standard convention often followed is 0+ for trout less than 1 year old, 1+ for trout between 1 and 2 years old, and 2+ trout between 2 and 3 years old (Elliott 1994).

Alevins (20 mm in total length) swim up or emerge from the gravel when most of the yolk is consumed (Klemetsen et al. 2003). The dispersal of the young fry happens almost immediately after emergence and have established feeding territories within a week (Raleigh et al. 1986). The young are aggressive, defend territories, form dominance hierarchies and compete intensively for resources (Klemetsen et al. 2003). Individuals that are unable to procure a territory may drift downstream and most of these will probably die (Elliott 1994). Fry tend to spend their first summer in the nursery or natal stream. (Elliott 1994). As trout age they may move away from the nursery areas during autumn and winter or they may continue to grow near where they were spawned; however, they use different habitat as they grow (Armstrong et al. 2003). During their first year in the river, the young dwell largely in shallow areas, often located along the riverbank with fast or moderately fast flowing water. As they grow older and larger, they prefer deeper, more slowly flowing parts of the stream (Klemetsen et al. 2003).

As trout are an anadromous polytypic species there are a number of life-cycle types observed. The first and simplest lifecycle is where the trout spends its entire life in its natal stream. In this scenario sexual maturity is usually attained at the age of two or three years. Iteroparity is frequent with adults spawning at least two or three times before death. The second version is similar except that 1+ and 2+ parr migrate from their natal streams to the parent river and adults do not return until just before spawning. The third version is where trout will migrate to a lake or a reservoir. The trout will emigrate from streams into the lake at the age of 1+ or more some waiting until their second, third of fourth years to emigrate. The spawning populations consist of iteroparous males and females returning from the lake and males that mature in the streams without emigrating. Finally, the fourth type of lifecycle includes trout that migrate to an estuary (estuarine or slob trout) or the sea (sea-trout). This is also the most variable of the life cycles. The age at which parr change to smolts and emigrate varies considerably [see Jonsson (2001)]. Mean smolt age increases with latitude as smolt age is related to fish size and as growth is related to temperature and the length of the growing season. Their time at sea is also variable. Individual sea trout may return to fresh water more than once in their lifetime but spawning does not occur on each migration (Elliott 1994).

3.4 Definition of Habitat Parameters and Field Methods

Four physical habitat variables have been more consistently studied than any other. These include water depth, water velocity, streambed substrate and cover and are generally considered the most important factors affecting habitat use by trout and salmon (Heggenes 1988c, 1990, Gibson 1993). The relative importance placed on these four habitat components is inconsistent. In a range of studies workers have suggested conflictingly that depth (Bohlin 1977, Egglishaw and Shackley 1982, Kennedy and Strange 1982, 1986, Heggenes 1988a, b), water velocity (Shirvell and Dungey 1983, Bachman 1984, Gatz et al. 1987), cover (Baldes and Vincent 1969, Mortensen 1977, Fausch and White 1981b), substrate (Karlstrom 1977, Gatz et al. 1987) or a combination of these variables (Karlstrom 1977, Bagliniere and Champigneulle 1982, Gatz et al. 1987) as being the most important habitat feature to influence the distribution and abundance of salmonids.

Cover is an important physical habitat characteristic for stream salmonids (Fausch and White 1981a, Cunjak and Power 1986). However it will not be developed as a component of physical habitat in this study. There are a number of reasons for this omission. The definition of cover is complex and can be difficult to assess quantitatively (Heggenes 1988c). The complexity is a result of the variety of form it can take including instream elements such as water depth, surface turbulence, loose substrate, large rocks, and other submerged obstructions, undercut banks, debris logged in the channel, and external features such as overhanging vegetation (Binns and Eiserman 1979). As well, depth velocity and substrate are the most pertinent physical parameters used in the development of hydraulic models (Heggenes 1996). For simplicity, the measures taken in the field were limited to water depth, velocity and substrate. This chapter will outline the methods used in the development and application of tolerance profiles for water depth, water velocity and substrate composition.

3.4.1 Literature summary and tolerance profiles

A survey of the literature concerning brown trout habitat preferences was completed. More specifically a search was conducted focusing on investigations that examined the choice of microhabitat used by brown trout: in particularly those studies concerning the requirements for water depth, water velocity and substrate in lotic systems. This data was then summarized, plotted and compressed visually to produce a tolerance profile that expresses the range of depth brown trout has been observed using (usable depth) and a smaller banding of water depth that this species seems to prefer or finds necessary to sustain a viable population (suitable depth).

Combinations of keyword were used to search three different databases that include the biological and ecological scientific literature. The keywords include *Salmo trutta* combined with depth, spawning, nursery, habitat, physical habitat, microhabitat, and habitat suitability. As well, literature was extracted from a number of reviews concerning salmonid life history and habitat preferences. These include summaries by Heggenes (1999), Haury et al. (1999), Armstrong et al. (2003), and Klemetsen et al (2003). Finally, key older papers were identified and a search was conducted to uncover recent papers that cited these works. Cited reference searches were conducted on papers by Lindroth (1955); Kennedy and Strange (1982), Shirvell and Dungey (1983), Bain et al. (1985), Raleigh (1986) and Heggenes (1988c).

A complete and comprehensive overview of *Salmo Trutta* habitat utilization was not the objective of this study nor is it warranted, considering the availability of the recent reviews mentioned above. The papers of interest in this study were those that reported specific details (i.e. data) about the physical parameters that brown trout were observed exploiting. Papers that reported the depth, velocity, and substrate which brown trout were observed were collected and summarized. The habitat parameters utilized by this species was reported in a number of different ways. Firstly, the parameter was report directly in a descriptive manner as a mean, a range, a standard deviation or standard error of the mean, or a combination of these parameters. Habitat utilization reported as direct measures will be referred to as 'observational data'. Secondly, the habitat observations can be reported as a histogram summarizing the number of fish observed in a number of categories (categorical data). Finally, habitat utilization was often reported as a preference histogram or curve. This methodology relates microhabitat that individuals are observed using (i.e. depth, current velocity, substrate size) to the availability of this habitat within a reach or river segment and produces categorical habitat preference curves [see Chapter 2, Section 2.10.1] (Bovee 1982, Raleigh et al. 1986). These techniques allow investigators to determine the habitat a population or age class of fish prefers in relation to what is available. Microhabitat summarized in this way will be referred to as 'preference data'.

Tolerance profiles were created for each of the four age classes delineated from the four main stages in the brown trout life cycle (see Section 3.3.4). Trout habitat preference seems to be influenced by size rather than age (Bohlin 1977, Kennedy and Strange 1982, Mäki-Petäys et al. 1997), thus length was use to classify the data available from the literature into these four categories, which include spawning, nursery (≤ 7 cm), juvenile (> 7 to 20 cm), and adult (stream residents > 20 cm) depth requirements. The observational, categorical and preference data used to generate each profile were reported in a figure illustrating how the tolerance profile was created through the summary, manual plotting and visual compression of this information. It was also summarized in a table. The graphics used to illustrate and distinguish between each data type are summarized in Figure 3.3. The preference ranges that are reported in these figures are those expressed by the author. If a preference range was not stated then an approximation of this range was estimated from the figure provided. When possible, variance estimates from observational data reported as standard error of the mean were converted to standard deviation of the mean using formulas reported by McGhee (1985). Factors that may influence microhabitat selection or our interpretation of the data were also recorded in the summary tables. These factors include study location, survey technique, fish size and the presence/absence and identity of co-occurring

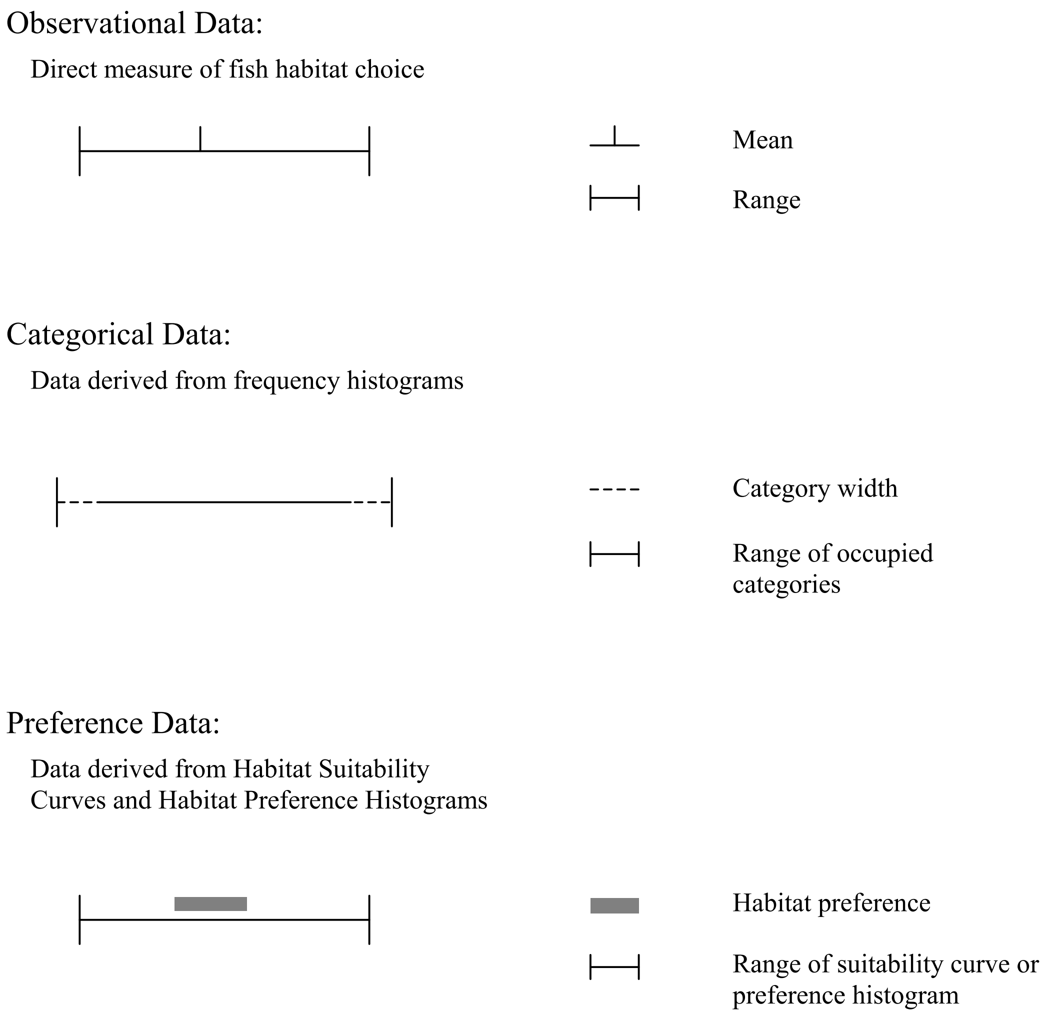

Figure 3.3: **Symbols used to identify the three data types employed in the construction of brown trout microhabitat tolerance profile.**

salmonid species. This does not include so called 'coarse' fish species unless they were examined for possible interactions with salmonids.

The objective in the construction of these tolerance profiles was to be as inclusive as possible; therefore, examples of the stream physical habitat parameters used by brown trout were taken from as many different locations in the worldwide distribution as could be obtained. The specific habitat used or favoured by trout in lotic environments is influenced by a number of factors including the habitat available, as well as inter and intra-specific competition. Also, the survey technique used which include electro-fishing, observations from the riverbank, and underwater surveys can influence the resulting data and the interpretation of habitat use (Heggenes 1988c). Where possible study location, survey technique, fish size, and the presence or absence and identity of co-occurring salmonid species (allopatric or sympatric population) was listed.

The tolerance profiles developed used both the data available and knowledge about *Salmo trutta* life history parameters obtained from the scientific literature, thus, the specific criteria used to create the profile for each age class was different. However, general principles were used in all examples. To begin, categorical data and variance estimates (standard deviation and standard error of the mean) do not give a specific endpoint to the range of habitat observed for each age class. Therefore, the minimum and maximum limits to the habitat tolerance profiles or 'usable habitat' was created using the reported ranges from the observational data. In some instances the range of the tolerance profile was adjusted to accommodate all members of the age class or when specific limits are not clearly expressed. Secondly, observational data does not express a preference for specific habitat in relation to the entire habitat available. As well, an individual mean for any given habitat parameter is limited in that it is an expression of central tendency for the conditions at the time of the study (i.e. a 'snapshot' in space and time). Consequently, the habitat preference reported from suitability studies and the range of means from observational and categorical data (if reported) was used to determine the 'suitable' range in the tolerance profile.

3.4.1.1 Water velocity

Water velocity is generally reported as either mean current velocity (0.6 depth) or snout or focal point velocity (i.e. the water velocity experienced by the fish). Many biologically oriented investigations tend to include snout or focal point water velocities as they are more directly related to what fish actually senses (Baldes and Vincent 1969, Shirvell and Dungey 1983, DeGraaf and Bain 1986). Workers studying microhabitat position for management position often use mean water column velocity (0.6 depth) as this measure is used in existing hydraulic models (Bovee 1982, Heggenes 2002). Thus, there is disagreement regarding the position in the water column most appropriate for recording water velocity in microhabitat studies. Some investigators have found a poor correlation between mean water column velocity and fish position (Heggenes 2002) while others have found the opposite relationship (Shirvell and Dungey 1983). Brown trout require heterogeneous hydraulic conditions (Fausch 1984, Hughes and Dill 1990) and velocity measured at focal point may be too narrow in focus (Hayes and Jowett 1994). Heggeness and Saltveit (1989) comment that areas with suitable snout velocity are rarely if ever limiting in trout streams; therefore, a more general description of water velocities may be more informative for management positions. In this study, tolerance profiles were created for both focal point and mean water velocities. Much of the literature regarding microhabitat choice focuses on substrate measure and I would be remiss not to include it. As well, the summary of this data furthers our understanding of trout microhabitat usage and can help inform choice of methodologies in future studies.

3.4.1.2 Substrate

A survey of the literature concerning brown trout (*Salmo trutta*) microhabitat preferences was completed focusing on studies concerning the requirements for streambed substrate in lotic systems as completed for the previous parameters. This data was then summarized and compressed to produce a tolerance profile that expresses the range of substrate brown trout has been observed using (usable substrate) and a second banding of streambed substrate that this species seems to prefer or finds necessary to sustain a viable population (suitable substrate). There was no substrate size that was considered not-useable by resident trout. The graphics used to illustrate and distinguish between each substrate data type are summarized in Figure 3.4. Substrate used for spawning was classified slightly differently as substrate requirements are much more rigid. In this instance, substrate that could be used for spawning and successful rearing of young is classified as 'suitable' and all other substrates are considered 'unsuitable'. This was done because the summary of the literature revealed that spawning substrate

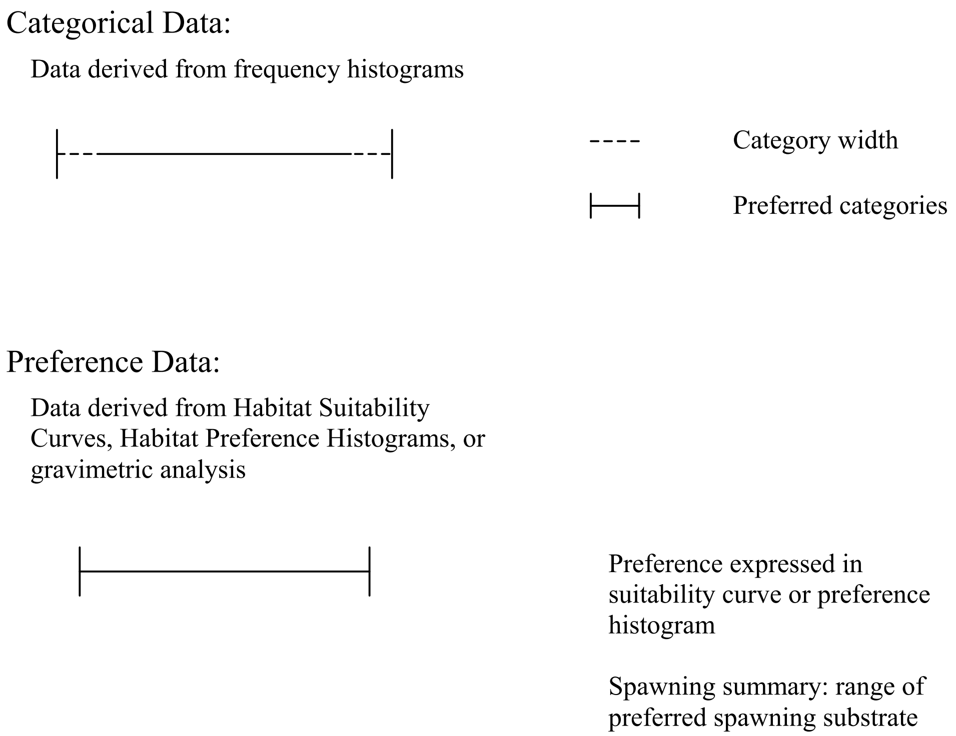

Figure 3.4: **Symbols used to identify the substrate data types employed in the construction of brown trout microhabitat tolerance profiles.**

was consistently within a narrow band (Figure 6.2) indicating very specific requirements for spawning substrate. Further, the literature reviews of spawning habitat indicate that failure to use appropriate spawning substrate often results in the demise of the eggs (Section 6.1.1).

Substrate preference ranges were estimated differently depending on the sampling methodology used in the papers reviewed. The literature survey revealed that a number of different assessment techniques were used to determine the streambed substrate chosen by brown trout. Broadly speaking, the substratum used by resident non-spawning fish (fry, juvenile or adults) was measured using visual classification techniques in which the observed substrate is assigned to a category based on particle size. These classifications can be documented as either a proportion of the observed substrate (e.g. 50% cobble, 30% pebble, 20% gravel) or more simply by recording only the dominant substrate within an area or along a transect (Bain et al. 1985, Heggenes 1988c). Spawning habitat assessment was generally completed using more quantitative techniques. These methods involve sampling a spawning site using a mechanical or freeze-core apparatus (Ottaway et al. 1981). The materials sampled are dried, sieved into fractions and then weighed. The substrate composition is then characterized into any number of statistics such as median grain size or geometric mean grain size (Crisp and Carling 1989). Instream quantitative measures of microhabitat usage or substrate particle size are rare.

For spawning substrates, streambed particle size estimates based on extracted materials from redds are reported as a single measure of central tendency (grain size statistic). Multiple measures within a given area will then allow an investigator to generate an estimate of variance around these grain-size-statistics. When available, I considered the 'preferred' substrate used by spawning trout to be within one standard deviation of the measure of central tendency. This was done as a way to define the variance of substrate used if no other value was reported. Workers have also quantified variance as a proportional subset of the total sample. This subset represents a majority of the material observed and is often expressed in terms of mass. For example, Beard and Carline (1991) state that '87% by weight of redd substrate particles ranged from 4 to 64 mm in diameter,…'. In these instances this proportion was used as the preferred range.

Substrate assessment of resident trout from visual assessments is generated using predefined particle size categories. Habitat usage based on this type of data is then assessed in two ways. First, substrate usage data can be summarized in a traditional histogram that reports the number of fish observed using each of the particle size classes predetermined by the classification system. Alternately, habitat utilization was reported as a preference histogram or curve. This methodology relates microhabitat that is being used to that available and is presented in categorical habitat preference curves. The predominate categories observed in both the traditional histogram and the preference histograms are considered the 'preferred habitat' in this study. Finally, the classification system used to assign substrate categories differs between studies. All data summarized was standardized by converting the size class reported by the author to a numerical value based on the scale used in that study. The numerical value of the particle size of interest was then reclassified according to the Wentworth scale. It is important to note that these scales are logarithmic and that the size (i.e. difference between largest and smallest particles within a category) of each category is not equal

Substratum type	Size class (mm)	(cm)
Organic fine materials		
Organic coarse materials		
Clay, silt	0.004-0.06	0.0004-0.006
Sand	0.07-2	0.0061-0.2
Fine gravel	2.1-8	0.21-0.8
Gravel	8.1-16	0.81-1.6
Small pebble	16.1-32	1.61-3.2
Pebble	32.1-64	3.21-6.4
Small cobble	64.1-128	6.41-12.8
Cobble	128.1-256	12.81-25.6
Large cobble	256.1-384	25.61-38.4
Boulder	384.1-512	38.41-51.2
Large boulder	> 512.1	> 51.21
Rough bedrock		
Smooth bedrock		

Figure 3.5: **Modified Wentworth scale after Bain et al. 1985 used for stream substrate particle size classification. Redrawn from Table II in Heggenes and Saltveit 1990.**

(Armstrong et al. 2003). For convenience, the figures that illustrate the construction of the substrate tolerance profiles (Section 6.2.1.) use equal sized particle size categories; however, the actual sizes vary getting increasingly larger from clay/silt to bedrock.

3.4.1.3 Tolerance profile construction

Once the information collected from the literature was summarized the data from each study used was plotted manually on graph paper (see Figure 4.1 as an example) using the symbols that indicate range, mean, mode, range, measures of variance, boundaries of histograms, and preferences outlined in the figures above (Figure 3.3 and 3.4). This information was then condensed manually into one line were all the broadest minimum and maximum value observed amongst all studies was used as the range and all measures of central tendency were plotted between these points. Expressions of preference were also clustered along this condensed line (see Figure 4.2 as an example). The measures of central tendency and expressions of preference would generally cluster within a narrow range between the minimum and maximum point. This narrower band was used as the suitable range of the physical habitat parameter. The points left of the suitable range to the minimum values and points right of the suitable range to the maximum value were considered useable. Any measure of a physical habitat parameter that was less than the minimum value or greater than the maximum value was considered not-useable.

3.4.2 Stream survey

The physical habitat surveys were conducted from the July 15 to September 2, 2002. A grid system was constructed within each study reach. A lateral guide-line was placed along the stream bank and a survey line (transect) was strung perpendicular to the guide line at points across the stream. The lateral line was numbered every 0.5 meters (0, 0.5, 1, 1.5 etc.) and the survey line was coded using letter characters every 0.5 meters (A, A.5, B, B.5 etc.) producing 0.25 m^2 quadrats with unique identifiers. The specific outline of each quadrat was identified using an aluminium frame, sub-divided into nine equally sized sections, that was moved from marked points every 0.5 meter along the transect line. Within each quadrat stream depth, water velocity, and substrate composition was recorded.

Depth was recorded using a meter stick placed at the centre of each quadrat and measures were recorded to the nearest centimetre. In dry quadrats, substrate measures were taken and recorded as dry for depth and velocity analysis to distinguish them from wet sections with no moving water. Water velocity was recorded using a Marsh-McBirney Inc. 'Flo-Mate' model 2000 portable flow meter. The probe on the flow-meter was adjusted so that velocity was recorded at 0.6 depth and placed at the centre of the quadrat. The flow-meter was set to display the average current velocity observed over 30 seconds. The instrument was allowed a stabilization period, then 3 separate '30-second averages' were recorded. For this project, focal point information was not available as observations of microhabitat choice were not conducted. Mean focal point velocities (0.6 depth), not snout or substrate water velocities, were recorded (see Section 3.4.1.1).

The visual assessment techniques generally employed for substrate analysis are not ideal as the valuation of the streambed is subjective (Bain et al. 1985, Heggenes 1988c). To achieve a greater degree of objectivity quantitative measures of particle size were recorded. Within each of the quadrats nine sections the central stone was measured on three axes. Particles less than 1 mm (clay, silt) was recorded as '<1'. Stones with dimensions larger than 25 cm, the practical limit for handling, were recorded as '>25'. If portions of the quadrat fell on the streams edge the subsections affected were recorded as streamside (SS). The mean of the three axes (length, width, height) in each of the nine quadrat sections was calculated and the average of these values was taken to represent the particle size of that quadrat. If the majority of the sub-section were <1, >25 or SS then the dominant category was used to classify the quadrat.

The edge of the stream at the beginning and end of each transect was recorded at the border between the bank-full width and the streamside. The specific stream edge could not be recorded due to the limited resolution of the technique employed. However, each of the subsections within the quadrat was recorded as falling in the streambed or on the streamside thereby increasing the sampling resolution. The survey began at the downstream limit of the study reach. Once a cross-sectional transect was completed the transect line was moved upstream to the next 0.5 meter marked point on the lateral guide-line. The transect line was 'squared' perpendicular to the guide-line and the survey resumed. This procedure was completed along the entire length of the study reach. The study reaches on March Burn and Burnhouse Burn were 22 and 24 meters long, respectively. The length of these survey reaches were designed to be approximately one stream meander in length (one stream wavelength), thereby, including at least one example of the riffle, run, and pool sequence commonly found in lotic systems. The length of the study reach was assessed using the methodologies outlined by Newbury and Gaboury (1993).

3.4.3 Stream habitat assessment

The habitat available for each age class of trout in the study reaches was assessed by combining the physical habitat parameters recorded during the habitat survey with the tolerance profiles constructed from the literature summary. The physical habitat parameters of each streambed quadrat was compared to the tolerance profile for each life stage of brown trout and coded as one of the following: usable, suitable, outside the tolerance range (not-useable), or dry. This procedure was completed for all streambed quadrats and was presented as histograms and in tabular form for the four age classifications was produced. A map (suitability map) of the study reach was constructed by sequentially plotting the streambed and/or streamside status of each quadrat for each transect. The mapping procedure was incredibly time consuming so was carried out for March and Burnhouse Burns only.

3.5 Fish Survey

The objective of the fish survey is to summarize the number, size and species of fish that are resident or make use of the study reaches. This is not intended as an intense population survey. Rather, the purpose of the survey is to help validate the model by demonstrating that the fish predicted or expected to be seen using the criteria of the model are actually present in the study reaches.

The study reaches were surveyed to quantify the numbers, size, and species of fish on four occasions from the autumn of 2002 to summer of 2003. All study-reaches were sample on the same day. The sample dates included September 25th and December 12th 2002 and March 18th and June 30th 2003. Two people sampled the streams, one operating a 300 V pulsed-DC backpack electro-shocker and the other using a length of seine net (approximately 1 meter long) held between two poles to catch the stunned fish. The entire length of both study reaches was surveyed over a 5-minute period. All fish captured were identified to species, counted, and measured for length (total length) to the nearest millimetre.

3.6 Statistical Analysis

Each quadrat was classified as suitable, useable, or not-suitable based on the criteria developed from the literature surveys of brown trout microhabitat use of water depth, current velocity and substrate. In chapters 4, 5 and 6 (depth, velocity, and substrate), the relative proportion of quadrats classified as suitable, useable, or not suitable was summarized and presented in tables and histograms. The relative proportion of these suitability criteria were compared statistically using chi-squared tests ($\alpha = 0.05$) between all seven data sets for each life stage. This was done for both the wetted and total streambed. Comparisons of the relative proportions of the suitability criteria were also made between the relative proportion of the suitability criteria at the downstream Bin Burn site before and after the spate using data from both the total and the wetted-only portions of the streambed.

In Chapter 7 and index was created that integrated the information for the three physical variables examined in the previous chapters (4,5 and 6: Section 7.2). These index values (resident trout: not-useable, satisfactory, good, high, and very high; spawning trout: not-useable, good, high, and very high) were tallied for each reach (seven datasets). The relative proportion of these integrated assessment values was compared between the seven data sets using chi-squared tests ($\alpha = 0.05$) for each life stage in both the total and wetted portion of the streambed. Chi-squared tests ($\alpha = 0.05$) were also used to compare the relative proportions of the suitability criteria between the relative proportion of the integrated assessment at the downstream Bin Burn site before and after the spate using data from both the total and the wetted-only portions of the streambed. Calculations were done using the statistical software SYSTAT© version 10 (SPSS Inc. 2000). Data analysis conducted in Chapter 7 is discussed therein.

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4.0 Depth

4.1 Introduction

This chapter will focus on the development and application of tolerance profiles for water depth while subsequent chapter will deal with water velocity and substrate composition (Chapters 5 and 6, respectively). The importance of water depth as an environmental variable to trout or any stream dwelling fish is obvious as this variable can be thought of as the height of water or the vertical space that is available to the resident organisms. Each chapter has two distinct objectives. These includes: 1) the development of size-class specific tolerance profiles for each of the habitat parameters being assessed and 2) the application of these profiles in order to assess the habitat availability at two sites on March Burn and Burnhouse Burn and to two sites on Bin Burn (plus a pre and post spate assessment - total of 7 separate assessments). These assessments will be conducted in order to evaluate the effectiveness of the technique and to demonstrate how this methodology could be employed to assess physical habitat available for brown trout in running freshwater ecosystems. As well, the results of the fish survey conducted at the six sites will be summarized. I will conclude the introduction with an overview of the requirements that brown trout have of the depth of water in a stream.

4.1.1 Instream depth requirements: overview

Brown trout spawn in shallow running water environments (i.e. rivers) although there are reports of lake spawning populations (Klemetsen 1967, Scott and Irving 2000, Sneider 2000, Brabrand et al. 2002). Spawning occurs in the autumn or early winter and hatch in the subsequent spring. During the first year in the river the young dwell largely in shallow areas, often located along the river margin (Lindroth 1955, Bohlin 1977, Kennedy and Strange 1982, Bardonnet and Heland 1994, Roussel and Bardonnet 1999, Roussel et al. 1999). Trout fry are often observed in areas without older trout (Bohlin 1977) and it is thought that the younger fish may prefer deeper habitat but are excluded from these locations through intraspecific interactive segregation by older and larger juvenile trout, which also prefer these areas (Jones 1975, Bohlin 1977). Individuals which are unable to catch food may drift downstream; many such individuals will die in a short period (Elliott 1994). If fish have access to a lake many will move there for feeding (Jonsson 1989). This movement may begin shortly after the young emerge from the gravel bottom, but the trout may also stay in the nursery area longer, and some may become stream residents (Jonsson 1985)

As they grow older and larger brown trout prefer deeper, more slowly flowing parts of the stream (Bohlin 1977, Egglishaw and Shackley 1982, Cunjak and Power 1986, Greenberg et al. 1996, Heggenes et al. 2002). Several authors have reported an increasing use of deeper habitats as the size of the fish increases from young of the year (3-8 cm) to adult fish (>20-25 cm) (Jenkins 1969, Bohlin 1977, Karlstrom 1977, Kennedy and Strange 1982, Bachman 1984, Cunjak and Power 1986). However habitat use is flexible, depending on variables such as habitat availability, (Elso and Greenberg 2001, Greenberg et al. 2001, Heggenes et al. 2002) time of the day (Heggenes et al. 1999), and season (Bjornn 1971, Cunjak and Power 1986, Mäki-Petäys et al. 1997). Brown trout occurring in rivers or lakes with free access to the sea often form anadromous populations. The migration to sea may be for the summer only, or the fish may stay at sea for two or more years before returning to their rivers of origin for spawning (Went 1962, Jonsson and Jonsson 2002). Andromous brown trout can be found in very small brooks. There, they survive because they can abandon the brook and move to deeper brackish water during periods of unsuitable low flow conditions such as periods of drought (Jonsson et al. 2001).

4.2 Results

4.2.1 Tolerance profiles

4.2.1.1 Spawning depth

A summary of the literature that reports brown trout spawning depth requirements is reported in Table 4.1. These observation were recorded from investigations in streams throughout the world including sites in France (Nihouarn 1983, Fragnoud 1987), Norway (Heggberget et al. 1988), Sweden (Rubin et al. 2004), Canada (Witzel and MacCrimmon 1983), the United States (Smith 1973, Beard and Carline 1991, Essington et al. 1998) and New Zealand (Shirvell and Dungey 1983, Scott and Irving 2000). They include studies of brown trout living in sympatry (Smith 1973, Heggberget et al. 1988, Grost et al. 1990, Beard and Carline 1991, Essington et al. 1998), allopatry (Rubin et al. 2004) as well as both sympatric and allopatric populations (Shirvell and Dungey 1983, Witzel and MacCrimmon 1983, Scott and Irving 2000). The bulk of these investigations occurred in natural streams; however, the investigation by Rubin and his colleagues (2004) and Schneider (2000) involved investigations in canals or the use of artificially created spawning habitat. To be as comprehensive as possible; when attempts to find the original documents failed; some of the values reported for spawning depth have been extracted from literature reviews by Raleigh and his colleagues (1986) and Haury and his colleagues (1999). These include studies by O'Donnell and Churchill

Table 4.1: Literature used to define the range of stream depths that can be used by brown trout (Salmo trutta) to spawn.

Source	Mean cm	Range (SE) or [SD] cm	Number of redds	Fish size (cm) mean (range)	Study location	Allo-sympatric populations	Other species present	Natural/ artificial stream	Location of measure	Notes
Berg 1977	-	28.3 to 60.3	-	-	-	-	-	-	-	Reported by Raleigh et al. 1986
Beard and Carline 1991 1987 1988	 28^{β} 27^{β}	 $27\text{-}30^{\varepsilon}$ $25\text{-}30^{\varepsilon}$	 90 113	20->30	USA (Pennsylvania)	Sympatric	Coarse fish	Natural	Deepest part of pit	β - Grand median; median from all samples from all sections ε - 95% confidence interval
Essington et al. 1988 Subsection 1 Subsection 2	 24 34	 [5] [10]	 48 60	-	USA (Minnisota)	Sympatric	Brook trout	Natural	-	-
Fragnoud 1987	23.9 cm	5.0 to 51.0	-	32	France (Eastern)	-	-	-	-	Mean depth of redd Reported by Haury et al. 1999
Grost et al. 1990	16	[5]	80	20-40	USA (Wyoming)	Sympatric	Brook trout	Natural	Mean over redds	Sampled trout >15 cm in stream to determine size
Heggberget et al 1988 large rivers small river	 43.1 50.0	 [17.9] [15.5]	 36 125	-	Norway	Sympatric	Atlantic Salmon	Natural	Centre of egg pocket	Large - Pooled data from Rivers Alta, Gaula, Driva. Small - Data from River Eira
Nihouarn 1983 width: 1.5-3 m width: 7-20 m	 - -	 $< 30^{\kappa}$ $30\text{-}60^{\lambda}$	-	-	France (Brittany)	-	-	-	-	76% (λ) and 98 %(κ) of values within this range. Reported by Haury et al. 1999
O'Donnell & Churchill 1943	45.7 cm	N/A	-	-	-	-	-	-	-	Reported by Raleigh et al. 1986
Reiser & Wesche 1977	-	6.4 to 18.3	-	-	-	-	-	-	-	Reported by Raleigh et al. 1986
Rubin et al. 2004	35.1	[15.1]	46	57.3 (18.5-89)	Sweden (Gotland)	Allopatric	-	Artificial	Redd and tail	Natural stream artificial spawning habitat
Scott & Irving 2000 Silver Stream Verter Burn Nardoo Stream Awakino River	 11 16 12 13	 [42.6](7) [7.5](2) [5.2](1) [2.4](1)	 37 14 27 6	-	New Zealand	Allopatric & Sympatric	Rainbow Trout	Natural	Point of least depth in water column	Standard Deviation calculated from sample size and standard error (McGhee 1985)
Shirvell & Dungey 1983	31.7 cm	6.0 to 82.0	140	42 (32-55)	New Zealand	Allopatric & Sympatric	Rainbow Trout	Natural	Snout of fish	-

Table 4.1: Continued

Source	Mean cm	Range (SE) or [SD] cm	Number of redds	Fish size (cm) mean (range)	Study location	Allo-sympatric populations	Other species present	Natural/ artificial stream	Location of measure	Notes
Smith 1973	42.6 cm	[54.5]	115	-	USA (Oregon)	Sympatric	Pacific Salmon	Natural	Upstream edge of redd	-
Waters 1976	-	12.2 to 91.4	-	-	-	-	-	-	-	Mode of optimal depth; range is for suitable depth Reported by Raleigh et al. 1986
Witzel & MacCrimmon 1983	25.5	7.0 to 58.0	110	(18-54.5)	Canada (Ontario)	Allopatric & Sympatric	Brook Trout	Natural	-	10 or more random positions over each redd

(1943), Waters (1976), Berg (1977), Reiser and Wesche (1977), Nihouarn (1983), and Fragnoud (1987). Limited information is available about the specific characteristics of these investigations.

Typically, brown trout spawn in the shallow areas of streams utilizing riffles or glide habitats (Bagliniere et al. 1979); often where there is an acceleration in the water current (Heggberget et al. 1988): however, see comments by (Ottaway et al. 1981). There does not seem to be any specific minimum requirements for spawning depths as trout spawn in the shallow areas of streams utilizing riffles or glide habitats (Bagliniere et al. 1979). Often in areas of swift current flow some salmonids have been observed spawning with their backs above the water. However, it is thought that they will need at least enough water to cover their bodies (Crisp 1993). The width, or dorsal height, of a brown trout is approximately 0.2 body lengths (Crisp and Carling 1989, Crisp 1993) and adult brown trout range in length from 40 to 60 cm (Scott and Crossman 1973). Thus, these salmonids will need approximately 8 to 12 cm of water to spawn. A minimum depth of 15 cm has been suggested by Reiser and Wesche (1977), although they report trout spawning in depths as low as 6.4 cm. Similarly, Shirvell and Dungey (1983) observed trout spawning in 6 cm of water while Witzel and MacCrimmon (1983) have measured spawning redds located in as little as 7 cm of water. Fragnoud (1987) report that spawning can occur at depths ranging from 5 to 51 cm. The lower end of this range, 5 cm, is the shallowest reported minimum depth for spawning brown trout.

The shallowest mean-spawning-depth (11.0 cm) was calculated from Scott and Irving's (2000) observations of Silver Stream in New Zealand. The other three streams investigated also had shallow mean-spawning-depths ranging from 12 to 16 cm. The deepest mean-spawning-depth (50 cm) was reported by Heggberget and his colleagues (1988) for small streams in Norway. Included within this range are mean spawning depths of 16 cm (Grost et al. 1990), 24 cm (Fragnoud 1987), 25.5 cm (Witzel and MacCrimmon 1983), 27 & 28 cm (Beard and Carline 1991), 24 & 34 cm (Essington et al. 1988), 31.7 cm (Shirvell and Dungey 1983), 35.1 cm (Rubin et al. 2004), 42.6 cm (Smith 1973), 43.1 cm (Heggberget et al. 1988), and 45.7 cm (O'Donnell and Churchill 1943).

The deepest reported spawning depth for brown trout was 91.4 cm observed by Water in 1976. However, there may be no theoretical maximum depth at which brown trout may spawn. The habitat utilized may be more limited by the availability of acceptable spawning gravel and water velocities in deeper waters (Raleigh et al. 1986). Studies of other salmonid species (rainbow trout; chinook, coho, and pink salmon) indicate that depth, with the exception of a minimum depth, does not significantly affect the selection of redd sites or the survival of embryos (Chambers 1956).

Based on the observations of the workers cited above and listed in Table 4.1 a profile, or tolerance range, of brown trout spawning depth can be constructed. It seems brown trout can use water as shallow as 5 cm deep (Fragnoud 1987) and have been reported spawning in water as much as 91 cm deep (Waters 1976). The extremes of the observed spawning range may not be ideal but they do represent depths that brown trout have been observed exploiting and thus, are capable of being used for spawning. These are the upper and lower limits and will be used to define 'usable' spawning depth. The measures of central tendency in spawning depth range from approximately 11 cm (Scott and Irving 2000) to 50 cm (Heggberget et al. 1988). These values represent a range of measures of central tendency that have been observed in a wide variety of locations and encompass many of the biotic and abiotic factors that influence spawning site selection. This series of midpoints may not be ideal in a site-specific context but they do represent depths that have been effectively utilized by *Salmo trutta* for spawning. Therefore, this range of midpoints will define the 'suitable' spawning depth of this species. The construction of suitable and usable spawning depth for brown trout, and the resulting tolerance range, is illustrated in Figure 4.1.

4.2.1.2 Nursery depth (fish length ≤ 7 cm)

Unlike spawning depth, studies that report microhabitat selection of fry come from a much narrower range of *Salmo trutta*'s worldwide distribution. Many of the investigations of natural populations came from Scandinavia including Norway, (Bremset and Berg 1999), Sweden (Lindroth 1955, Greenberg et al. 1996) and Finland (Mäki-Petäys et al. 1997). There was one study from England (Heggenes et al. 2002), one from France (Bardonnet and Heland 1994), and four from the United States (Raleigh et al. 1986, Harris et al. 1992, LaVoie and Hubert 1996, Pender and Kwak 2002). With the exception of the Finnish study (Maki-Petays et al. 1997), all of the trout populations coexisted with either Atlantic salmon (Lindroth 1955, Bremset and Berg 1999, and Heggenes et al. 2002) grayling (Greenberg et al. 1996) or brook trout (Harris et al. 1992, LaVoie and Hubert 1996). Raleigh and his colleagues (1986) did not report on co-existing populations and the trout in Pender and Kwak's (2002) study co-occurred with non-salmonid species that may have influenced the population characteristics of the trout. As well, all of these studies employed underwater survey techniques or electro-fishing methods in their study reaches with the exception of Harris and his colleagues (1992) who used surface observations to collect data. Finally, there

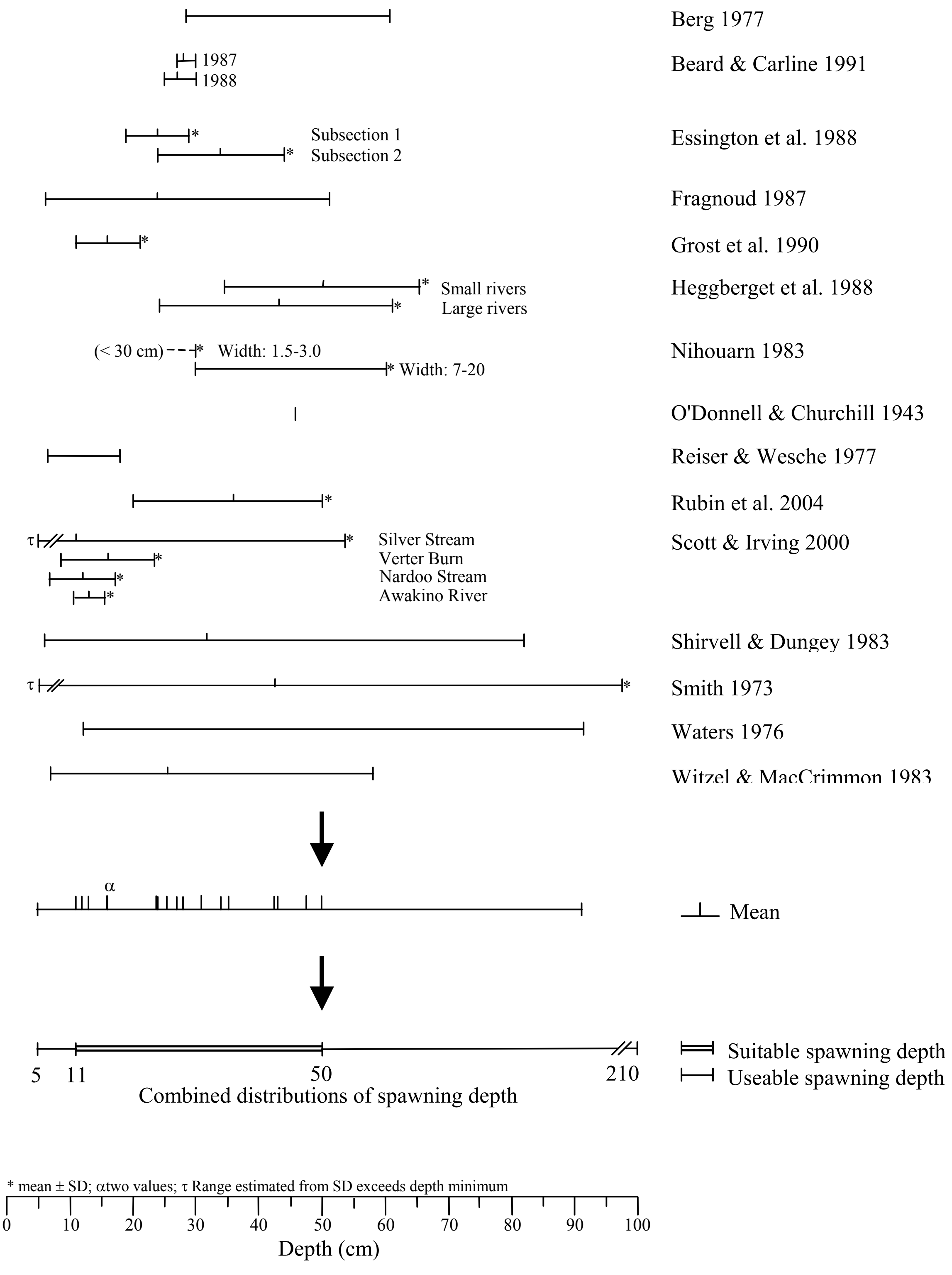


Figure 4.1: Procedure used to define the range of stream depths that can be used by brown trout (*Salmo trutta*) to spawn.

was one artificial stream study, conducted in France, which examined microhabitat choice of brown trout alevins in the presence and absence of potential predators [larger trout and sculpins (*Cottus gobio* L.)] (Bardonnet and Heland 1994).

Young trout (alevins) swim up from the gravel when most of the yolk is consumed, emerge from the gravel and disperse thorough-out the immediate environment. Brown trout fry are aggressive and quickly establish feeding territories and form dominance hierarchies (Kalleberg 1958, Mills 1971, Hèland et al. 1995, Lahti et al. 2001, Klemetsen et al. 2003). The depth of the environments that fry occupy, based on the microhabitat studies obtained in this study, are quite variable. However, the range of depths observed was generally not reported. When the variance was reported it was done so as standard deviation or standard error (Mäki-Petäys et al. 1997, Bremset and Berg 1999, Pender and Kwak 2002) and it is not possible to determine specific habitat ranges from these values. However, Heggenes and his colleagues (2002) did report a range; observing trout fry in water from 8 to 114 cm deep. Alternately, Lindroth (1955) and Greenberg and his colleagues (1996) present their data in histograms and Raleigh and his co-workers (1986) report habitat preference as a suitability-curve. The occupied depth intervals in the histograms include ranges of 5-15 to 80-90 cm (Lindroth 1955), and 0-15 to 75-90 cm (Greenberg et al. 1996). Raleigh reports a range of approximately 8 to 135 cm.

A number of studies did report mean depths of the nursery habitat that they examined. There were 13 values reported and came from studies in northern Europe including Norway (Bremset and Berg 1999), Sweden (Greenberg et al. 1996), Finland (Mäki-Petäys et al. 1997), England (Heggenes et al. 2002), and the United States (Pender and

Kwak 2002). With the exception of fish in the River Todalselva in Norway, the mean depth of nursery habitat used by brown trout ranged from 10 cm, in the River Vindøla, (Bremset and Berg 1999) to 35.6 in the River Vojmån in Sweden (Greenberg et al. 1996). The fry in the River Todalselva occupied a mean depth of 93 cm (Bremset and Berg 1999)

The way the data obtained has been reported makes it difficult to determine a specific minimum depth requirement. However the data we have suggest that minimally trout fry will need at least 8 cm of water. This value may be biased due to sampling limitations as the smallest members of this cohort who may be as small as 2 cm (Klemetsen et al. 2003), which may be difficult to identify in their natural setting, and based on their size could potentially use water less than 1 cm deep [based on the logic used by Carling and Crisp (Crisp and Carling 1989, Crisp 1993) for spawning depth]; although the larger members of the cohort (7 cm) would need approximately 2 cm of water. The deepest water that fry have been reported using range from 114 to 135 cm (Heggenes et al. 2002 and Raleigh et al 1986, respectively). However, like spawning habitat there may be no maximum theoretical limit for the depth of nursery habitat, thus, the depth of water that brown trout seem capable of using ranges from 2 to > 135 cm.

It would seem that young brown trout will use any water that they can physically access. However, beginning with the earliest habitat studies (Le Cren 1973) and continuing with more recent investigation (Bohlin 1977, Hermansen and Krog 1984, Wesche et al. 1987) a preference for shallow flowing areas often less than 20-30 cm has been suggested. This is corroborated by the observations of Lindroth (1955) and Mäki-Petäys et al. (1977) who propose preference depths of 20-30 cm and 5-35 cm respectively. Shallower depth preferences were observed by Harris and his colleagues (1992), 7 and 10 cm alternately in day and night observations: and Lavoie and Hubert (1996) who suggest that the fry in their study prefer depths of 2.5 to 7 cm. Interestingly, Bardonnet and Heland (1994), observing trout alevins in artificial streams, found they prefer depths of 10 or 20-30 cm in the presence or absence of potential predators, respectively. However in this review, Greenberg et al. (1996) as well as Raleigh et al. (1986) suggest deeper preferences of > 45 cm and 40-55 cm. As well, Bremset and Berg (1999) and others (DeGraaf and Bain 1986, Morantz et al. 1987) found that juvenile salmonids, including brown trout and Atlantic salmon, have no preference for particular water depths. In fact, Heggenes (1996) found a strong preference in young brown trout for deeper water (>60 cm). There is a relatively strong positive relationship between fish length and of depth of suitable habitat (Greenberg et al. 1996, Mäki-Petäys et al. 1997, Heggenes et al. 1999) with the largest trout selecting the deepest habitat. This size structured habitat selection, observed in brown trout in particular, is thought to be an effect of intense intraspecific competition for space. Larger and dominant individuals colonize preferred deep-slow habitats, while smaller subordinate individuals are restricted to shallower fast flowing areas (Heggenes et al. 1999). Dominance is almost always determined by size in salmonids and there are indications that smaller trout colonize deeper-slower areas in the absence of larger trout (Bachman 1984, Huntingford et al. 1990). Deeper waters are also preferred not only in the absence of large conspecifics but the absence of predators of other species as well (Bardonnet and Heland 1994).

Rather than a preference, occupancy in shallow-fast moving habitat may indicate refugia use for smaller conspecifics after being excluded from more energetically profitable deep slow-moving areas. Therefore, these areas may be considered as critical rather than preferred habitat. In order to account for intaspecific size-based competition, that may occur within this size class (≅ 2-7 cm) and to account for allopathic and sympatric trout populations a broader range of suitable habitat seems more desirable that the limited preference ranges suggested to date. Thus, suitable brown trout nursery habitat will be defined as ranging from 2 to 55 cm. This depth series incorporates all the preference ranges suggested, and with the exception of the mean depth of fry observed in the River Todalselva (Bremset and Berg 1999) includes all mean depths reported in the literature reviewed. As discussed earlier, the depth that young fry in this category are capable of utilizing ranges from 2 to > 135 cm. These will be the ranges used for suitable and usable nursery depth for brown trout in the remained of this study. This tolerance range for nursery habitat and summary of the microhabitat studies are is illustrated in Figure 4.2 and outlined in Table 4.2

4.2.1.3 Juvenile depth (fish length: > 7 to 20 cm)

Studies that have investigated juvenile *S. trutta* microhabitat usage come from a much broader range of the species worldwide distribution than those that report on nursery habitat choice. Again, many of these investigations come from northern Europe, including Norway (Heggenes and Saltveit 1990, Bremset and Berg 1999, Heggenes and Dokk 2001), Sweden (Greenberg et al. 1996), Finland (Maki-Petays et al. 1997) and the United Kingdom (Heggenes et al. 2002). Additionally, there are studies from southern European countries including France (Roussel et al. 1999), Italy (Vismara et al. 2001), and Spain (Rincon and Lobon-Cervia 1993). As well there are studies for North America including Canada (Cunjak and Power 1986) and the United States (Raleigh et al. 1986, Shuler et al. 1994). The brown trout populations investigated occurred in both

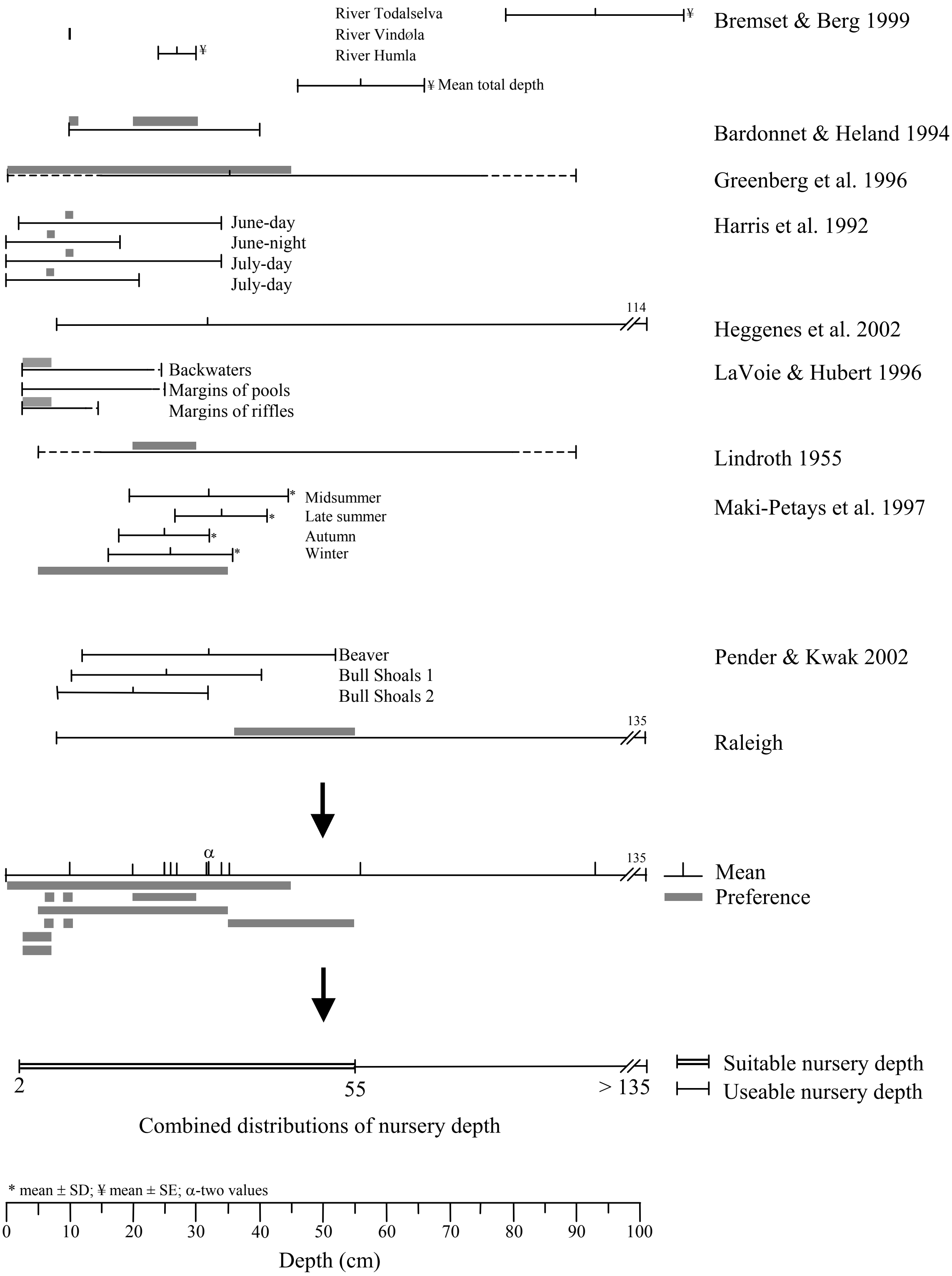


Figure 4.2: Procedure used to define the range of stream depths that can be used by brown trout (*Salmo trutta*) fry (≤ 7 cm. - Nursery Habitat).

Table 4.2: Literature used to define the range of stream depths that can be used by brown trout (*Salmo trutta*) fry (≤ 7 cm).

Source	Mean depth cm	Range [SD] (SE) cm	Number of fish sampled	Preferred depth (FH/SC)	Category range (FH/SC)	Category size (FH/SC) cm	Fish size; mean (range) cm	Study Location	Allo or sympatric Popn's	Other species	Survey method
Bremset and Berg 1999				-	-	-					
R. Todalselva	93	(14)					3.9 (3.1-4.9)	Norway (central)	Sympatric	Atlantic Salmon	Underwater
R. Vindøla	10						4.2 (3.2-4.6)				
R. Humla	27	(3)					5.8 (5.2-6.7)				
Total	56	(10)	22								
Bardonnet and Heland 1994						-	Alevins	France (Lab study)	Sympatric Allopatric	Older B. trout, sculpins	Surface observations
Predators	-	10-40	-	10							
No predators					20-30						
Greenberg et al. 1996								Sweden (northern)	Sympatric	Artic Grayling	Underwater
Diurnal dive	35.3	-	88	< 45	0-15 to 75-90	15	(2.5-6.0)				
Dusk dive	-	-	26	< 45	0-15 to 60-75						
Stone dive	-	-	121	45-75	0-15 to 105-120						
Harris et al. 1992											
June-day	-	-	25	9.8	3-33.6	-	2.6	USA (Wyoming)	Sympatric	Brook trout	Surface
June-night			25	6.7	0-18.3						
July-day			25	9.8	0-33.6		3.9				
July-night			25	6.7	0-21.4						
Heggenes et al. 2002	31.7	8 – 114	273	-	-	-	< 7.0	England (southwest)	Sympatric	Atlantic Salmon	Underwater

Table 4.2: Continued.

Source	Mean depth cm	Range [SD] (SE) cm	Number of fish sampled	Preferred depth (FH/SC)	Category range (FH/SC)	Category size (FH/SC) cm	Fish size; mean (range) cm	Study Location	Allo or sympatric Pop's	Other species	Survey method
LaVoie and Hubert 1996 Backwaters$^{\chi}$ Margins of pools$^{\chi}$ Margins of riffles$^{\chi}$	-	-	22 132 170	2.5-7.0 - 2.5-7.0	2.5-4.5 to 22.5-24.5 <2.5 to >25.0 <2.5 to 12.5-14.5	2 2 2	5.6$^{\chi}$	USA (Wyoming)	Sympatric	Brook trout	Surface: electro-fishing
Lindroth 1955	-	20 to 30	195	20-30	5-15 to 80-90	10	< 7.0 $^{\beta}$	Sweden	Sympatric	Atlantic Salmon	
Mäki-Petäys et al. 1997 Midsummer Late summer Autumn Winter	 32 34 25 26	 [12.6]α [7.2] [7.1] [9.8]	 33 43 26 20	5-35$^{\gamma}$	-	-	4.0 to 9.0	Finland	Allopatric	-	Electro-fishing
Pender and Kwak 2002 Beaver Bull Shoals 1 Bull Shoals 2	 32 25 20	 [19.6]$^{\alpha}$ [14.7]$^{\alpha}$ [11.7]$^{\alpha}$	 24 96 34	 - - -			<6.5 small ≥6.5 large	USA (Missouri & Arkansas)	Sympatric	-	Electro-fishing
Raleigh et al. 1986$^{\delta}$	-	-	> 190	40-55	8 - 135	-	< 14.5	USA (Utah)	Sympatric	-	Underwater

α - S.D. estimated from reported Std. Error [see: (McGhee 1985)]; β – Size of fish not reported. Sampled March-April (1953) thus likely < 7 cm; γ – when observed over all seasons; δ – Based on data from Gosse et al. (1977) and Gosse (1981). Profile interpreted from Raleigh's (1986) SI curve (depth) Figure 7; pg 48; FH/SC – Fish preference reported as a frequency histogram (FH) or habitat suitability curve (SC); χ - median value for August and September allopatric (Rincon and Lobon-Cervia 1993, Maki-Petays et al. 1997, Roussel et al. 1999) and sympatric populations (Cunjak and Power 1986, Heggenes and Saltveit 1990, Shuler et al. 1994, Greenberg et al. 1996, Bremset and Berg 1999, Heggenes and Dokk 2001, Vismara et al. 2001, Heggenes et al. 2002) which shared resources with Atlantic salmon (Heggenes and Saltveit 1990, Bremset and Berg 1999, Heggenes and Dokk 2001, Heggenes et al. 2002), grayling (Greenberg et al. 1996, Vismara et al. 2001) rainbow trout (Shuler et al. 1994), and brook trout (Cunjak and Power 1986). These studies all examined brown trout microhabitat position using underwater observational techniques with the exception of the Finnish (Maki-Petays et al. 1997) and Italian studies (Vismara et al. 2001).

As brown trout age over their first summer some may migrate out of the stream into a lake or a fjord. However, those that remain in the stream will tend to move towards deeper more slowly moving parts of the watercourse (Heggenes et al. 2002). This is evident in this review as the depth in which trout have been observed has increased from > 135 cm to > 200 cm as they move from nursery (Figure 4.2) and into juvenile (Figure 4.3) habitat. Once again, trout will be observed in any water that is accessible to them as evident from the broad reported ranges of some authors. For example, both the work summarized by Raleigh (1986) in the United States and Heggenes and Dokk's (2001) work in Norway report juvenile trout being present in depth ranging from just over 0 to > 200 cm (Figure 4.3). Of course, fish cannot use 'no' water but will be able to access water at least as deep as they are wide (Crisp and Carling 1989, Crisp 1993). The smaller members of the cohort will be able to use at least 2 cm of water while the larger members need approximately 4 cm. As previously discussed with nursery habitat selection, there is probably no theoretical upper limit to water depth that can be used by

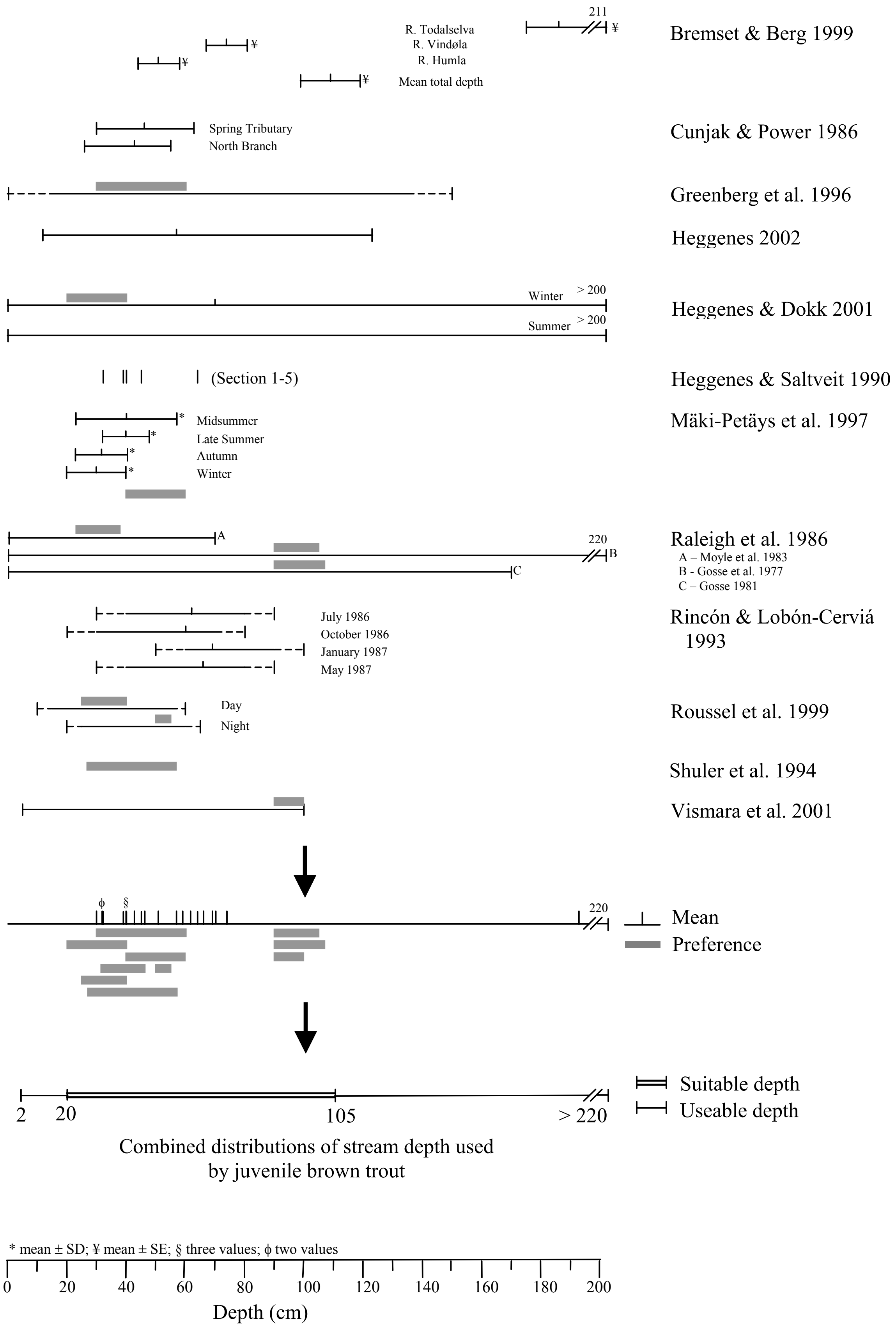


Figure 4.3: Procedure used to define the range of stream depths that can be used by juvenile brown trout (*Salmo trutta*) (> 7 cm – 20 cm).

juvenile trout, however, they may be forced into shallower habitats by larger conspecifics or predators of other species (Bardonnet and Heland 1994, Heggenes et al. 1999).

The opposing pressures of preference for energetically favourable deeper habitat (Gilliam and Fraser 1987, 1988, Ludwig and Rowe 1990) and exclusion from these areas by larger fish results in the distribution of juvenile trout generally observed in natural systems. With one exception, the mean depth observed in the literature summarized in this study ranges from 30 cm during the winter in the River Kuusinkijoki in northeast Finland (Maki-Petays et al. 1997) to 74 cm in the River Vindøla in Norway (Bremset and Berg 1999, Heggenes and Dokk 2001). There are 20 mean depths observed between this range, calculated from measures in eight different studies in six different countries (Table 4.3). As well, the authors of these studies expressed 10 'preference' ranging from 20 cm to 45 cm in the River Kuusinkijoki in Finland to 90 to 107 cm extracted from the SI curve created by Raleigh et al. (1986) in the United States based on data gathered by Gosse (1981). Thus, the bulk of the fish observed in these studies were observed, to preferentially selected habitat between the depths of 20 and approximately 105 cm. The only major exception comes from the River Todalselva in Norway where the mean depth for the fish observed was 193 ±18 (SE) (Bremset and Berg 1999). Bremest and Berg's (1999) investigation was concerned with salmonid (Atlantic salmon and brown trout) microhabitat selection in pools, where these types of studies are more commonly conducted in shallower riffle-run habitats (Cunjak and Power 1986), which may account for the unusually high mean depth reported. Although unusual, this observation does support Heggenes' (1996) argument that trout prefer deeper habitats and will utilize them if available.

Table 4.3: Literature used to define the range of stream depths that can be used by juvenile brown trout (*Salmo trutta*) (>7 – 20 cm).

Source	Mean depth cm	Range [SD] (SE) cm	Number of fish sampled	Preferred depth (FH/SC)[α] cm	Category range (FH/SC)	Category size (FH/SC) cm	Fish size; mean (range) cm	Study Location	Allopatric/ sympatric Pop'ns	Other species	Survey method
Bremset & Berg 1999											
R. Todalselva	193	(18)	72	-	-	-	1+ and > 1+	Norway	Sympatric	Atlantic Salmon	Underwater
R.Vindøla	74	(7)									
R. Humla	51	(7)									
Total	109	(10)									
Cunjak & Power 1986											
Spring Tributary	46.2	29 - 68	11	-	-	-	11.6 (10 – 12)	Canada (Ontario)	Sympatric	Brook Trout	Underwater
North Branch	43.1	26 - 55	9				11.1 (9 – 12)				
Greenberg et al. 1996								Sweden (Norrland)	Sympatric	Grayling	Underwater
Diurnal dive	56.2	-	55	30-60	0-15 to 135-150	15	7-11				
Dusk dive			83	75-135	0-15 to 120-135						
Stone dive			32	30-45	0-15 to 90-105						
Heggenes 2002	56.8	12 –123	941	-	-	-	> 7	England (southwest)	Sympatric	Atlantic Salmon	Underwater
Heggenes & Dokk 2001	70	[52]	120	20-45 (winter)	5 - >200	-	13 (TL) (5-16)	Norway	Sympatric	Atlantic Salmon	Underwater
Heggenes & Saltveit 1990											
Section 1	39	-	26	-	-	-	12	Norway (western)	Sympatric	Atlantic Salmon	Underwater
Section 2	32	-	18	-	-	-	13				
Section 3	40	-	139	-	-	-	9				
Section 4	64	-	81	-	-	-	9				
Section 5	45	-	42	-	-	-	9				

Table 4.3: Continued.

Source	Mean depth cm	Range [SD] (SE) cm	Number of fish sampled	Preferred depth (FH/SC)	Category range (FH/SC)	Category size (FH/SC) cm	Fish size; mean (range) cm	Study Location	Allopatric or sympatric populations	Other species	Survey method
Mäki-Petäys et al. 1997											
Midsummer	40	[17.4](3.7)	22	40-60	5 to >80	10	10-15	Finland (northeast)	Allopatric	-	Electro-fishing
Late Summer	40	[8.3](1.6)	27								
Autumn	32	[8.7](2.1)	17								
Winter	30	[9.9](2.0)	24								
Raleigh et al. 1986											
Moyle at al. 1983	-	-	> 190	23-38	0-70	-	5-30	USA (Utah)	-	-	Underwater
Gosse et al. 1977				90-105	0-220						
Gosse 1981				90-107	0-167						
Rincón & Lobón-Cerviá 1993											Underwater
July 1986	61.5	-	54	-	30-40 to 80-90	10	12.6	Spain	Allopatric	-	
October 1986	59.1		58		20-30 to 70-80	10	13.1				
January 1987	68.9		39		50-60 to 90-100	10	14.1				
May 1987	66.4		42		30-40 to 80-90	10	13.4				
Roussel et al. 1999											Underwater
Day	-	-	98	25-40	10-15 to 55-60	5	10-20	France (Brittany)	Allopatric	-	
Night				50-55	20-25 to 60-65	5					
Shuler et al. 1994	-	-	154	27-57	-	-	13-19	USA (Colorado)	Sympatric	Rainbow trout	Underwater
Vismara et al. 2001	-	-	528*	90-100	5-10 to 90-100	10	12-22	Italy	Sympatric	Grayling	Electro-fishing

* includes adults and juveniles; FH/SC – Fish preference reported as a frequency histogram (FH) or habitat suitability curve (SC)

Based on the observation of juvenile *S. Trutta* we construct a general tolerance profile for water depth microhabitat choice. This age-class of brown trout can use water depths ranging from approximately 5 cm (if we are to include the larger members of the cohort) to depths greater than 220 cm. However, suitable juvenile trout habit will be between approximately 20 and 105 cm. These will be the ranges used for suitable and usable juvenile brown trout water depth in the remainder of this study. The creation of this tolerance range and a summary of the microhabitat studies employed are illustrated in Figure 4.3 and outlined in Table 4.3.

4.2.1.4 Adult non-spawners (fish length > 20 cm)

Like studies that have investigated juvenile *S. trutta* microhabitat choice, work involving water depth selection of adult non-spawners covers a large portion of the species current worldwide distribution. Unlike other tolerance profiles, a larger portion of these studies comes from North America. There are three studies from the United States (Baldes and Vincent 1969, Raleigh et al. 1986, Strakosh et al. 2003) and one study from Canada (Cunjak and Power 1986). As well, there are three studies from Scandinavia (Heggenes 1988a, Greenberg et al. 1996, Maki-Petays et al. 1997, Mäki-Petäys et al. 1997), one from southern Europe (Vismara et al. 2001) and two from New Zealand (Shirvell and Dungey 1983, Hayes and Jowett 1994). These studies involved allopatric (Baldes and Vincent 1969, Heggenes 1988a, Maki-Petays et al. 1997, Strakosh et al. 2003), sympatric (Cunjak and Power 1986, Greenberg et al. 1996, Vismara et al. 2001) as well as a study that involves both types of populations (Shirvell and Dungey 1983). The survey methods used in these studies include surface observations (Baldes and Vincent 1969, Shirvell and Dungey 1983, Hayes and Jowett 1994), underwater surveys (Cunjak and Power 1986, Raleigh et al. 1986, Greenberg et al. 1996) and electro-fishing techniques (Heggenes 1988a, Maki-Petays et al. 1997, Vismara et al. 2001). As well seasonal habitat changes are included as both Mäki-Petäys and his colleagues (1997) and Cunjak and Power (1986) examined population during the winter.

Of the studies reporting observations from natural riverine systems, the shallowest depth reported for adult non-spawners comes from Shirvell and Dungey (1983). They report large brown trout (32-55 cm) using depths as shallow as 14 cm. As previously discussed, brown trout are capable of using any water deep enough to access. As the largest fish observed was 65 cm (Hayes and Jowett 1994) and using the 0.2 width to length conversion (Crisp and Carling 1989, Crisp 1993) the minimum depth, which can accommodate all members of this cohort, will be set at 15 cm. Hayes and Jowett (1994) also report the deepest water depth, 310 cm, used by *S. Trutta*. Thus, the range of water depths that this species has been observed using, from the literature surveyed, ranges from approximately 15 to 310 cm. This range will be employed as the 'useable' component of the habitat tolerance profile.

As trout grow their use of deeper habitat increases (Bohlin 1977, Kennedy and Strange 1982). With the exception of Baldes and Vincent's (1969) artificial stream study, the distribution of reported means ranges from 33 cm from the autumn survey River Kuusinkijoki in Finland (Maki-Petays et al. 1997) to 84.5 cm in the River Vojmån in northern Sweden (Greenberg et al. 1996). The preference ranges expressed by authors or calculated from suitability index curves begins at 50 cm or >50 cm (Heggenes 1988a, Maki-Petays et al. 1997) with the deepest preferred depth of the ranges being, of 135 cm, being calculated by Greenberg and his colleagues (1996). Brown trout's preference for deep water (Heggenes 1996) and this species aggressive territoriality, which allows it to displace members of sympatric populations (Kennedy and Strange 1986, Bremset and Berg 1999, Heggenes et al. 1999) would suggest that adult brown trout will utilize the deepest stream locations available (Wesche et al. 1987, Heggenes 1988a). Unlike members of smaller cohorts, who use shallower habitats as refugia after being dislocated by larger conspecifics, adult brown trout may use deeper habitats both because they are the most energetically favourable positions (Bohlin 1977, Fausch 1984, Hughes and Dill 1990) but also as refugia from wading or diving predators (Power 1987, Schlosser 1987, Harvey and Stewart 1991). Thus, the deeper stream areas such as pools, should be considered the crucial habitat within the available range. In fact (Heggenes et al. 1999) suggest that trout may be space restricted in streams as suitable water depths may be limiting.

Based on the observation of adult non-spawning *S. Trutta* (> 20 cm) a general tolerance profile for water depth microhabitat choice was constructed. This size-class of brown trout can use water depths ranging from approximately 15 cm to depths of 310 cm or possibly greater. However, suitable adult non-spawning trout habitat will be between approximately 33 and 135 cm. These will be the ranges used for suitable and usable juvenile brown trout water depth in the remainder of this study. The creation of this tolerance range and a summary of the microhabitat studies employed are illustrated in Figure 4.4 and outlined in Table 4.4.

4.2.1.5 Summary of tolerance profiles

The tolerance profiles for water depth, created for each size class, are contrasted in Figure 4.5. An interesting trend is evident in that the range of depths available for use

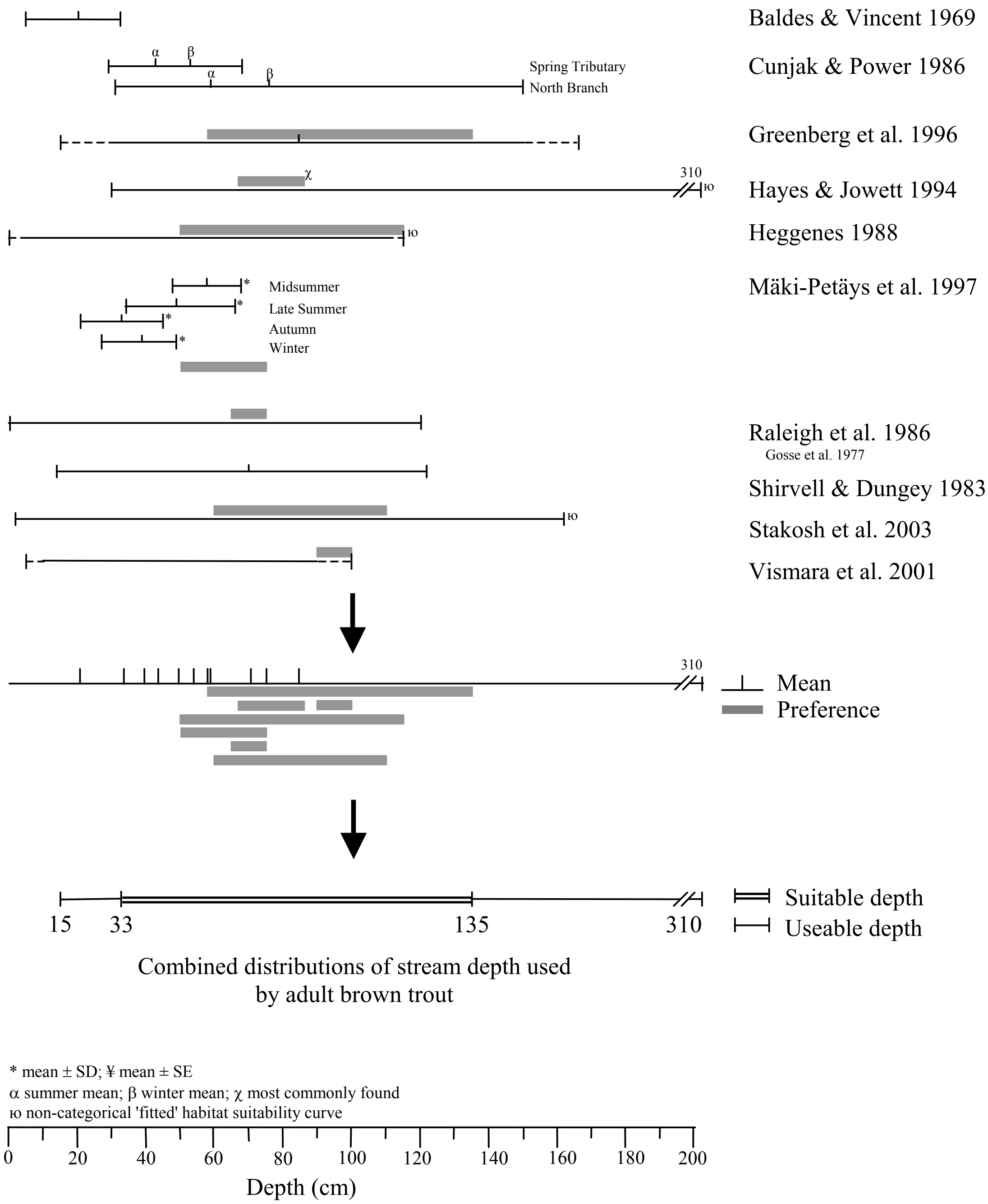


Figure 4.4: Procedure used to define the range of stream depths that can be used by adult brown trout (*Salmo trutta*) (> 20 cm).

Table 4.4: Literature used to define the range of stream depths that can be used by adult brown trout (*Salmo trutta*) (>7 – 20 cm).

Source	Mean depth cm	Range [SD] (SE) cm	No. of fish sampled	Preferred depth (FH/SC)	Category range (FH/SC)	Category size - cm (FH/SC)	Fish size; mean (range) cm	Study Location	Allo-sympatric popn's	Other species	Survey method
Baldes & Vincent 1969	20.3	5.1 – 32.5	108	-	-	-	20.3[α]	USA (Colorado)	Allopatric	-	Surface (artificial stream)
Cunjak & Power 1986											
Spring Trib.	43.0[β];53.4[χ]	29 - 68[χ]	16[χ]	-	-	-	20.3 (12-40)[χ]	Canada (Ontario)	Sympatric	Brook Trout	Underwater
North Branch	58.6[β];75.6[χ]	31 - 150[χ]	36[χ]				21.9 (12-50)[χ]				
Greenberg et al. 1996											
Diurnal dive	84.5	-	111	60-135	15-30 to 150-175	15	12-35	Sweden (Norrland)	Sympatric	Grayling	Underwater
Dusk dive	-	-	55	60-120	30-45 to 135-150						
Hayes & Jowett 1994		30-310	189	67-98[δ]	-	-	45-65	New Zealand	-	-	Surface
Heggenes 1988											
Low density	-	-	19[ε]	> 50	0-5 to 110-115	5	24.7 (11.2-43.3)	Norway (southeast)	Allopatric	-	Electro-fishing
High density			130[φ]				17.0 (12.1-27.5)				
Mäki-Petäys et al. 1997											
Midsummer	58	[10.3](2.3)	20	50-75	5 to >80	10	10-15	Finland (northeast)	Allopatric	-	Electro-fishing
Late Summer	49	[16.6](4.8)	12								
Autumn	33	[12.0](3.0)	16								
Winter	39	[9.5](3.0)	10								
Raleigh et al. 1986 Gosse 1981	-	-	> 190	65-75	0-120	-	> 24	USA (Utah)	-	-	Underwater
Shirvel & Dungey 1983	65	14-122	140	-	-	-	42 (32-55)	New Zealand	Allopatric Sympatric	Rainbow trout	Surface

Table 4.4: Continued.

Source	Mean depth cm	Range [SD] (SE) cm	No. of fish sampled	Preferred depth (FH/SC)	Category range (FH/SC)	Category size - cm (FH/SC)	Fish size; mean (range) cm	Study Location	Allo-sympatric popn's	Other species	Survey method
Stakosh et al. 2003	-	-	144	60-110	10-170	-	≥ 17	USA (Conn)	Allopatric	-	Underwater
Vismara et al. 2001	-	-	528[ε]	90-100	5-10 to 90-100	10	12-22	Italy	Sympatric	Grayling	Electro-fishing

α mean size of 6 fish only; β summer; χ winter; δ most commonly found; ε juvenile and adults combined

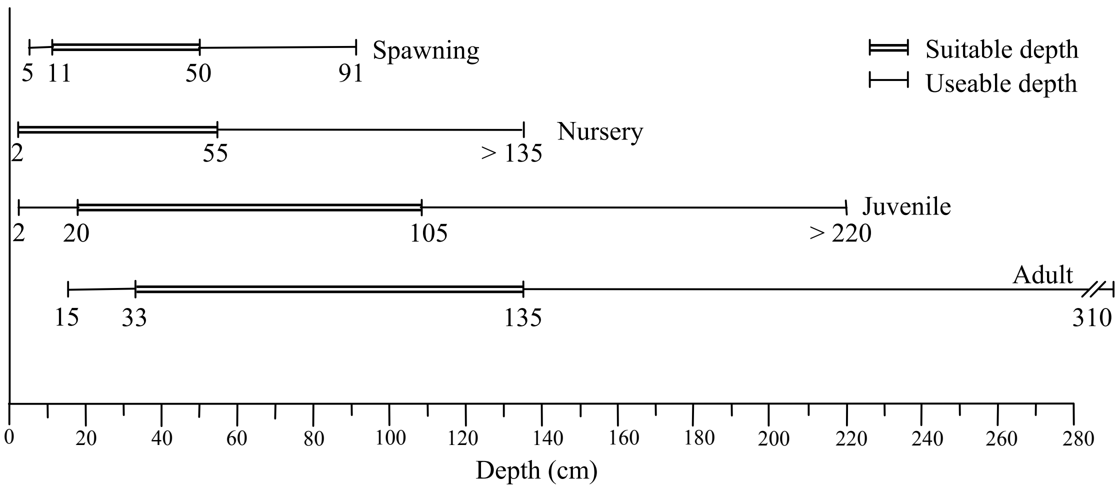


Figure 4.5: Comparison of tolerance profiles constructed for water depths for four size classes of brown trout (*Salmo trutta*), which including spawning, nursery (≤ 7 cm), juvenile (>7 – 20 cm), and adult residents (> 20 cm).

increases with the size-class and the depths that can be used and are suitable for use by brown trout also increases with depth. This seems to support what has been remarked on in the literature summaries in that larger fish prefer deeper habitat. Unlike the other profiles, young fry ≤ 7 cm (nursery habitat) are suited to the shallowest habitat available within the tolerance range. The profile was set to include all member of the cohort but presumable the smaller members are well suited to using marginal habitats (less than 2 cm) and will inhabit any water depth available no matter how shallow. It has been mentioned that no theoretical limits exist, the deepest observations for brown trout increase with size class. Fry in their first summer (using nursery habitat) can use habitat > 135 cm, juveniles > 220 cm, while adults can use depths up to 310 cm. Again, these values reflect a trend of deeper water for larger fish, but may also be a consequence of habitat availability and observational bias (Heggenes 1988b).

4.2.2 Habitat suitability

The profiles constructed in Section 4.2.1 were then applied to the data gathered from each quadrat measured in the reaches studies and the proportion of available habitat for each study reach at each life stage is summarized in the following sections.

4.2.2.1 Spawning depth

The proportion of suitable, useable, and not-useable water depth for both the total and wetted streambed that is available at each site was compared using a chi-squared test. This was done to compare all six sites as well as the data collected before and after the spate at the downstream Bin Burn site (i.e. a total of seven data sets were compared). Further chi-square tests were conducted to compare the proportions of suitable, useable, and not-useable habitat for both the total and wetted streambed that was available before and after the spate in Bin Burn at the downstream site. The results of these analyses are displayed in Table 4.5 and 4.6. The chi-squared value for both the total (X^2 = 297.5, df = 18, p = 0.000) and the wetted (X^2 = 96.1, df = 12 p = 0.000) area of the streambed indicate that the relative proportions of the spawning depth categories are not independent of reach (i.e. habitat classifications vary from site to site). The proportion of each depth suitability classification for spawning depth is outlined in Table 4.7. For the total streambed, suitable spawning depths ranged from 9.5% to 72.1% at the upstream sites of Burnhouse Burn and March Burn, respectively. The spawning depth classed as useable ranged from 15.4% to 22.6% at the upstream sites of March and Bin Burn, respectively. The lowest amount of non-useable spawning depth was found at March Burn-upstream and the highest at Bin Burn-upstream. Dry streambed ranged from 4.9% at March Burn-upstream to 61.4% at Burnhouse Burn-upstream.

Table 4.5: Results for chi-square test (α = 0.05) comparing the proportions of quadrats that contained depths that were considered suitable, useable, not-usable and dry for brown trout in both the total and wetted streambed at all study reaches.

Life Stage	Streambed	X^2 value	DF	Cells < 5†	p-value	X^2 (Likelihood)*
Spawn	Total	297.5	18	0	0.000	no
	Wetted	96.1	12	0	0.000	no
Nursery	Total	251.7	18	9	0.000	yes
	Wetted	58.9	12	9	0.000	yes
Juvenile	Total	315.1	18	2	0.000	yes
	Wetted	124.2	12	2	0.000	yes
Adult	Total	305.5	18	2	0.000	yes
	Wetted	114.6	12	2	0.000	yes

† Number of cells with expected values less than 5; * Used likelihood ratio chi-square

Table 4.6 Results for chi-square test (α = 0.05) comparing the proportions of quadrats that contained depths that were considered suitable, useable, and not-unusable and dry for spawning brown trout for both the total and wetted streambed at Bin Burn (downstream) before and after the spate.

Life Stage	Streambed	X^2 value	DF	Cells < 5†	p-value	X^2 (Likelihood)*
Spawn	Total	97.2	3	0	0.000	no
	Wetted	31.2	2	0	0.000	no
Nursery	Total	80.0	2	0	0.000	no
	Wetted	15.1	1	0	0.000	no
Juvenile	Total	108.1	3	0	0.000	no
	Wetted	41.2	2	0	0.000	no
Adult	Total	101.5	3	0	0.000	no
	Wetted	35.5	2	0	0.000	no

† Number of cells with expected values less than 5; * Used likelihood ratio chi-square

In the wetted portion of the streambed the highest proportion of suitable spawning depth was found at March Burn-upstream (75.8%) and the lowest at Burnhouse Burn-upstream (24.6%). March Burn-upstream had the lowest proportion of useable spawning depth (22.1%) and Burnhouse Burn-upstream had the highest (48.2%). March Burn-upstream also had the lowest proportion of unusable spawning depth

Table 4.7: Summary of the proportion of water depth available for the four life stages of brown trout for all study reaches including both total and the wetted portions of the streambed.

		Spawn Total (%)	Wetted (%)	Nursery Total (%)	Wetted (%)	Juvenile Total (%)	Wetted (%)	Adult Total (%)	Wetted (%)
March Burn Upstream	Suitable	72.1	75.8	95.1	100.0	32.5	34.2	9.5	10.0
	Useable	21.0	22.1	0.0	0.0	62.6	65.8	46.5	48.8
	Not useable	2.0	2.1	0.0	0.0	0.0	0.0	39.2	41.2
	Dry	4.9	-	4.9	-	4.9	-	4.9	-
March Burn Downstream	Suitable	31.2	52.0	57.9	96.5	13.1	21.8	4.3	7.1
	Useable	15.4	25.6	0.9	1.5	45.7	76.0	14.9	24.8
	Not useable	13.5	22.4	1.2	2.0	1.3	2.2	40.9	68.1
	Dry	40.0	-	40.0	-	40.0	-	40.0	-
Burnhouse Burn Upstream	Suitable	9.5	24.6	37.1	96.2	0.8	2.1	0.0	0.0
	Useable	18.6	48.2	0.0	0.0	35.5	91.9	4.1	10.7
	Not useable	10.5	27.2	1.5	3.8	2.3	6.0	34.5	89.3
	Dry	61.4	-	61.4	-	61.4	-	61.4	-
Burnhouse Burn Downstream	Suitable	13.6	34.3	38.2	96.3	2.1	5.4	0.0	0.0
	Useable	15.7	39.5	0.0	0.0	35.6	89.9	7.2	18.2
	Not useable	10.4	26.2	1.5	3.7	1.9	4.8	32.4	81.8
	Dry	60.4	-	60.4	-	60.4	-	60.4	-
Bin Burn Upstream	Suitable	36.1	48.1	70.5	94.0	25.1	33.4	12.7	16.9
	Useable	22.6	30.1	2.9	3.9	48.3	64.4	19.7	26.2
	Not useable	16.4	21.8	1.6	2.2	1.6	2.2	42.7	56.9
	Dry	25.0	-	25.0	-	25.0	-	25.0	-
Bin Burn Downstream (Pre-spate)	Suitable	35.2	52.4	66.2	98.7	16.6	24.8	3.4	5.1
	Useable	21.5	32.0	0.0	0.0	49.9	74.4	21.4	31.9
	Not useable	10.4	15.5	0.9	1.3	0.6	0.8	42.3	63.0
	Dry	32.9	-	32.9	-	32.9	-	32.9	-
Bin Burn Downstream (Post-spate)	Suitable	12.5	29.9	37.7	90.6	3.8	9.2	1.7	4.2
	Useable	19.0	45.7	0.0	0.0	33.9	81.4	5.0	11.9
	Not useable	10.2	24.4	3.9	9.4	3.9	9.4	34.9	83.9
	Dry	58.4	-	58.4	-	58.4	-	58.4	-

.

(2.1%) and Burnhouse Burn-upstream had the highest (27.2%). The relative proportions of suitable, useable, and not-useable spawning depth for both total and wetted stream areas is illustrates in Figures 4.6 (A-G). Both the discharge and surface area within the study reaches varies from site to site (Table 4.8); thus, the area available with suitable spawning depth at each site can be calculated by multiplying the proportion of suitable and useable habitat by the wetted surface area at each reach. Burnhouse Burn-upstream and downstream had the smallest amount of streambed with adequate spawning depth available (8.4 m^2 and 18.4 m^2, respectively) and Bin Burn had the largest [25.2 m^2, 50.1 m^2, and 68.1 m^2: upstream, downstream (pre-spate), downstream (post spate)]. March Burn had intermediate values for the upstream and downstream study reaches (11.7 m^2 and 38.3 m^2, respectively - Table 4.9).

The spate in Bin Burn-downstream followed by the debris removal and reconstruction resulted in a different looking reach. The chi-square analysis (Table 4.6) indicates that the proportion of suitable, useable and not-useable spawning depth is different, pre- and post-spate, at this stream section. The discharge at the time of sampling was similar ($Q_{pre} = 0.0703$ m^3/sec, $Q_{post} = 0.0771$ m^3/sec; Table 4.8) the total streambed area was not (pre-spate – 87.7 m^2; post-spate - 216.3 m^2). The total area that had spawning depths available for brown trout (suitable and useable) went from 50.1 m^2 before the spate to 68.1 m^2 after the spate.

4.2.2.2 Nursery depth (fish length ≤ 7 cm)

Like spawning depth, the chi-squared analysis for the relative proportion of depth that is suitable, useable, and not-useable for use by young trout (≤ 7 cm) is dependent on site for both the total ($X^2 = 251.7$, df = 18, p = 0.000) and the wetted ($X^2 = 58.9$, df = 12 p =

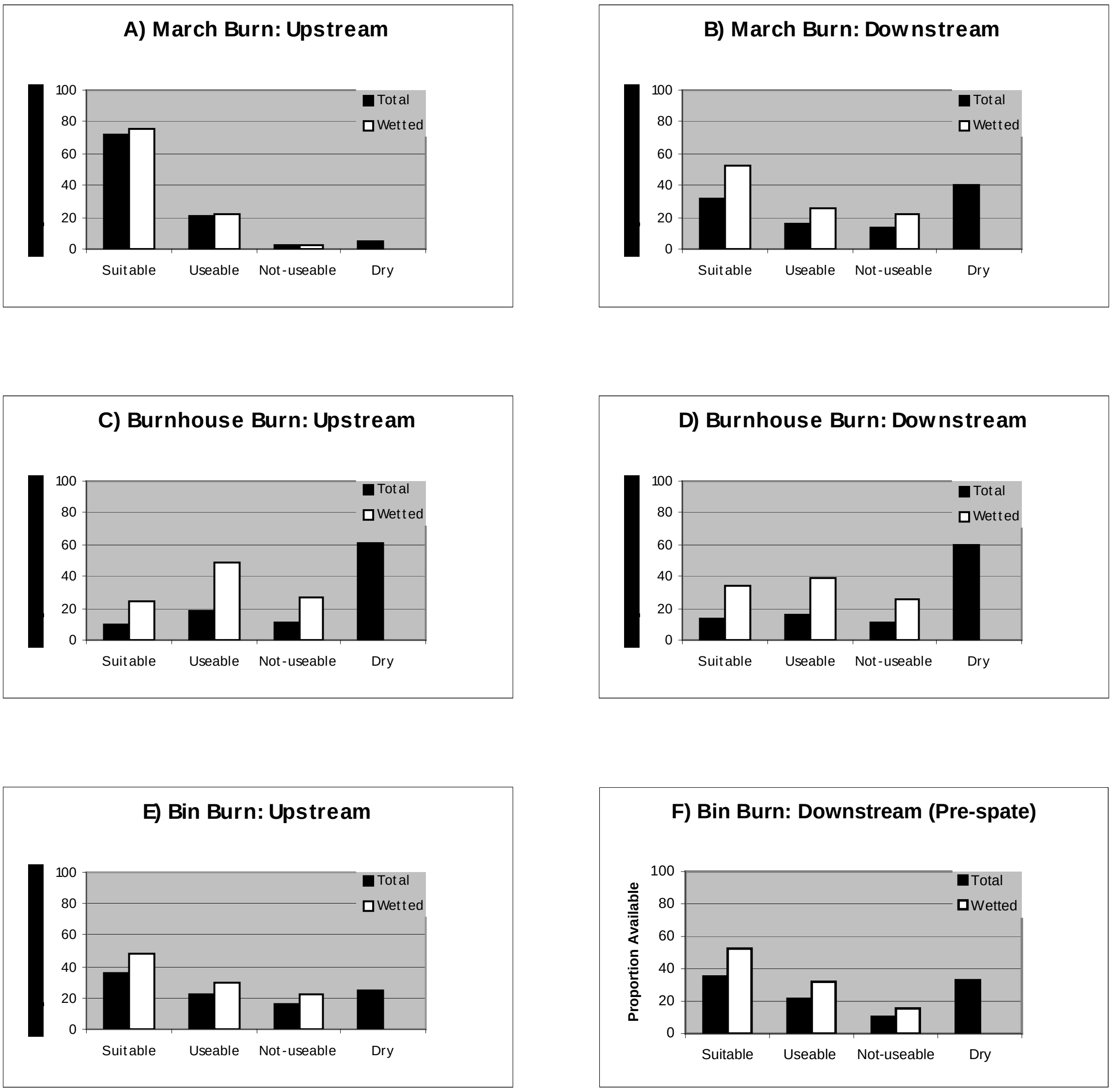


Figure 4.6 (A-G): Histograms of the proportion of spawning depth at all study sites classified as suitable, useable, not-useable, and dry for both the total and wetted streambed.

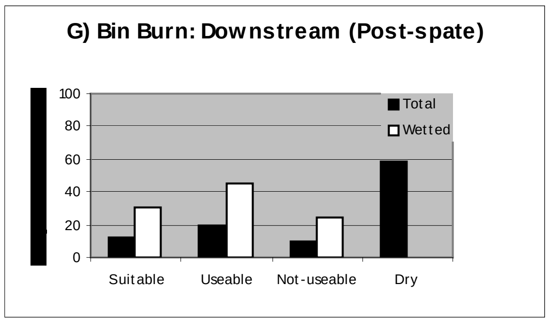


Table 4.8: Discharge and surface area (total, wetted and dry) at all study sites examined in the Carron Valley.

Site	Discharge (m^3/sec)	Std Dev (m^3/sec)	Number of transects	Total streambed (m^2)	Wetted Streambed (m^2)	Dry Streambed (m^2)
March Burn US (22/07/2002)	0.0017	0.0011	3	12.6	12.0	0.5
March Burn DS (17/07/2002)	0.0341	0.0119	3	80.5	48.8	31.7
Burnhouse US (20/08/2002)	0.0031	0.0026	3	30.1	11.6	18.5
Burnhouse DS (28/08/2008)	0.0058	0.0007	3	58.8	24.9	33.9
Bin Burn US (14/08/2002)	0.0294	0.0228	3	43.0	32.2	10.8
Bin Burn DS (pre-spate) (24/07/2002)	0.0703	0.0424	3	87.7	59.4	28.4
Bin Burn DS (post-spate) (06/08/2002)	0.0771	0.0237	3	216.3	90.1	129.3

0.000) area of the streambed (i.e. habitat classifications vary from site to site – Table 4.5). The relative proportion of suitable, useable, not-useable, and dry streambed is outlined in Table 4.7 and illustrated in Figure 4.7 (A-G). These streams were much better suited as nursery habit. When looking at the total streambed, suitable depth for small trout ranged from 37.1% at Burnhouse Burn-upstream to 95.1% at March Burn-upstream. Useable depths ranged from 0% at all sites except March Burn-downstream and Bin Burn-upstream, which had the highest proportion of useable depths (2.9%). The proportion of total streambed not useable as spawning habitat was very low ranging from 0% at March Burn-upstream to 3.9% at Bin Burn-downstream (post-spate). The proportions of dry streambed are the same as previously noted.

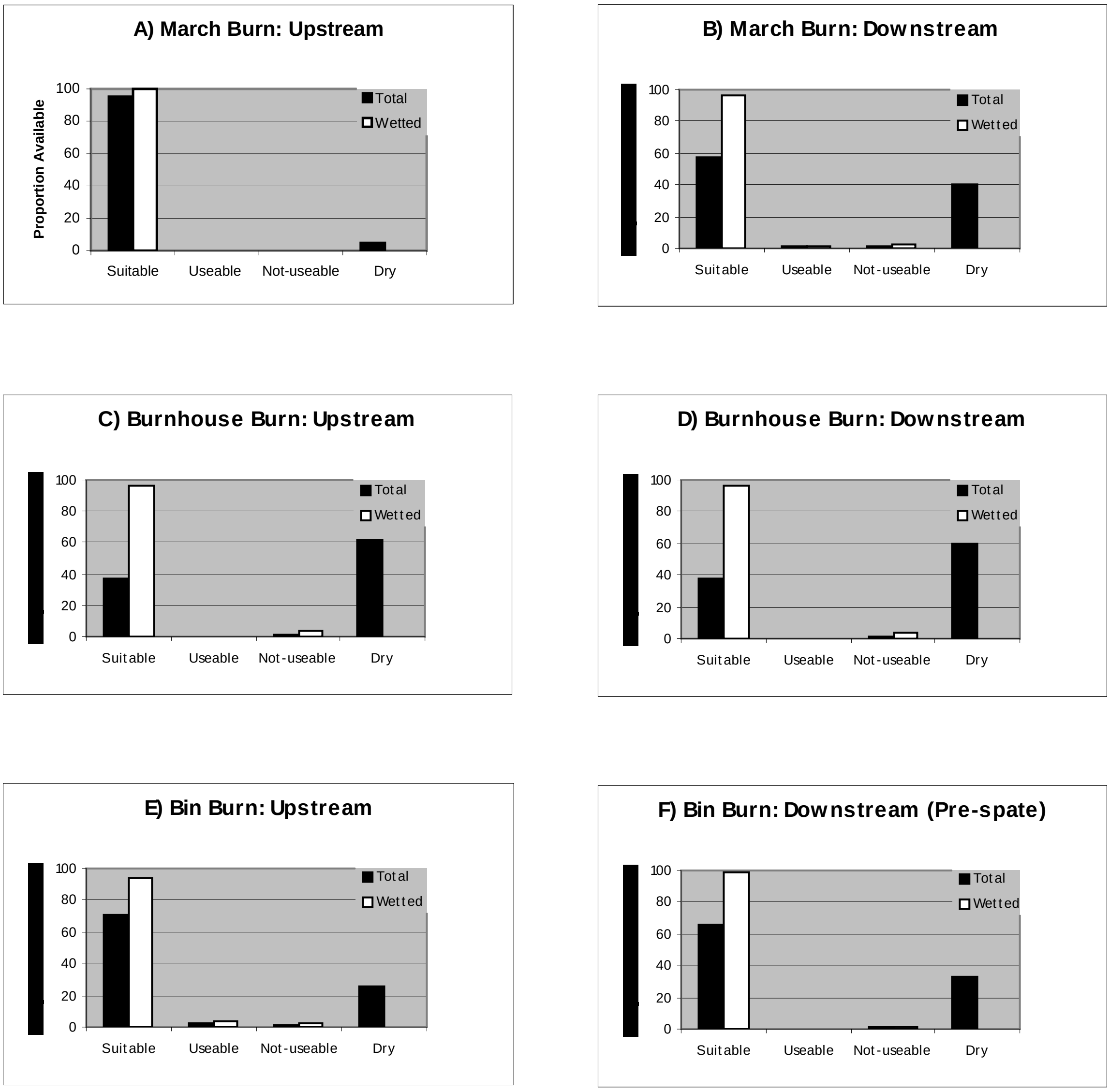
When considering only the wetted area of the streambed all streams were well suited as nursery habitat, at least with respect to water depth. The proportion of quadrats that are

Table 4.9: The total area that has accessible water depths for each study reach for age class. Total accessible area calculated by multiplying total wetted area by the sum of the proportion of the streambed that was either suitable or useable.

Age Class	Site	Wetted Area (m^2)	Proportion of wetted area; suitable or useable (%)	Total area suitable or useable (m^2)
Spawning	March Burn-us†	12.0	97.9	11.7
	March Burn-ds‡	48.8	78.5	38.3
	Burnhouse Burn-us	11.6	72.8	8.4
	Burnhouse Burn-ds	24.9	73.8	18.4
	Bin Burn-us	32.2	78.2	25.2
	Bin Burn-ds (pre flood)	59.4	84.4	50.1
	Bin Burn-ds (post flood)	90.1	75.6	68.1
Nursery	March Burn-us	12.0	100.0	12.0
	March Burn-ds	48.8	98.0	47.8
	Burnhouse Burn-us	11.6	96.2	11.2
	Burnhouse Burn-ds	24.9	96.3	24.0
	Bin Burn-us	32.2	97.9	31.5
	Bin Burn-ds (pre flood)	59.4	98.7	58.6
	Bin Burn-ds (post flood)	90.1	90.6	81.6
Juvenile	March Burn-us	12.0	100.0	12.0
	March Burn-ds	48.8	97.8	47.7
	Burnhouse Burn-us	11.6	94.0	10.9
	Burnhouse Burn-ds	24.9	95.3	23.7
	Bin Burn-us	32.2	97.8	31.5
	Bin Burn-ds (pre flood)	59.4	99.2	58.9
	Bin Burn-ds (post flood)	90.1	90.4	81.5
Adult	March Burn-us	12.0	58.8	7.1
	March Burn-ds	48.8	31.9	15.6
	Burnhouse Burn-us	11.6	10.7	1.2
	Burnhouse Burn-ds	24.9	18.2	4.5
	Bin Burn-us	32.2	43.1	13.9
	Bin Burn-ds (pre flood)	59.4	37.0	22.0
	Bin Burn-ds (post flood)	90.1	16.1	14.5

† upstream; ‡ downstream

classified as suitable range from 90.6% to 100% [Bin Burn-downstream (post-spate) and March Burn-upstream, respectively). The bulk of the sites have no 'usable' nursery depth with the exception of March Burn-downstream (1.5%) and Bin Burn-upstream



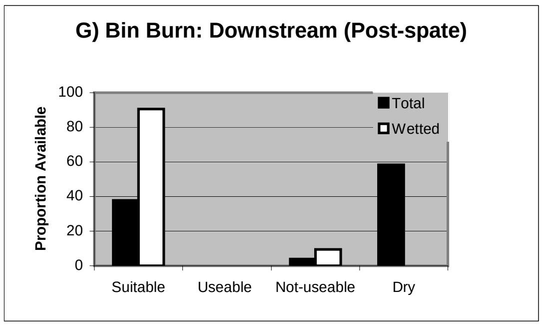


Figure 4.7 (A-G): Histograms of the proportion of water depth available to young trout (≤ 7 cm) at all study sites classified as suitable, useable, not-useable, and dry for both the total and wetted streambed.

(3.9%). Very little of the available depth were not-useable, ranging from 0% at March Burn-upstream to 9.4% at Bin-Burn-downstream (post-spate).

The area available with adequate depth (proportion of suitable and useable summed multiplied by the wetted area of the reach) for nursery brown trout followed a similar pattern as that seen for spawning depths. Burnhouse Burn-upstream and downstream had the smallest amount of streambed with adequate spawning depth available (11.2 m^2 and 24.0 m^2, respectively) and Bin Burn the largest [31.5 m^2, 58.6 m^2, and 81.6 m^2: upstream, downstream (pre-spate), downstream (post spate)]. March Burn had intermediate values for the upstream and downstream study reaches (12.0 m^2 and 47.8 m^2, respectively - Table 4.9).

Again, the spate in Bin Burn-downstream resulted in changes in the depth available for nursery fish. The chi-square analysis (Table 4.6) indicates that the relative proportion of nursery depths classified into the suitability criteria is different, pre- and post-spate, at this stream section. The chi-squared values for both the total and wetted streambed indicate that there is a relationship between site and habitat availability (total: $X^2 = 80.0$, df = 2, p = 0.000; wetted: $X^2 = 15.1$, df = 1, p = 0.000). This analysis is somewhat different from the previous chi-squared as there were no usable depths both before and after the spate; thus, these cells were eliminated from the analysis. This resulted in a 2x2 not a 2x3 contingency table. Again, The discharge at the time of sampling this site pre- and post-spate was similar ($Q_{pre} = 0.0703$ m^3/sec, $Q_{post} = 0.0771$ m^3/sec; Table 4.8); however, the total area that had depths available for young trout (≤ 7 cm) (suitable and useable) went from 58.6 m^2 before the spate to 81.6 m^2 after the spate.

4.2.2.3 Juvenile depth (fish length > 7 to 20 cm)

The chi-squared test comparing the relative proportions of the suitability criteria with sites revealed that there is a relationship between site and the water depth available to juvenile brown trout. The chi-squared values for both the total and wetted streambed were $X^2 = 315.1$ (df = 18, p = 0.000) and $X^2 = 124.2$ (df = 12, p = 0.000), respectively (Table 4.5). Like the water depth available as nursery habitat most of the quadrats can be classified as either suitable or useable by juvenile trout. However, where the analysis of depth available for nursery fish was dominated by suitable habitat, the juvenile classifications are dominated by the 'useable' category. In other words, the water depth available at these sites is less suitable but still available to juvenile trout [Table 4.7 and Figure 4.8 (A-G)]. In the total streambed, suitable depth ranged from 0.8% at Burnhouse Burn-upstream to 32.5% at March Burn-upstream. Useable depth ranged from 33.9 % at Bin Burn-downstream (post-spate) to 62.6% at March Burn-upstream. Water depths that were not useable by juvenile trout ranged from 0% at March Burn-upstream to 3.9% at Bin Burn-downstream (post-spate).

When considering the wetted area alone the bulk of the quadrats were either suitable or useable by juvenile trout. The proportion of suitable quadrats ranged from 2.1% at Burnhouse Burn-upstream to 34.4% at March Burn-upstream. The bulk of the available habitat in the wetted streambeds was classified as useable. Useable depths ranged from 64.4% to 91.9% at Bin Burn-upstream and Burnhouse Burn-upstream, respectively. There were very few quadrats with depth that were not useable by juvenile trout. Quadrats classified as 'not-useable' ranged from 0% at March Burn-upstream to 9.4% at Bin Burn-downstream (post-spate).

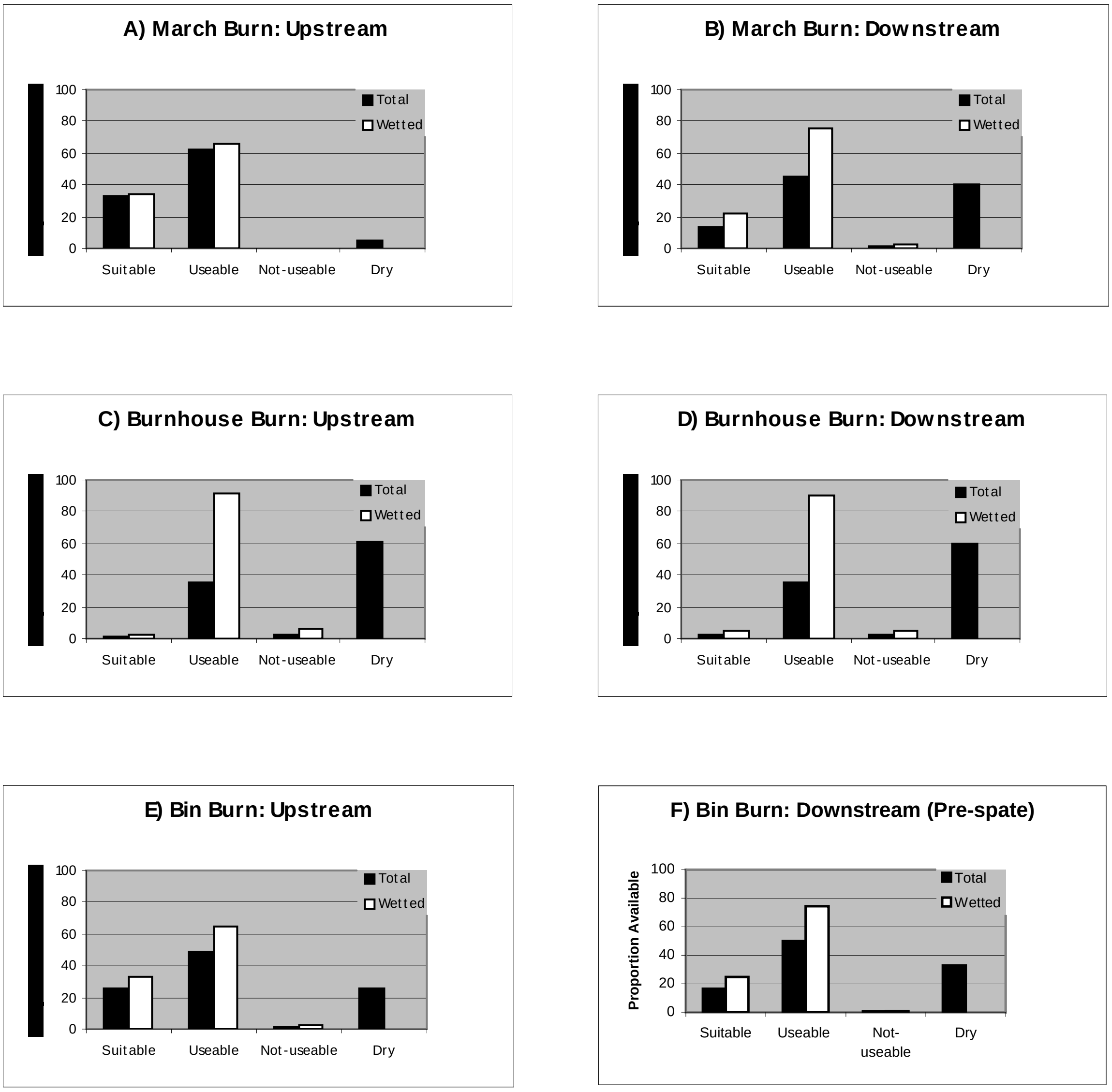
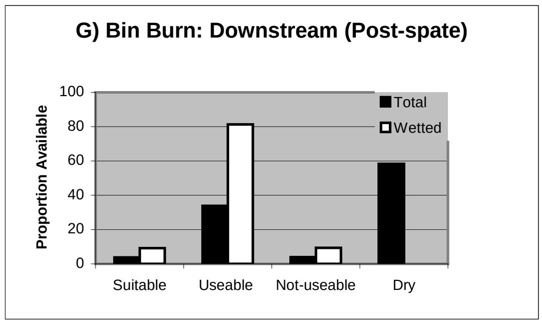


Figure 4.8 (A-G): Histograms of the proportion of water depth available to juvenile trout (> 7 cm to 20 cm) at all study sites classified as suitable, useable, not-useable and dry for both the total and wetted streambed.



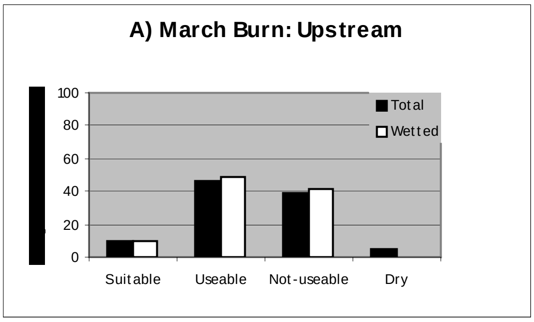
The area available with adequate depth (proportion of suitable and useable summed multiplied by the wetted area of the reach) for juvenile brown trout followed a very similar pattern as that seen for nursery depths. Burnhouse Burn-upstream and downstream had the smallest amount of streambed with adequate spawning depth available (10.9 m^2 and 23.7 m^2, respectively) and Bin Burn the largest [31.5 m^2, 58.9 m^2, and 81.5 m^2: upstream, downstream (pre-spate), downstream (post spate)]. March Burn had intermediate values for the upstream and downstream study reaches (12.0 m^2 and 47.7 m^2, respectively - Table 4.9).

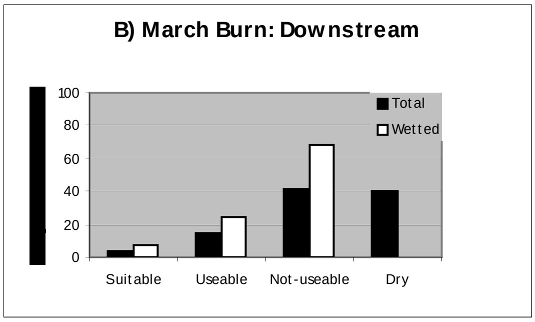
Again, the chi-squared analysis revealed that the proportions of habitat classified based on the suitability criteria are different at Bin Burn-downstream before and after the spate. The chi-squared value for the total streambed (X^2 = 108.1, df = 3, p = 0.000) and the wetted surface (X^2 = 41.2, df = 2, p = 0.000) are outlined in Table 4.6. The percentage of the available water depths (suitable plus useable) were 66.5% and 99.2% for the total and wetted streambed before the spate and 37.8% and 90.4% after the spate. The total useable area with water depth available for use by juvenile trout went from 58.9 m^2 before the spate to 81.5 m^2 after the spate.

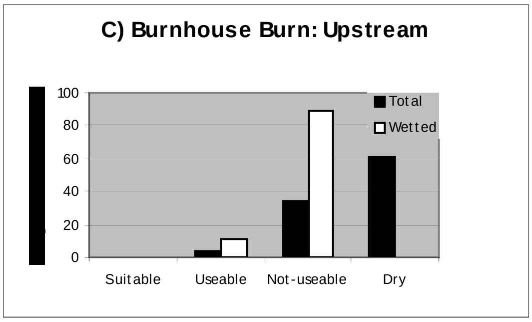
4.2.2.4 Adult non-spawners (fish length > 20 cm)

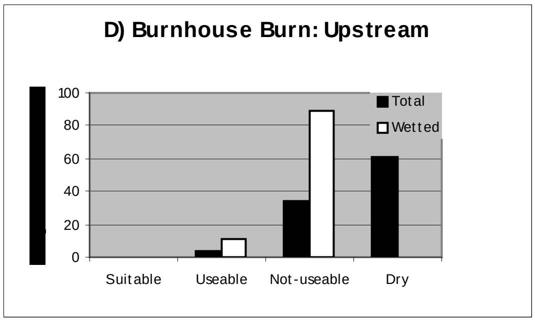
Finally, like the other age classes, the chi-squared test comparing the relative proportions of the suitability criteria with sites revealed that there is a relationship between site and the water depth available to adult brown trout. The chi-squared values for both the total and wetted streambed were X^2 = 305.5 (df = 18, p = 0.000) and X^2 = 114.6 (df = 12, p = 0.000), respectively (Table 4.7). Unlike nursery and juvenile depth classifications a smaller proportion of the water depths available could be considered suitable for adult trout [Figure 4.9(A-G)]. Suitable water depth ranged from 0% at Burnhouse Burn (both up- and downstream sites) to 12.7% at Bin Burn-upstream. A higher proportion of water depth could be considered useable ranging from 4.1% to 46.5% at Burnhouse Burn-upstream and March Burn-upstream, respectively. Most notably, is the proportion of water depth that is considered 'not-useable'. The total streambed has 32.4% to 42.7% (Burnhouse Burn-downstream and Bin Burn-upstream) of the quadrats measured that are unavailable for use by adult trout. Of course, the proportion of dry habitat is the same as the other classifications. Similar trends are evident in the wetted portion of the streambeds. Suitable habitat ranges from 0% (Burnhouse Burn, up- and downstream) to 16.9% (Bin Burn upstream). Useable habitat ranges from 10.7 to 48.8% at Burnhouse Burn-upstream and March Burn-downstream, respectively. A high proportion of the wetted streambed is unavailable for use by adult trout. A range of 41.2% (March Burn-upstream) to 89.3% (Burnhouse Burn-upstream) of the wetted surface area is not useable by adult trout (Table 4.7).

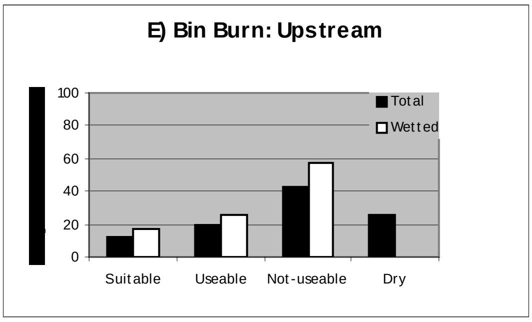
The area available with adequate depth (proportion of suitable and useable summed multiplied by the wetted area of the reach) for adult brown trout followed a very similar pattern as that seen for the other age classes; however, the amount available was noticeably less. Burnhouse Burn-upstream and downstream had the smallest amount of streambed with adequate spawning depth available (1.2 m^2 and 4.5 m^2, respectively), Bin Burn the largest [13.9 m^2, 22.0 m^2, and 14.5 m^2: upstream, downstream (pre-spate), downstream (post spate)] and March Burn had intermediate values for the upstream and downstream study reaches (7.1 m^2 and 15.6 m^2, respectively - Table 4.9).











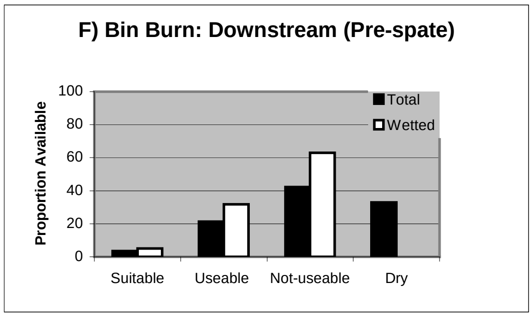
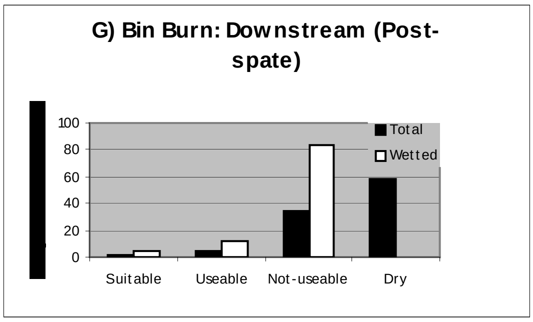


Figure 4.9 (A-G): Histograms of the proportion of water depth available to adult trout (> 20 cm) at all study sites classified as suitable, useable, not-suitable and dry for both the total and wetted streambed.



Lastly, the chi-squared analysis revealed that the proportions of habitat classified based on the suitability criteria are different at Bin Burn-downstream before and after the spate Table 4.6). The chi-squared value for the total streambed is $X^2 = 101.5$ (df = 3, p = 0.000) where as the wetted surface is $X^2 = 35.5$ (df = 2, p = 0.000). The percentage of the available water depths (suitable plus useable) were 24.8% and 37.0% for the total and wetted streambed before the spate and 6.8% and 16.1% after the spate. The total useable area with water depth available for use by adult trout dropped from 22.0 m^2 before the spate to 14.5 m^2 after the spate.

4.2.3 Habitat maps

A graphical representation of the habitat available for spawning (Figure 4.10), nursery (Figure 4.11), juvenile (Figure 4.12) and adult size classes (Figure 4.13), was produced for the downstream sites at March and Burnhouse Burn. The boundary between the stream channel and the streamside is clearly differentiated as the streamside segments are coloured black. The wetted area, or stream course within the stream channel, can be distinguished from the dry streambed as shaded and clear quadrats, respectively. Using these codes a basic description of each stream can be produced. The size of the study reach; the amount of habitat (suitable or otherwise), the proportion of the wetted and dry surface, and the total habitat available can all be presented in tabular or graphical form; as has been done in the earlier sections of this chapter. However, habitat maps provide additional information in that they can show specific locations of desirable or undesirable habitat; the shape of the channel in the streambed, and stream features such as deep areas (pools) and shallow areas (such as runs and riffles). This can be useful information for designing sampling protocols; habitat conservation projects, or help illuminate problem areas in projects that will affect stream hydrology.

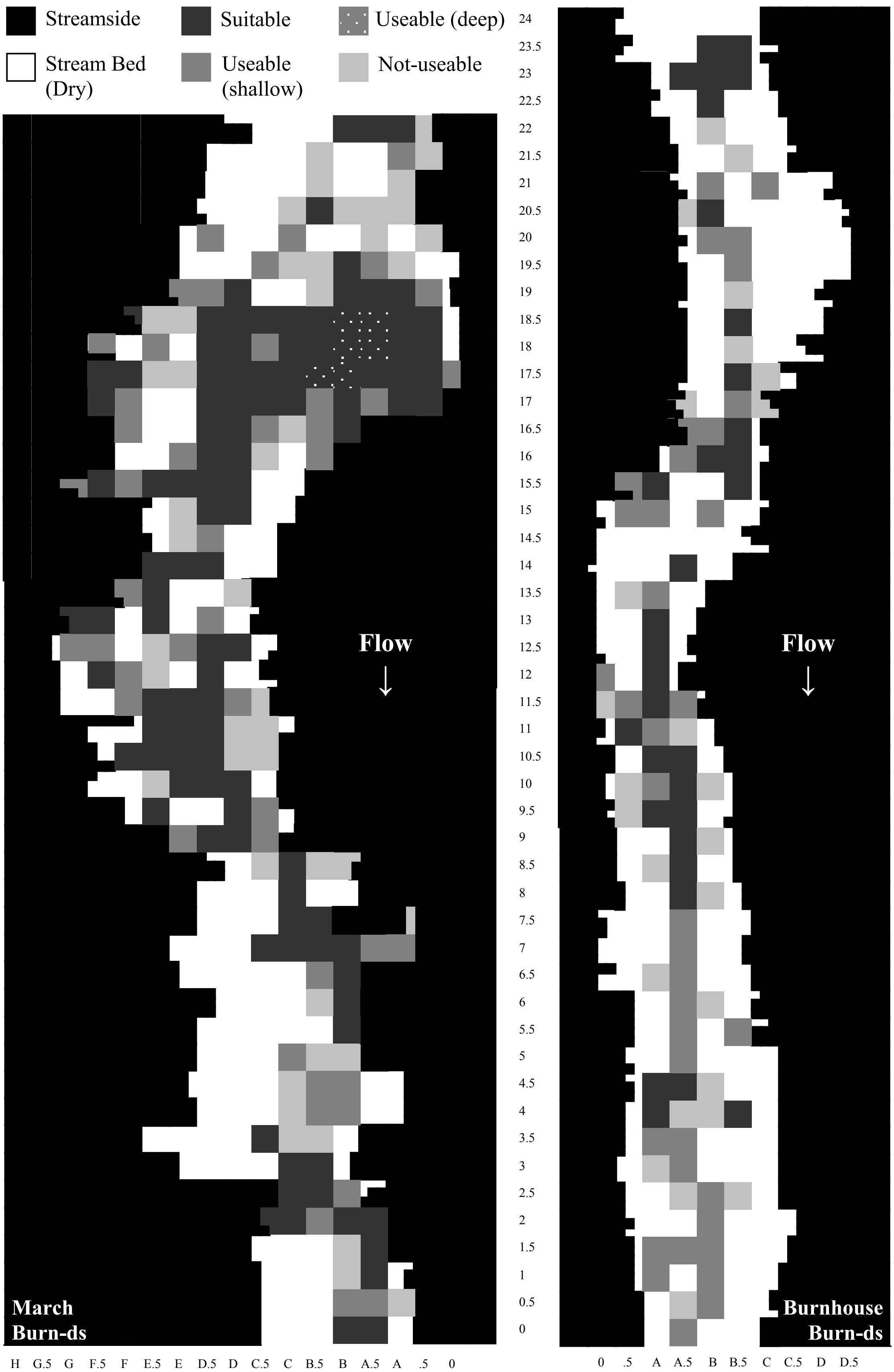


Figure 4.10: Water depth available for spawning brown trout (*Salmo trutta*) at the downstream sites of March (Q = 0.0341 m^3/sec) and Burnhouse Burns (Q = 0.0058 m^3/sec) based on assessed depth requirements.

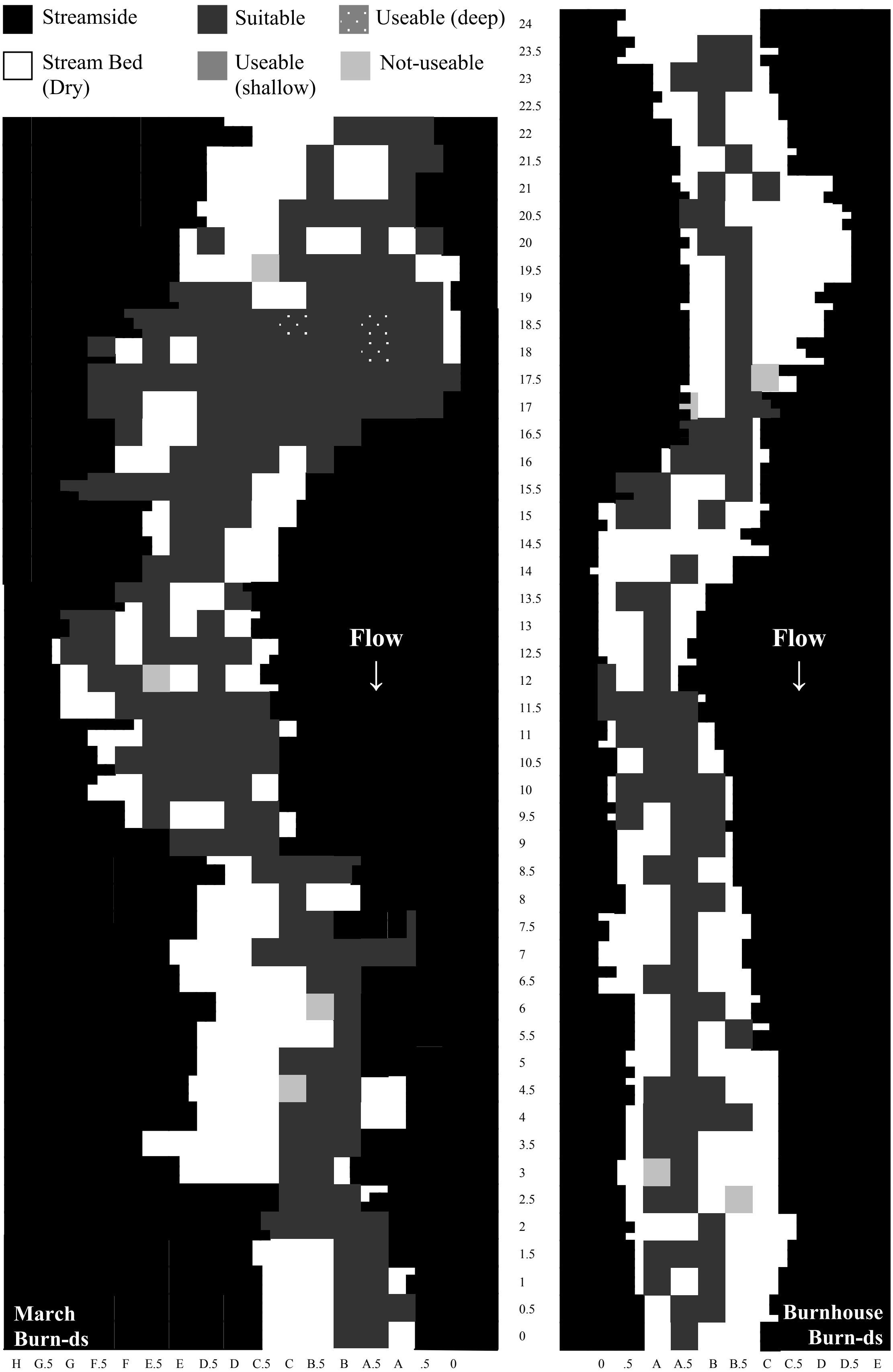


Figure 4.11: Water depth available for brown trout (*Salmo trutta*) fry (≤ 7 cm) at the downstream sites in March (Q = 0.0341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) base don assessed depth requirements.

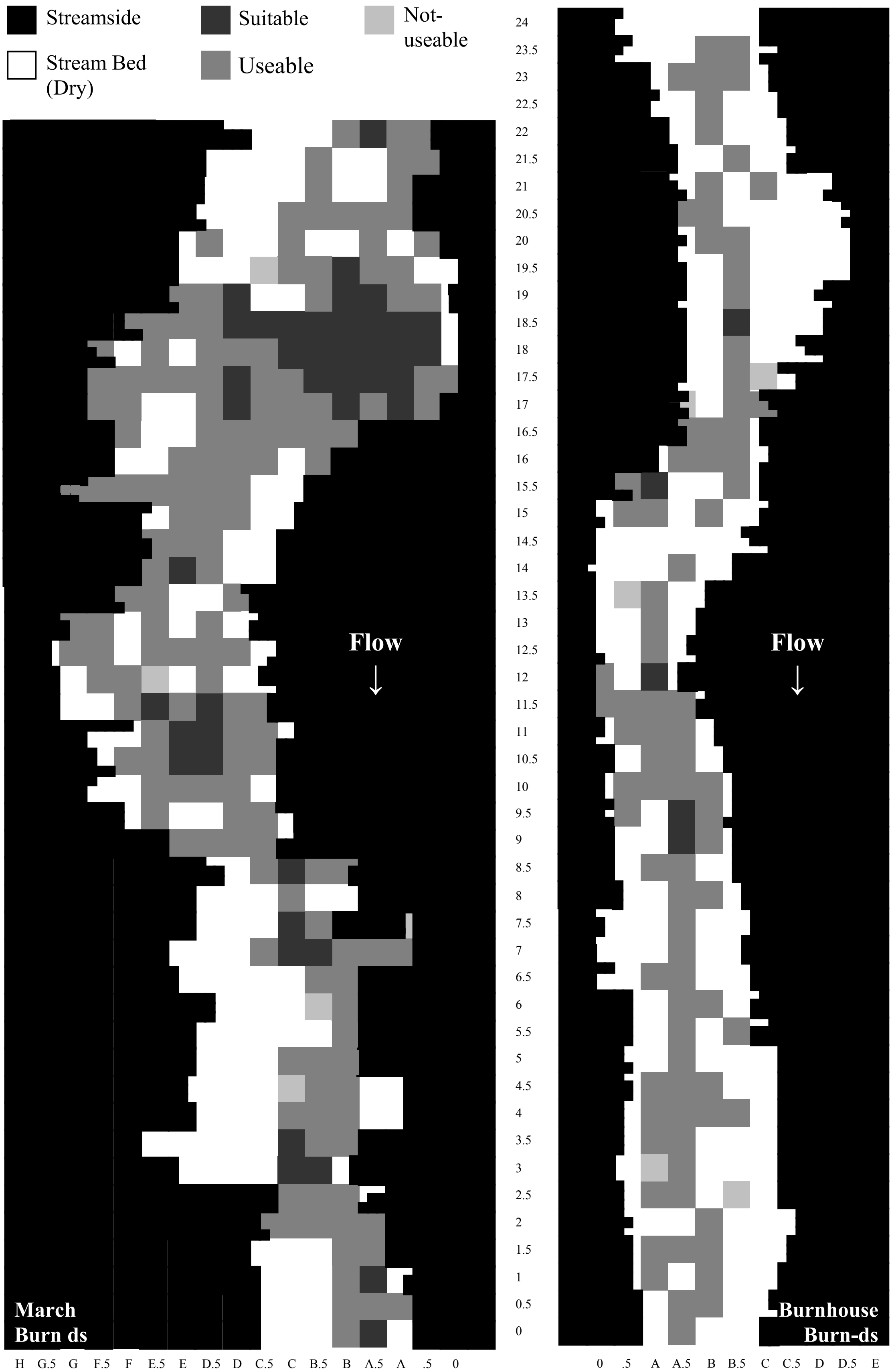


Figure 4.12: **Water depth available for juvenile brown trout (*Salmo trutta*) at the downstream sites in March (Q = 0.0341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on assessed depth requirements.**

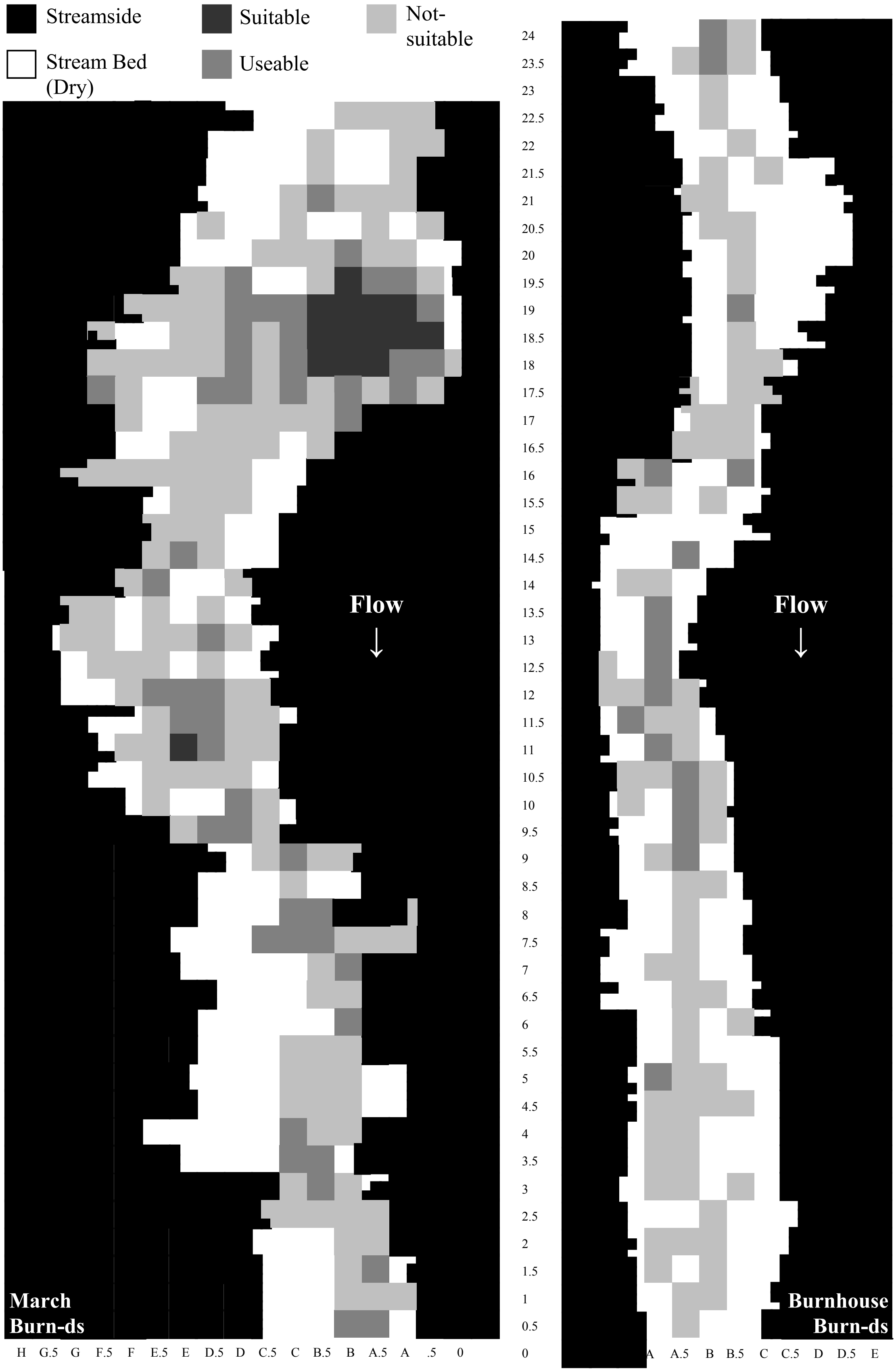


Figure 4.13: Water depths available for non-spawning adult resident brown trout (*Salmo trutta)* at the downstream sites in March (Q = 0.0341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on the depth requirements.

In Figure 4.10 to 4.13, we can see immediately that there is a pool at the upstream end of the March Burn reach, that the Burnhouse Burn site lacks similar features, and that the March Burn site has a larger streambed area. By studying the maps sequentially, it becomes evident that as fish get older there is a decreasing amount of habitat available. The pool in March Burn provides the best area for all age classes and provide the only refuge for older fish whereas useable habitat is more evenly distributed at the Burnhouse Burn site. We can see as we progress through the nursery, juvenile, and adult representations how the number of suitable quadrats decreased and the number of quadrats that are not-useable increases, thus confirming the observation noted in the tables and histograms. We can also note there is an abundance of water depths suitable for spawning.

4.2.4 Fish survey

In all three streams, brown trout (*Salmo trutta*) were the only species observed. The number of fish sampled and their size are reported in Table 4.10 (A-C). There were no fish found at the upstream sites so only the downstream-site data is presented. During the December survey it was noticed that both resident young and adult spawners were occupying Burnhouse and Bin Burn. The numbers and lengths of these two cohorts were reported separately. No spawning adults were observed in March Burn in December, however, a number of adult trout were observed downstream of an old gauging station that blocked their upstream migration. Some spawners must be able to bypass this obstruction, as some young resident fish were observed upstream in the study reach. However, the numbers of fish observed in March Burn were consistently less that those observed in Burnhouse Burn. For example, 57 young fish were

Table 4.10 (A-C): Brown trout observed at the downstream sites of March Burn (A), Burnhouse Burn (B) and Bin Burn (C) between September 2002 and June 2003.

A) March Burn

Sample date	Number of fish	Mean length TL* (cm)	Std. Dev. (cm)	Minimum length (cm)	Maximum length (cm)
25 09 2002	17	8.98	2.23	5.7	13.9
03 12 2002	10	9.07	1.37	7.1	10.4
18 03 2003	6	9.17	1.56	7.3	10.8
30 06 2003	3	12.03	0.80	11.2	12.8

* - Total length

B) Burnhouse Burn

Sample date	Number of fish	Mean length TL* (cm)	Std. Dev. (cm)	Minimum length (cm)	Maximum length (cm)
25 09 2002	32	5.98	1.55	4.2	12.0
03 12 2002 (fry)	8	5.95	0.68	5.2	7.0
03 12 2002 (spawning)	10	28.29	2.69	24.0	34.0
18 03 2003	14	6.52	0.77	5.4	7.9
30 06 2003	57	3.66	1.09	2.4	8.9

* - Total length

C) Bin Burn

Sample date	Number of fish	Mean length TL* (cm)	Std. Dev. (cm)	Minimum length (cm)	Maximum length (cm)
25 09 2002	9	9.58	1.55	6.8	11.7
03 12 2002 (fry)	4	12.25	4.63	8.5	19
03 12 2002 (spawning)	10	29.02	2.50	24.8	33
18 03 2003	10	9.51	1.32	7.6	11.6
30 06 2003	51	4.76	1.09	2.7	10.7

* - Total length

observed in Burnhouse Burn in June 2003 while only three were observed in March Burn.

The number of fish found in Bin Burn was also consistently lower than Burnhouse Burn; however, the fish survey occurred after the spate thus many resident fish might have been lost. It is assumed that the numbers found in Bin Burn are lower than what would be found in years without a large spate. The young fish in Burnhouse Burn were consistently smaller than those in Bin or Burnhouse Burn. For example, in September 2002 the mean length of trout was 5.98 cm in Burnhouse Burn, 8.98 cm in March Burn and 9.58 cm in Bin Burn. This trend held for all sampling periods with the exception of the June 2003 survey where the largest fish were found in March Burn. Burnhouse Burn always had the smallest fish. Clearly, the largest fish observed were the spawners observed in Burnhouse and Bin Burn in December, which ranged in size from 24 cm to 34 cm and 24.8 cm to 33 cm, respectively. The smallest resident fish was 2.4 cm observed in June 2003 in Burnhouse Burn and the largest was 13.9 cm; observed in September in March Burn.

4.3 Discussion

The statistical analysis indicates that relative proportion of suitability criteria differs from site to site; however, an examination of the general trends in habitat availability indicates that the same basic patterns are occurring at all sites. There seems to be adequate water depth available for spawning trout. Upwards of 30% [Bin Burn-downstream (post spate)] of the wetted area of the burns examined in the Carron Valley have depths that are suitable for spawning. The proportion of a streambed used by spawning trout estimated from stream survey varies but figures of 2.7% (Rubin et al. 2004) and 6% (Hewitt and Newcomb 2000) have been reported. These proportions of quadrats in the reaches examined in this study seems reasonable high; however, there utility to spawning trout must also be evaluated in terms of water velocity and substrate size.

From the perspective of water depth, these streams seem best suited as nursery areas, are less well suited as juvenile habitat, and do not appear to be well matched for adult residents. These trends are clearly noted in the histograms as the dominant age class shifts from suitable (nursery – Figure 4.8), to useable (juvenile – Figure 4.9), to not-useable (adult – Figure 4.10). The one exception is the upstream site at March Burn where using the adult criteria; slightly more habitat is designated as useable than not-useable (46.5% vs. 39.2% - Table 4.7). It is important to reiterate that the 'useable' designation in these tolerance profiles are at the limits of the habitat that brown trout have been observed occupying. Although the proportions of habitat that can be used by juvenile trout seems relatively high, the bulk of the depths reported are at the limits of what the age class are generally though to prefer or find necessary as refugia from older fish. As well, changes in habitat use seem to be size related; thus, the habitat available for the juvenile fish present would be more suited to the smaller fish in the age class (>7 to 20 cm in length).

The conclusion that the burns examined in the Carron Valley are spawning and nursery areas is not unexpected. It has been long known that brown trout, like other salmonids, return to flowing waters to spawn and that after emergence young fry occupy shallow regions of lotic systems (Elliott 1994, Armstrong et al. 2003, Klemetsen et al. 2003). As well, the brown trout size cohorts 'expected' to occupy these streams based on the tolerance profiles for each age class correspond to the fish that have been observed at these sites. With the exception of the adult spawners present in December all resident fish observed during all the sampling periods fell within the nursery or juvenile age classes. All mean lengths were below 7 cm (upper limit in the nursery category). All other mean lengths were below approximately 12 cm, a length towards the lower limit of the juvenile age classes (Table 4.10 A-C). As well, the length of fish sampled corresponded with the total area available. Among the reaches where fish were observed, the smallest individuals were found at the Burnhouse Burn-downstream site which has the least surface area classifies as suitable or useable (Table 4.9.) and the larger fish were found in the other two reaches which had larger available amount of acceptable habitat. Although the March Burn site had less useful surface area than Bin Burn it did have a large pool, which would provide stable deep habitat during a larger portion of the hydraulic cycle allowing for the establishment of larger fish. Again, this corresponds with what would be expected, as larger fish tend to be associated with deeper waters (Heggenes et al. 2002).

There were no fish present at any of the upstream sites. Table 4.9 indicates that wetted area available that is either suitable or useable is similar or larger than that available at the downstream Burnhouse Burn site, where fish were found. A forestry road that cut across all burns, which separated the upstream and downstream sites, may partially explain the absence of fish at these sites. The streams flowed through a small culvert at the junction with the forestry road and these culverts appeared laden with debris. The terrain became increasingly steep with elevation in this region resulting in numerous small and mediums sized waterfall may have provided further obstruction to the upstream distribution of fish.

The spate, debris removal and stream reconstruction seems to have benefited the downstream site; at least from the perspective of water depths for brown trout. Both total and wetted area increased at the post-spate site as did the total area that was considered suitable or useable for all life stages of trout expect adults. The total area acceptable for use by adult trout decreased presumably because a larger channel would distribute the available flow over a larger area resulting in shallower depth.

The habitat maps produced by marrying the tolerance profiles with the depth survey data are a good graphical illustration of the habitat available for each depth class. The rational for using a grid system has been discussed in Chapter 3 but it is important to mention that this method is limited as an assessment tool for many applications due to the time and expense needed to produce such a detailed reproduction of a stream reach. However, these maps do illustrate how tolerance profiles can be used to assess habitat availability in streams, which is why the method was chosen for this study. A profile of the stream can be generated that distinguishes the stream channel from the streambed, and features such as pools and the route of the watercourse within the streambed can be identified; however, the resolution of these maps is limited by the size of the quadrat used, which produces an unnatural blocky representation of the stream. As well, this method produces artefacts, such as discontinuities in the watercourse (Burnhouse Burn; rows 14.5 and 24 Figures 4.6 to 4.9), which does not represent the true nature of the burns investigated. The limited resolution of this technique may also produce inaccurate estimation of the variable being measure (water depth in this instance), which, again, is a result of the scale or size of the quadrate being used.

The application of the tolerance profiles to the streams reaches studies for each age class seems to have resulted in a realistic assessment of the of the water depth available to brown trout. However, the boundaries between useable and suitable habitat and those that discriminate between size-classes are not intended to be final or unmoveable. If more information is obtained about the life history parameters and microhabitat used by *Salmo trutta* in future studies adjustment can be made to these profiles. The data obtained and the interpretation of brown trout microhabitat used are known to be influenced by the methods used to collect habitat-use data, the scale at which the information was collected, and by a host of other biotic and abiotic factors including habitat availability and inter specific competition (Heggenes 1988b). An attempt was made to be as inclusive as possible in terms of the studies available regarding microhabitat use; however, the set is not complete. For example, the bulk of the investigations surveyed came from Western Europe, North America, and New Zealand. Much less information concerning water depth utilization is available from other parts of this species current range including South America, Africa, Australia, central Asia, and parts of its natural range including eastern Europe, south eastern Asia, and North Africa. Thus, a survey of habitat utilization by brown trout in a wider range of its habitat should be considered for future studies.

Reassuringly, the profiles constructed, when applied to the depths observed in the study streams, do produce an assessment; presented as tables, graphs, and habitat maps that are complemented by what is expected from the information that is know about *Salmo trutta* life history and is reinforced by the fish surveys undertaken. The useable or suitable ranges within each profile may shift, narrow or widen, as more information about trout water depth choice becomes available but the conclusions from the application of these altered profiles to streams such as March Burn or Burnhouse Burn would not be expected to change. In other words, it is unlikely that revised tolerance profiles would conclude that either of these streams, based on the data collected in late summer 2002, could be used predominantly by large resident brown trout. Further reassurance comes from a comparison of the tolerance profiles themselves. Figure 4.5 revealed that the tolerance profiles shift to deeper habitat for increasingly large size-classes and that the 'width' or range of habitat available also increased. This is in agreement with what is known about their biology. As previously discussed, it has commonly been observed that brown trout use increasingly deep water as their size increases (Bohlin 1977, Kennedy and Strange 1982), that all age classes prefer deep waters (Wesche et al. 1987, Heggenes 1988a), and that smaller sub-dominate fish are excluded from these preferred habitats (Kennedy and Strange 1986, Bremset and Berg 1999, Heggenes et al. 1999). Thus, the depth at which these fish are found and the range of depths available increases with size as illustrated when the tolerance profiles are compared (Figure 4.5).

Interestingly, the summary of the literature concerning the use of water depth by brown trout and the construction of these profiles has highlighted a somewhat unexpected conclusion. Namely, that the shallower areas within streams may be critical habitat in that they act as refugia by the smaller cohorts and the smaller fish within a cohort. In other studies, brown trout preference for deeper habitat has lead to the conclusion that the availability of deep water limits the abundance of large trout in streams (Heggenes et al. 1999). Similarly, this species aggressive territoriality and the risk of predation from larger fish in general results in the displacement of smaller individuals to marginal less energetically favourable habitats. It also suggests that the availability of shallow water may limit the abundance of small or newly emerged brown trout. Shallow areas within streams inaccessible to older cohorts and larger fish of co-occurring species, particularly when a full complement of trout size classes is present, seems necessary in order to ensure that enough young of the year trout survive in order to maintain a sustainable population within a given watercourse. The conclusion that shallow habitat is critical habitat is speculative but warrants further investigation.

The problems mentioned above and others associated with summarizing data from disparate sources, such as the inability to summarize the available information quantitatively, are not unique to the development of tolerance profiles for water depth or to this study (Morantz et al. 1987, Heggenes et al. 1999). Similar issues will be encountered during the construction of tolerance profiles in subsequent chapters. Further, the choice of microhabitat used by brown trout is influenced not only by depth but also by other cues both biotic and abiotic (Moyle and Baltz 1985, Fausch and White 1986, Orth 1987). From a physical perspective depth, accompanied by stream flow, and substrate composition are also important considerations for habitat selection by *Salmo trutta*. A thorough assessment of the habitat available for *Salmo trutta* in March Burn and Burnhouse Burn, as well as thorough discussion of the limitations encountered during the construction of the tolerance profiles is not appropriate at this time. Chapter 7 integrates the information and conclusions obtained from the construction and application of tolerance profiles for water depth, stream velocity and substrate composition to these streams. The successes and difficulties encountered with are further elaborated therein.

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5.0 Velocity

5.1 Introduction

This chapter will focus on the development and application of tolerance profiles for water velocity just as the previous chapter focused on water depth. Lotic systems are, by definition, fluvial environments. Thus, water velocity is one of the most important physical parameters that influences the choice of microhabitat utilized by organisms living in riverine ecosystems (Shirvell and Dungey 1983, Bremset and Berg 1999, Heggenes et al. 1999). The objectives of this chapter are 1) the development of size-class specific tolerance profiles for each water velocity and 2) the application of these profiles in order to assess the habitat available at the six sites (March, Burnhouse, and Bin Burn; up and downstream) and as assessment of the Burnhouse Burn downstream site before and after a disruptive spate. This analysis will be conducted in order to determine the effectiveness of the technique and to demonstrate how the methodology can be used to assess the water velocity component of the physical habitat available to brown trout in running waters. The habitat expected based of the application of the tolerance profiles will be compared against the results of the fish survey to help determine validity of the technique. The remainder of the introduction will outline the requirements that brown trout have for water velocity in a stream.

5.1.1 Instream water velocity requirements: overview

The velocity of water is an essential component of brown trout habitat selection and is one of the primary factors influencing riverine ecosystems. Not only does moving water physically support individual trout, which are morphologically adapted to flowing environments, it also provides support for ecosystem functions such as providing a means for invertebrate drift, oxygenation of redds, and the selection and modification of sediment (Haury et al. 1999). Velocity has been shown to affect the choice of microhabitat used by brown trout (Shirvell and Dungey 1983) and the criteria used in this selection process differs with age (size) and life stage of the fish (Heggenes 1988c, Haury et al. 1999). However, regardless of age or life stage energy expenditure is always an important consideration and trout would be expected to maximize the benefits of any microhabitat choice against the cost of maintaining position in their moving environments (Bachman 1984, Fausch 1984). This cost-benefit relationship begins at the earliest life stages of this species.

Brown trout usually spawn in running water (Raleigh et al. 1986). A female spawning adult selects a place where there is clean flowing water and silt free gravel (Crisp 1993). Moving water is important both to help dislodge bed material during the construction of the redd and to provide oxygen and carry away metabolites produced by the buried eggs which could prove toxic (Crisp and Carling 1989). Traditionally, potential spawning sites are thought to be located at sites characterized by upwelling of water through the gravel or where water currents flow downward into the gravel (Benson 1953). The ideal site for spawning has been described as the tail end of a pool as it merges into a riffle, although, this has been contested (Ottaway et al. 1981). Haury and his colleagues (1999), in a review of brown trout habitat usage, report workers who observed redds in convex banks of streams, hollowed out banks, in close proximity to submerged vegetation, trunks or overhanging branches, and in still and deep areas. Haury and colleagues (1999) report Huet (1962) suggesting that spawning is not possible with zero current while Crisp (1993) contends that trout may try to spawn in still water; however, they show a preference for moving water and that the upper limit is related to the size of the female spawner (less than two female body lengths per second). This positive body size velocity relationship in redd site selection is thought to result from larger spawning females being able to withstand higher velocities (Witzel and MacCrimmon 1983). While reasonably swift velocities (> 15 cm/sec) are preferred for redd construction and spawning Crisp (1993) suggests that deeper slower flowing waters may be required as places to wait before spawning, resting places between spawning episodes, and as cover from predation.

After emergence, young trout reside in shallow areas, often located along riverbanks with relatively low velocities (Shuck 1945, Lindroth 1955, Roussel and Bardonnet 1999). There is evidence of competitive exclusion or segregation as 0+ trout are dominated and expelled by 1+ trout from shallow riffles (Heggenes 1988c). As well, an upper limit to velocity for young trout exists as fry are susceptible to downstream displacement (Ottaway and Forrest 1983). There is a change in the preference for water velocity with changes in fish size (Bohlin 1978). Some workers reporting that as trout grow older and larger they prefer deeper more slowly flowing parts of the stream (Heggenes 2002). However, Greenberg and his colleagues (1986) in Sweden show that parr use faster water velocities than young of the year. Adults are often reported in relatively slow moving portions of the stream (Bohlin 1977, Egglishaw and Shackley 1982, Raleigh et al. 1986, Heggenes 1988b) nevertheless Karlström (1977) observed adult trout in turbulent waters and Cunjack and Power (1986) report that mature individuals occupy deeper and faster flowing areas than juveniles. Interestingly in a Finnish study (Mäki-Petäys et al. 1997), trout showed no selection for specific water velocities but used them in proportion to their availability. As well, they found only slight, non-significant, differences in the mean velocities used by different size classes.

At lower water temperatures (autumn and winter) brown trout utilize habitats with slower water velocities relative to those chosen in the summer (Karlstrom 1977, Cunjak and Power 1986)

In fluvial ecosystems there is an energetic cost for an organism associated with activities such as holding position or maintaining a feeding territory (Fausch 1984). Thus, it would be expected that chosen microhabitats would be characterized by low water velocities (Heggenes 1988c) which would maximize their energy saving value (Bachman 1984). However, salmonids are primarily drift feeders and they rely on the current to deliver food (Everest and Chapman 1972). Therefore, drift-feeding species such as brown trout are thought to occupy low velocity focal positions adjacent to high velocity currents that supply drift prey at a relatively high rate, thus, maximizing food intake and minimizing energy expenditure (Morantz et al. 1987). These sorts of water velocity differences have been proposed as a criterion to assess position choice rather than velocities at the fish's position alone (Fausch and White 1981).

5.2 Results

5.2.1 Tolerance profiles

5.2.1.1 Spawning water velocity

The literature that reports on the water velocity used by spawning brown trout comes from a diverse range of the worldwide distribution of this species. Included are studies from Europe (1981, Nihouarn 1983, Fragnoud 1987, Heggberget et al. 1988, Schneider 2000), North America (Smith 1973, Witzel and MacCrimmon 1983, Grost et al. 1990, Essington et al. 1998, Pender and Kwak 2002), and New Zealand (Shirvell and Dungey

1983, Scott and Irving 2000). All of the studies examined sympatric populations of brown trout and the New Zealand (Shirvell and Dungey 1983, Scott and Irving 2000) and Canadian studies (Witzel and MacCrimmon 1983) studied both sympatric and allopatric populations. All of these investigations occurred in natural streams with the exception of the Swiss study which examined spawning in a canal (Schneider 2000). To be as comprehensive as possible, some of the values reported for spawning velocity, which could not be obtained otherwise, have been extracted from a literature review by Haury and his colleagues (1999); including the values reported by Fragnoud (1987) and Nihouarn (1983). Limited information is available about the specific characteristics of these investigations. The literature used in this synopsis is summarized in Figure 5.1 and reported in more detail in Table 5.1.

Salmonids typically spawn in flowing waters. Determining a preference range of spawning velocity used by brown trout is confused somewhat because velocity is measured at different positions in the water column. Hayes (1994) commented that studies conducted with a biological focus tend to use snout (or focal point) velocity and investigations with a hydraulic perspective tend to use 0.6 depth. Further, snout velocities are not measured at a consistent distance above the substrate or redd surface [e.g. distances above redd include: 2 cm (Shirvell and Dungey 1983); 4.5 cm (Ottaway et al. 1981); and 12 cm (Smith 1973)]. This literature summary separates measures of mean column velocity (0.6 depth) and snout velocity. Both sets of data and the position in the water column where the measure was taken are listed. The exact distance from the riverbed that velocity was measured varies from study to study. The measure used in each study included in the literature survey is outlined in Table 5.1.

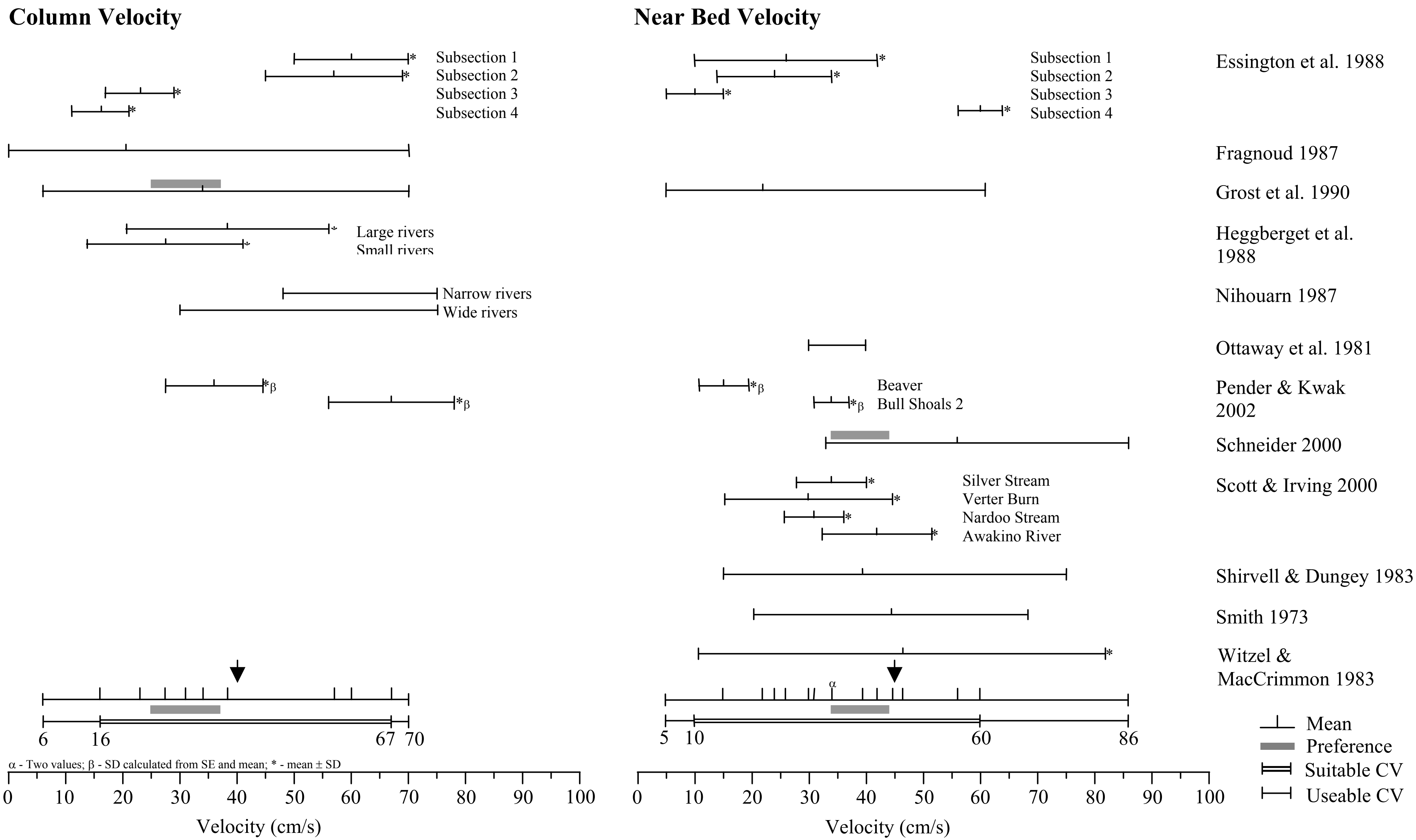


Figure 5.1: Procedure used to define the range of stream water velocities that can be used by brown trout (*Salmo trutta*) to spawn.

Table 5.1: Literature used to define the range of velocities that can be used by brown trout (*Salmo trutta*) to spawn.

Source	Water column velocity (WC) † *cm/s*	Near bed velocity (NBV) † *cm/s*	Location of measure	Number of redds or spawners observed	Fish size mean [range] *cm*	Study location	Allo-sympatric populations	Other species	Natural-artificial stream	Notes
Essington et al. 1998 Subsection 1 Subsection 2 Subsection 3 Subsection 4	60±10 57±12 23±6 16±5	26±16 24±10 10±5 60±4	WC – 0.6 depth NBV – 3 cm off substrate	Brook: 36 Brown: 108	-	USA (Minnisota)	Sympatric	Brook trout	Natural	
Fragnoud 1987	20.5 [0-70]	-	WC – average at the redd	-	-	France (eastern)	-	-	-	Reported by Haury et al. 1999
Grost et al. 1990	34±15 [6-70]	22±11 [5-61]	WC – 0.6 depth NBV- on substrate	80	20-40	USA (Wyoming)	Sympatric	Brook trout	Natural	WC and NBV measured at the pit front (prefer 24.5-36.6) Avoid 0-12 cm/s
Heggberget et al 1988 large rivers small river	38.3±17.7 27.4±13.6	-	WC – mean 5 cm above substrate and 5 cm below the water surface	36 125	0.5 kg (Alta) 0.9 kg (Gaula) 2.1 kg (Driva) 0.8 kg (Eira)	Norway	Sympatric	Atlantic Salmon	Natural	Large - Pooled data from Rivers Alta, Gaula, Driva. Small - Data from River Eira
Nihouarn 1983 width: 1.5-3 m (a) width: 7–20 m (b)	30-75 48-75	-	WC - average at the redd	-	-	France (Brittany)	-	-	-	Reported by Haury et al. 1999; a) 58% & b) 75% between these values
Ottaway et al. 1981	-	30-40	NBV- 4.5 cm above substrate	272*	-	England (north)	Sympatric*	-	Natrural	* - ≅4% of redds Atlantic salmon
Pender & Kwak 2002 Beaver Bull Shoals 2	36 (6) $^{\alpha}$ 67 (9) $^{\alpha}$	15 (3) $^{\alpha}$ 34 (5) $^{\alpha}$	-	8 15	37.8 38.5	USA (Missouri & Arkansas)	Sympatric	-	Natural	α - 2 standard errors of the mean
Schneider 2000	-	56±10 [33-86]	NBV- 10 cm off substrate	30	-	Switzerland	Sympatric	Grayling and others	Artificial (canal)	Avoided water < 34 cm/s & > 84 cm/s. Preferred 34-44 cm/s

Table 5.1: Continued.

Source	Water column velocity (WC) † *cm/s*	Near bed velocity (NBV) † *cm/s*	Location of measure	Number of redds or spawners observed	Fish size mean [range] *cm*	Study location	Allo-sympatric populations	Other species	Natural-artificial stream	Notes
Scott & Irving 2000 Silver Stream Verter Burn Nardoo Stream Awakino River	-	34±6.1 (1) 30±14.8 (2) 31±5.2 (1) 42±9.6 (2)	NBV- as close to the bottom as possible	37 14 27 6	-	New Zealand	Allopatric & Sympatric	Rainbow Trout	Natural	Standard Deviation calculated from sample size and standard error (McGhee 1985)
Shirvel & Dungey 1983	-	39.4±11.0 [15-75]	NBV – 2 cm above highest point on redd	140	30-60	New Zealand	Allopatric Sympatric	Rainbow trout	Natural	Three replicate measures of velocity taken
Smith 1973	-	44.5±54.1 [20.4 - 68.3]	NBV – 12 cm above substrate	115	-	USA (Oregon)	-	-	Natural	-
Witzel & MacCrimmon 1983	-	46.5±(1.4) [10.8-80.2]	NBV – 10 cm above redd surface	112	-	Canada (Ontario)	Allopatric & Sympatric	Brook Trout	Natural	1 to 3 positions above the redd surface

† mean±SD (SE) (cm/s^2)[range]

The minimum velocity available from the literature for mean column velocity used by spawning trout was reported by Fragnoud (1987) who found individuals spawning in water that was not moving (0 cm/s). In the only other reported range, Grost and his colleagues (1990) observed trout spawning in mean water column velocities as low as 6 cm/s. It is important to note that the number of values that can be used to determine both the upper and lower limits of microhabitat choice is incomplete because the full range of observation are generally not reported. However, the minimum value expressed by Fragnoud (1987) is supported by Crisp (1993) who suggests that trout and salmon will occasionally attempt to spawn in still water. This is contradicted by comments by Huet (1962) who state that spawning cannot occur in areas with no current. Fortunately, there are five datasets that report ranges of velocity taken near the substrate (i.e. snout or focal point velocities) and minimum measures vary from 5 to 33 cm/s. Thus, the lowest snout velocity reported in this summary is 5 cm/s. The maximum water column velocity reported by both Grost and is colleagues (1990) and Fragnoud (1987) was 70 cm/s and the maximum velocity for measures taken near the substrate range from 61 cm/s (Grost et al. 1990) to 86 cm/s (Schneider 2000). Thus the maximum velocity reported in this survey for spawning brown trout measured near the substrate was 86 cm/s.

The literature review yielded nine mean column velocity ranging from 16 cm/s (Essington et al. 1998) to 67 cm/s (Pender and Kwak 2002). There are 15 mean column values reported for measures of the water velocity near the riverbed (or redd) selected by spawning brown trout. These vary from a 10 cm/s to 60 cm/s both reported by Essington and his colleagues (1998) in an study conducted in the United States. These mean values are used to outline the range of suitable velocities that are used by *S. Trutta* based on observation from its worldwide distribution. Thus, trout seem to prefer water column velocities that range from 16 to 67 cm/s and near bed velocities that range from 10 to 60 cm/s.

Other investigators have summarized a range of velocities that can be used by brown trout to spawn. Most notably Raleigh and his colleagues (1986) summarizes velocities used by trout from six studies most of which were conducted in the USA. Based on this review, these workers recommend a tolerance range of 15-90 cm/s with optimal spawning velocities of 40 to 70 cm/s. However, it is unclear in this summary whether the velocities that this recommendation is based was taken from the water column, the substrate or both sampling methods. Crisp (1996) outlined the environmental requirements of salmon and trout and suggests that trout prefer velocities, measure at 0.6 depth, from 15 cm/s to a value less than 2 female body length per second. This suggests that the maximum velocity that can be used by spawning salmonids is related to body size (Ottaway et al. 1981, Witzel and MacCrimmon 1983, Rubin et al. 2004). Adult brown trout range in length from 40 to 60 cm (Scott and Crossman 1973); thus, in some population large spawning females could use velocities as high as 120 cm/s.

Although not exactly the same, the useable and suitable ranges are similar to those proposed in other literature summaries. This work differs from these studies in that a useable and suitable range (similar to Raleigh and colleagues 'tolerance' and 'optimal') is defined for both water column and substrate measures of velocity and that studies from a broader range of the species worldwide distribution are utilized. The tolerance profile for water velocity measured at 0.6 depth will include a useable range from 6 to 75 cm/s and a suitable range from 16 to 67 cm/s. The tolerance profile for substrate (snout or focal point velocities) will have a 'useable' range from 5 to 86 cm/s and a 'suitable' range from 10 to 60 cm/s. These tolerance profiles will be used for the remainder of this study.

5.2.1.2 Nursery water velocity (fish length ≤ 7 cm)

The data that summarizes the velocities used by young of the year brown trout fry is from a much narrower range of the worldwide distribution of this species when compared to previous surveys. The data available comes primarily from the United States (Raleigh et al. 1986, Harris et al. 1992, Kocik and Taylor 1996, LaVoie and Hubert 1996, Pender and Kwak 2002) and northern Europe (Greenberg et al. 1996, Mäki-Petäys et al. 1997, Bremset and Berg 1999, Heggenes et al. 2002). All of these studies involved sympatric populations of *S. trutta.* Like the spawning microhabitat summary the methods used to record water velocity was measured in two general ways. This included mean water column velocity and near bed (snout or focal point) velocity. The literature used in this synopsis is summarized in Figure 5.2 and reported in more detail in Table 5.2.

Small fry were observed by LaVoie and Hubert (1996) in backwaters of Douglas Creek, (Wyoming, USA), with no velocity and in margins of riffles and pools with very little moving water. Similar results were observed earlier by Harris and colleagues (1992). Heggenes (2002) who measured snout velocity in a study in southwest England observed trout fry using velocity as low as 0 cm/s. As well Pender and Kwak (2002)

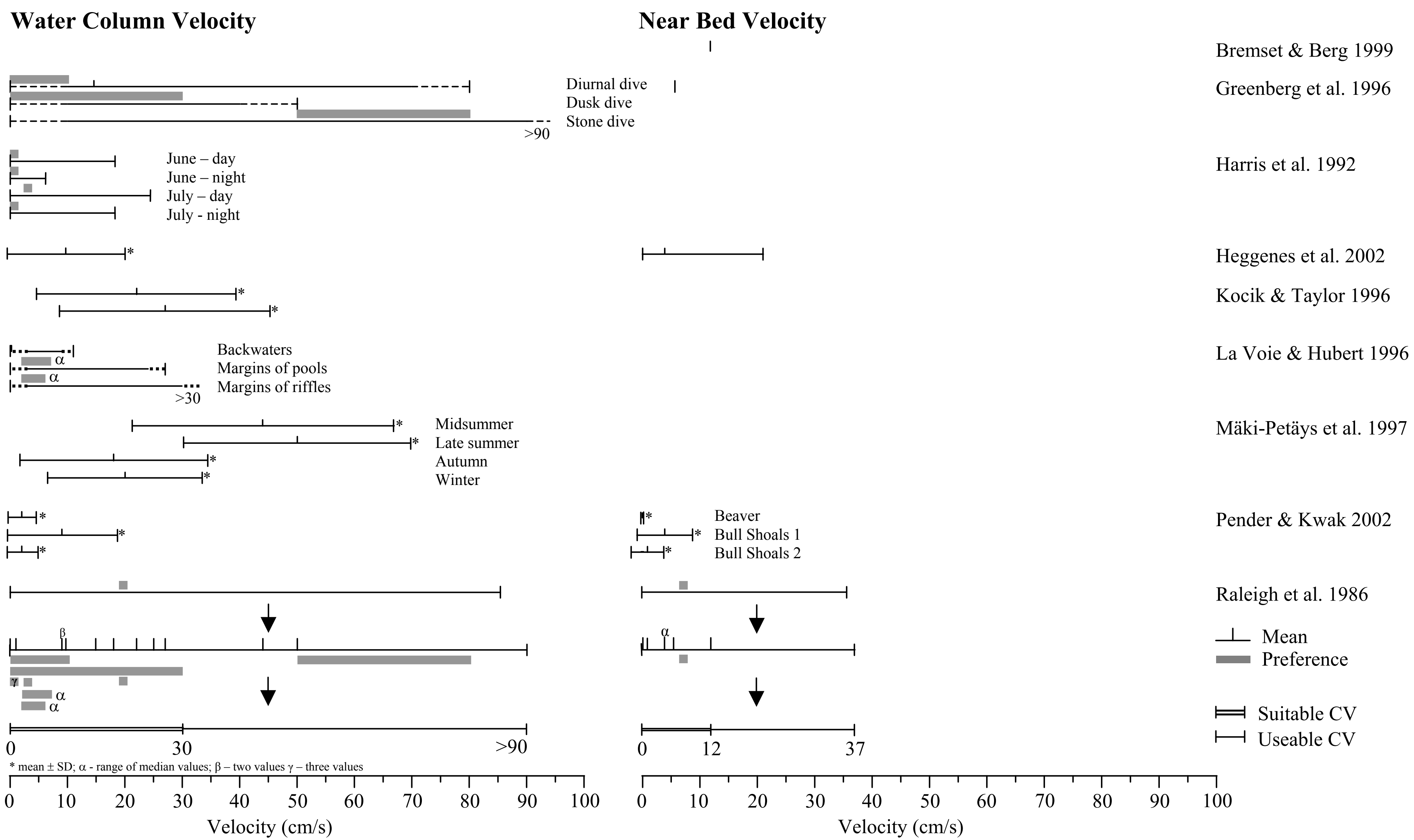


Figure 5.2: Procedure used to define the range of stream water velocities that can be used by brown trout (*Salmo trutta*) fry (length ≤ 7 cm).

Table 5.2: Literature used to define the range of stream water velocities that can be used by brown trout (*Salmo trutta*) fry (≤ 7 cm).

Source	Water column velocity (WC) † *cm/s*	Near bed velocity (NBV) † *cm/s*	Location of measure	Number of fish sampled	Preferred velocity FH/SC *cm/s*	Category range (FH/SC)	Category size (FH/SC) *cm*	Fish size mean (range) *cm*	Study Location	Allo – sympatric pop's	Other species	Survey method
Bremset and Berg 1999	-	12	NBV – focal velocity	22	-	-	-	(3.1-6.7)	Norway (central)	Sympatric	Atlantic salmon	Underwater (Pools)
Greenberg et al. 1996												
Diurnal dive	14.9	5.7	WC – 0.6 depth NBV – snout position	88	< 10^{β}	0-10 – 70-80	10	(2.5-6)	Sweden (Norrland)	Sympatric	Grayling	Underwater
Dusk dive	-	-		26	$0\text{-}30^{\beta}$	0-10 – 40-50	10					
Stone dives	-	-		121	$50\text{-}80^{\beta}$	0-10 – >90	10					
Harris et al. 1992												
June – day	-	-	-	25	0	0-18.3	-	2.6	USA (Wyoming)	Sympatric	Brook trout	Surface
June – night				25	0	0-6.1						
July – day				25	3.1	0-24.4		3.9				
July - night				25	0	0-18.3						
Heggenes et al 2002	9.7±10.3	4.0±4.2 [0-21]	WC – 0.6 depth NBV – snout position	273	-	-	-	< 7	England (southwest)	Sympatric	Atlantic salmon	Underwater
Kocik and Taylor 1996												
Spring	22 (2.13)	-	WC- 0.6 depth	65	-	-	-	5.3 (June) 8.8 (Oct) 9.3 (Feb)	USA (Michigan)	Sympatric	Rainbow trout	Underwater
Summer-Fall	27 (0.94)			382								
LaVoie and Hubert 1996												
Backwaters	0^{χ}	-	-	22	-	<3 to 9-11	3	5.6^{χ}	USA (Wyoming)	Sympatric	Brook charr	Surface; electro-fishing
Margins of pools	$2\text{-}7^{\chi}$			132		<3 to 24-26	3					
Margins of riffles	$2\text{-}6^{\chi}$			170		<3 to >30	3					
Mäki-Petäys et al. 1997												
Midsummer	44±(4.0)	-	CV – 0.6 depth	33	-	-	-	4-9	Finland (northern)	Sympatric	Grayling	Electro-fishing
Late Summer	50±(3.0)			43								
Autumn	18±(3.2)			26								
Winter	25±(3.0)			20								

Table 5.2: Continued.

Source	Water column velocity (WC) † *cm/s*	Near bed velocity (NBV) † *cm/s*	Location of measure	Number of fish sampled	Preferred velocity FH/SC *cm/s*	Category range (FH/SC)	Category size (FH/SC) *cm*	Fish size mean (range) *cm*	Study Location	Allo – sympatric pop's	Other species	Survey method
Pender and Kwak 2002 Beaver Bull Shoals 1 Bull Shoals 2	 2 (1) $^{\alpha}$ 9 (2) $^{\alpha}$ 2 (1) $^{\alpha}$	 0.1 (0.1) $^{\alpha}$ 4 (1) $^{\alpha}$ 1 (1) $^{\alpha}$	WC – mean velocity NBV – bottom velocity	 24 96 34	-	-	-	-	USA (Missouri, Arkansas)	Sympatric	Sculpins	Electro-fishing
Raleigh et al. 1986 $^{\delta}$	-	-	WC – 0.6 depth NBV – snout velocity	> 190	19.8 (WC) 7.3 (NBV)	0-85.3 (WC) 0-36.6 (NBV)	-	< 5.5	USA (Utah)	-	Rainbow trout	Diving

†mean±SD (SE) [range]; α - 2 standard errors of the mean; β - mean velocity; χ - median values for August and September; δ - summarized from data collected by Gosse et al. (1977) and Gosse (1981)

who recorded the bottom velocity at trout positions also recorded use of velocities which ranged from a mean of 0.1 to 4 cm/s at the Beaver and Bull Shoals 1 sites respectively. The fastest mean water column velocity reported in this survey was recorded by Greeneberg and his colleagues (1996) while examining fry sheltering under stones. They report mean water column velocities greater than 90 cm/s. Correspondingly, Greenberg and his colleagues (1996) report a preferred velocity for stone sheltered fry of 50 to 80 cm/s. The fastest near bed velocity measured was reported by Raleigh and his colleagues (1986) who report some young trout using snout velocities of approximately 37 cm/s.

Although mean water column velocities have been reported as high as 90 cm/s the bulk of the measures were generally below 30 cm/s (Greenberg et al. 1996, Kocik and Taylor 1996). There were 11 measures of mean values and nine preference ranges for water column velocities between 0 and 30 cm/s in the literatures surveyed. There are two exceptions. The first is the mid and late summer observations by Mäki-Petäys and his colleagues (1997) in northern Finland who report mean water column velocities of 44 and 50 cm/s, respectively. The available velocities were higher in mid and late summer (mean 44 and 50 cm/s respectively) compared to those available in autumn and winter (18 and 25 cm/s respectively), which may account for the high water velocities observed (see Table 1 in Mäki-Petäys et al. 1996). It is also important to mention that the Finnish workers recorded trout velocity at a position towards the midpoint of the water column (0.6 depth). Velocities at this depth are higher than those that may be experienced by the fish themselves who tend to shelter close to the substrate during periods of high flow (Rincon and Lobon-Cervia 1993, Quinn and Kwak 2000). Similarly, the preference range reported for stone sheltering fry in Sweden (Greenberg et al. 1996), recorded in the water column, would exceed the water velocities utilized by the fish sheltering under stones. All six measures of central tenancy for near bed measures of velocities used by young trout fry were between 0 and 12 cm/s.

Based on the summary of the literature available the suitable range of mean water column water velocity for newly emerged trout fry ([7 cm in length) will be from 0 to 30 cm/s. The useable range will be from 30 to 90 cm/s. The tolerance profile for snout velocities are much lower with a suitable range varying from 0 to 12 cm/s and a useable range varying from 12 to 37 cm/s.

5.2.1.3 Juvenile water velocity (fish length: >7 to 20 cm)

Research that has investigated brown trout's occurrence in relation to velocity comes from North America (Cunjak and Power 1986, Raleigh et al. 1986, Shuler and Nehring 1993, Shuler et al. 1994), northern (Heggenes 1988a, Heggenes and Saltveit 1990, Greenberg et al. 1996, Mäki-Petäys et al. 1997, Bremset and Berg 1999, Mäki-Petäys et al. 2000, Heggenes and Dokk 2001, Heggenes et al. 2002), central (Roussel et al. 1999), and southern Europe (Rincon and Lobon-Cervia 1993, Vismara et al. 2001). All of these studies with the exception of those conducted by Heggenes and his colleagues (1988), Rincón and Lobón-Cerviá (1993) and Roussel and colleagues (1999) involved sympatric populations of *S. trutta*. In the sympatric populations brown trout co-occupied the study sites with other salmonids including Arctic charr (Heggenes and Dokk 2001), Atlantic salmon (Heggenes and Saltveit 1990, Bremset and Berg 1999, Heggenes and Dokk 2001, Heggenes et al. 2002), brook trout (Cunjak and Power 1986), grayling (Greenberg et al. 1996, Mäki-Petäys et al. 1997, Vismara et al. 2001), and rainbow trout (Shuler and Nehring 1993, Shuler et al. 1994). The bulk of these investigations looked at microhabitat selection using underwater observations. However, the work by Heggenes (1988), Mäki-Petäys and his colleagues (1997), and Vismara and his colleagues (2001) employed electro-fishing techniques. Some of the sampling conducted by Shuler and his colleagues (1994) was completed using an angling protocol. The literature used in this synopsis is summarized in Figure 5.3 & 5.4 and reported in more detail in Table 5.3.

Like the nursery velocity requirements, the microhabitat used by juvenile brown trout has been divided by sampling methodology and presented in two figures. Figure 5.3 summarizes velocity measured in the water column and Figure 5.4 summarizes water velocities measured closer to the substrate (snout or focal point velocities). Juvenile trout seem to prefer lower velocities. The lowest mean water column velocity reported was 12.9 cm/s (Heggenes et al. 2002), Bremset and Berg (1999) report a minimum of 0 cm/s and Greenberg and his colleagues (1996) suggest preferred velocities of < 10 and < 20 cm/s based on data collected during diurnal and dusk dives, respectively. Vismara and his colleagues (2001) also suggest juvenile brown trout preferring water column velocities of less than 20 cm/s. The lowest mean near bed velocity, 2.2 cm/s, was measured by Cunjak and Power (1986). Heggenes and Dokk (2001) report a median of 2 cm/s during a winter survey, while Heggenes and colleagues (2002) report focal point velocities ranging from 0 to 40 cm/s and Mäki- Petäys and colleagues (2000) suggest preference ranges for near bed velocities starting a 0 cm/s. Finally, Roussel and his colleagues (1999) report a preference for snout velocity of –2 cm/s. Most fish will orient themselves into the current (Heland et al. 1995) so the negative value reported is more likely positive from the fish's perspective but relatively low regardless. The

Water Column Velocity

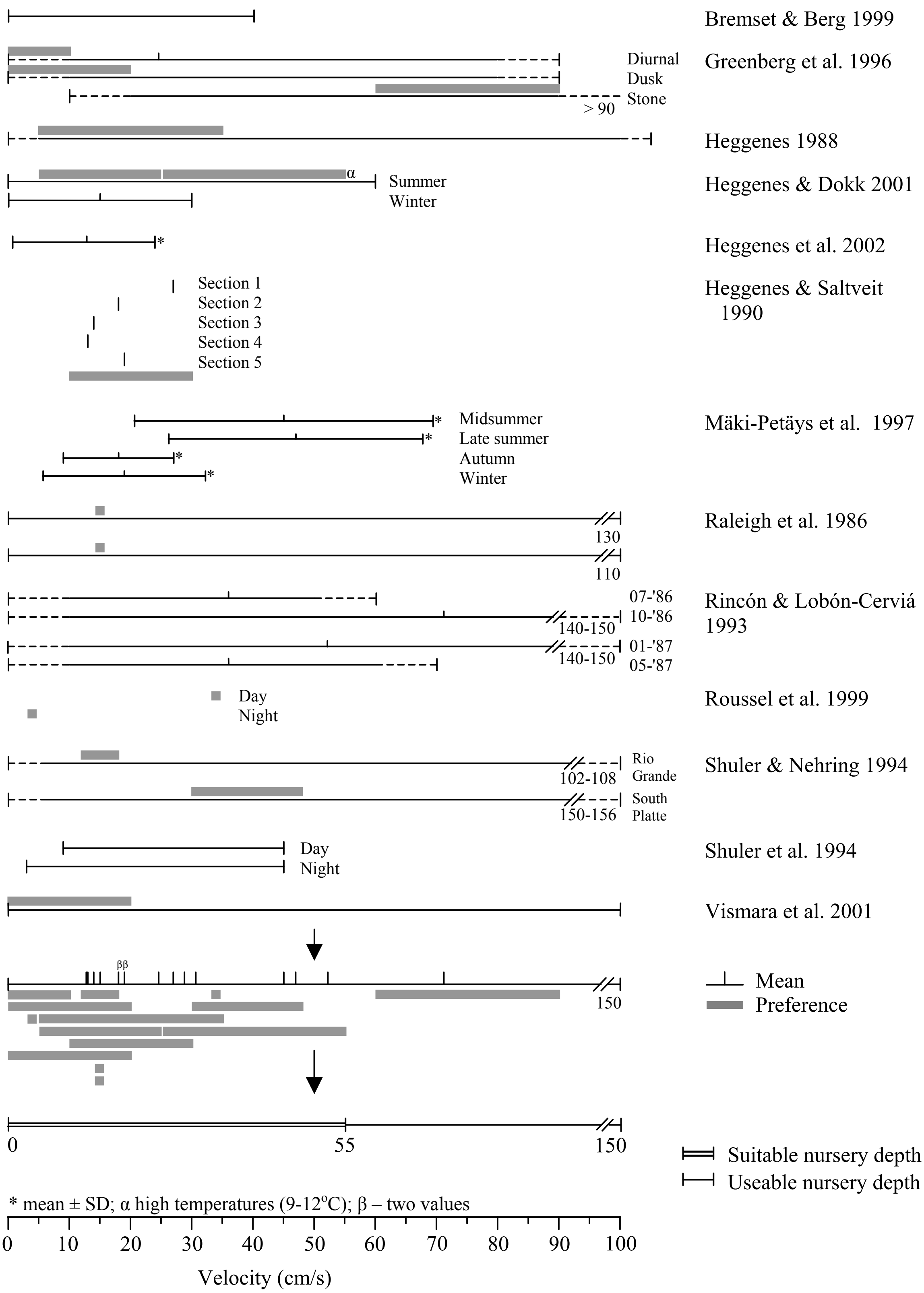
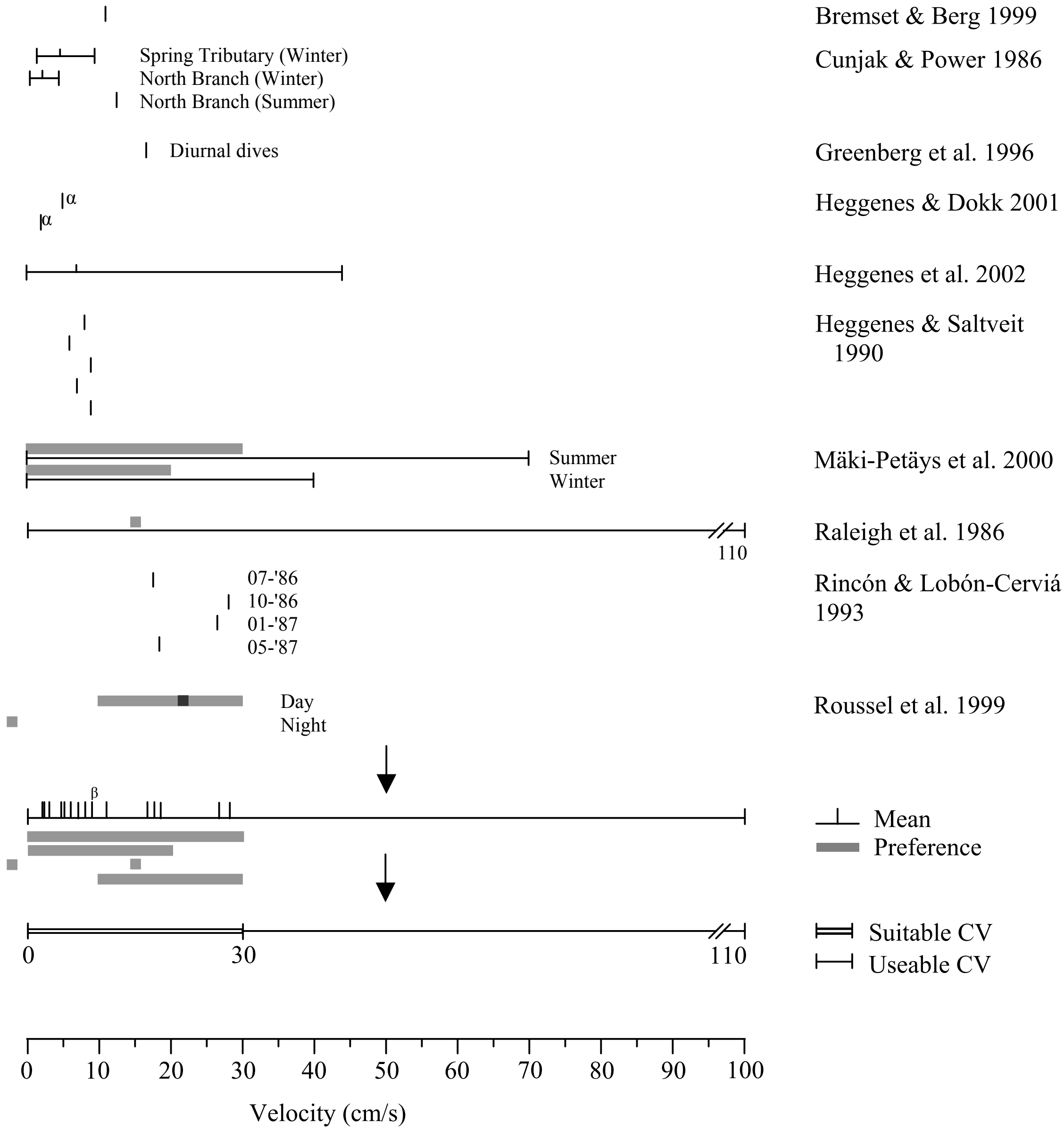


Figure 5.3: Procedure used to define the range of velocities measured in the water column that can be used by juvenile brown trout (*Salmo trutta*) (> 7 cm – 20 cm).

Near Bed Water Velocity



* mean ± SD; α median; β two values

Figure 5.4: Procedure used to define the range of velocities measured near the substrate that can be used by juvenile brown trout (*Salmo trutta*) (> 7 cm – 20 cm).

Table 5.3: Literature used to define the range of stream velocities that can be used by juvenile brown trout (*Salmo trutta*) (>7 – 20cm).

Source	Water column velocity (WC) † *cm/s*	Near bed velocity (NBV) † *cm/s*	Location of measure	Number of fish sampled	Preferred velocity (FH/SC)$^{\alpha}$ *cm/s*	Category range (FH/SC) *cm/s*	Category size (FH/SC) *cm/s*	Fish size; mean ±SD (range) *cm*	Study Location	Allo - sympatric Populations	Other species	Survey method
Bremset and Berg 1999	[0-40]	11	NBV-focal point	72	-	-	-	(5.5-18.5)	Norway (central)	Sympatric	Atlantic salmon	Underwater
Cunjak & Power 1986 Spring Trib. North Branch	-	$^{\alpha}$ 4.7 [1.5-9.5] $^{\alpha}$2.2 [1.5-4.5]; $^{\beta}$13.6	NBV-Focal point	11 9	-	-	-	11.6 (10–12) 11.1 (9–12)	Canada (Ontario)	Sympatric	Brook Trout	Underwater
Greenberg et al. 1996 Diurnal dive Dusk dive Stone dives	24.6 - -	16.8 - -	WC – 0.6 depth NBV – snout position	55 83 32	<10 <20 60 to >90	0-10 to 80-90 0-10 to 80-90 10-20 to >90	10 10 10	7-11	Sweden (Norrland)	Sympatric	Grayling	Underwater
Heggenes 1988	-	-	WC- 0.6 x depth	130	5-35	0-5 to 100-105	5	17.0 (12.1-27.5)	Norway (southeast)	Sympatric	Brown trout	Electro-fishing
Heggenes and Dokk 2001 Summer Winter	- 15±16	5 (median) 2 (median)	WC – 0.6 Depth NBV-focal point	120	25-55$^{\delta}$ 5-25 0-15	0-60 0-30	-	3 [5-16]	Norway (southwest)	Sympatric	Atlantic salmon Arctic Charr	Underwater
Heggenes and Saltveit 1990 Section 1 Section 2 Section 3 Section 4 Section 5	27 18 14 13 19	8 6 9 7 9	WC – 0.6 depth (0.2/0.8 depth >80cm)	26 18 139 81 42	10-30	-	-	12 13 9 9 9	Norway (western)	Sympatric	Atlantic Salmon	Underwater
Heggenes et al. 2002	12.9±11.1	7.0±6.5 [0-44]	WC – 0.6 depth NBV-focal point	937	-	-	-	≥7	England (southwest)	Sympatric	Atlantic Salmon	Underwater
Mäki-Petäys et al. 2000 Summer Winter	-	-	BV – 0.8 depth	200	0-30 0-20	0-70 0-40	-	9.1±0.8 9.5±0.8	Finland (northern)	-	-	Artificial streams Surface observation

Table 5.3: Continued.

Source	Water column velocity (WC) † *cm/s*	Near bed velocity (NBV) † *cm/s*	Location of measure	Number of fish sampled	Preferred velocity (FH/SC)$^{\alpha}$ *cm/s*	Category range (FH/SC) *cm/s*	Category size (FH/SC) *cm/s*	Fish size; mean (range) *cm*	Study Location	Allo - sympatric Populations	Other species	Survey method
Mäki-Petäys et al. 1997 Midsummer Late Summer Autumn Winter	45±(5.2) 47±(4.0) 18±(2.2) 19±(2.7)	-	WC – 0.6 depth	22 27 17 24	-	-	-	10-15	Finland (northern)	Sympatric	Grayling	Electro-fishing
Raleigh et al. 1986 (based on Gosse et al. 1977 & Gosse 1981)	-	-	WC- 0.6 depth NBV- snout velocity	>190	15 (WC) 15 (NBV)	0-130 0-110	-	5.7-9.3	USA (Utah)	-	Rainbow trout	Underwater
Rincón and Lobón-Cerviá 1993 July 1986 October 1986 January 1987 May 1987	30.6 71.1 52.2 28.9	17.7 28.1 26.7 18.5	WC – 0.6 depth (0.2/0.8 depth >80 cm) NBV- snout velocity	54 58 39 42	-	0-10 to 50-60 0-10 to 140-150 0-10 to 140-150 0-10 to 60-70	10	12.6 13.1 14.1 13.4	Spain	Allopatric	-	Underwater
Roussel et al. 1999 Day Night	-	-	WC – 0.3, 0.6, 0.9 depth NBV – snout velocity	43 55	34^{ϕ}; 22 $(10\text{-}30)^{\gamma}$ 4^{ϕ}; -2^{γ}	-5-0 to $45\text{-}50^{\phi}$ -5-0 to $35\text{-}40^{\gamma}$ -5-0 to $20\text{-}25^{\phi}$ -5-0 to $15\text{-}20^{\phi}$	5 5	10-20 (FL)	France (Brittany)	Allopatric	-	Underwater
Shuler and Nehring 1994 Rio Grande South Platte	-	-	WC- 0.6 depth	62 -	12-18 30-48	0-6 to 102-108 0-6 to 150-156	6 6	13-19 13-19	USA (Colorado)	Sympatric	Rainbow trout	Underwater; Angling

Table 5.3: Continued.

Source	Water column velocity (WC) † *cm/s*	Near bed velocity (NBV) † *cm/s*	Location of measure	Number of fish sampled	Preferred velocity (FH/SC)$^{\alpha}$ *cm/s*	Category range (FH/SC) *cm/s*	Category size (FH/SC) *cm/s*	Fish size; mean (range) *cm*	Study Location	Allo - sympatric Populations	Other species	Survey method
Shuler et al. 1994 Day Night	[9-45] [3-45]	-	WC – 0.6 depth	62 92	-	-	-	13-19	USA (Colorado)	Sympatric	Rainbow trout	Underwater Surface (angling)
Vismara et al. 2001	-	-	WC – 0.6 depth (0.2/0.8 depth >75cm)	315	< 20	0-100	-	12-22	Spain	Sympatric	Grayling	Electro-fishing

† mean ±SD (SE) [range]; α - winter mean; β- summer mean; δ - high temperatures (9-12°C); ϕ - Mean Velocity; γ - Focal Velocity

trout in this instance were occupying areas will little or no velocity (Roussel et al. 1999).

The highest mean water column velocity comes from an American study by Shuler and Nehring (1994) who report observing juvenile trout in velocities of 150-156 cm/s. Rincón and Lobón-Cerviá (1993) report an upper range of 140 – 150 cm/s in their preference curves. For this summary, suitability curves and frequency histograms are somewhat challenging for use in constructing the limits for tolerance profiles as they place data into categories and specific limits are hard to determined. In this instance the highest observations of mean water column velocity are expressed as frequency distributions (Rincon and Lobon-Cervia 1993, Shuler and Nehring 1993) and will be used to determine the upper limit of the 'usable' portion of the tolerance profile. It is difficult to determine exactly where within the category range observations were made; however, they would have been equal or greater than the lower limit of the category. Thus, the lower limit of Shuler and Nehring's (1994) 150-156 cm/s category is the highest mean water column velocity reported and will be used as the upper limit of the tolerance profile. Raleigh and colleagues (1986) recommend a snout velocity of 110 cm/s, which will be used as the upper limit for the near bed velocity tolerance profile.

The bulk of the mean values and preference ranges for measures of mean column velocity fall between 0 and 55 cm/s. In fact, of the 16 measures of central tendency and the 15 preference ranges expressed (totalling 31 observations) 29 falls below the 55 cm/s limit. The exceptions include the preference range of trout sheltering under stones examined by Greenberg and colleagues (1996) (60 to >90 cm/s) and the mean of observation observed by Rincón and Lobón-Cerviá (1993) in their October 1986 survey

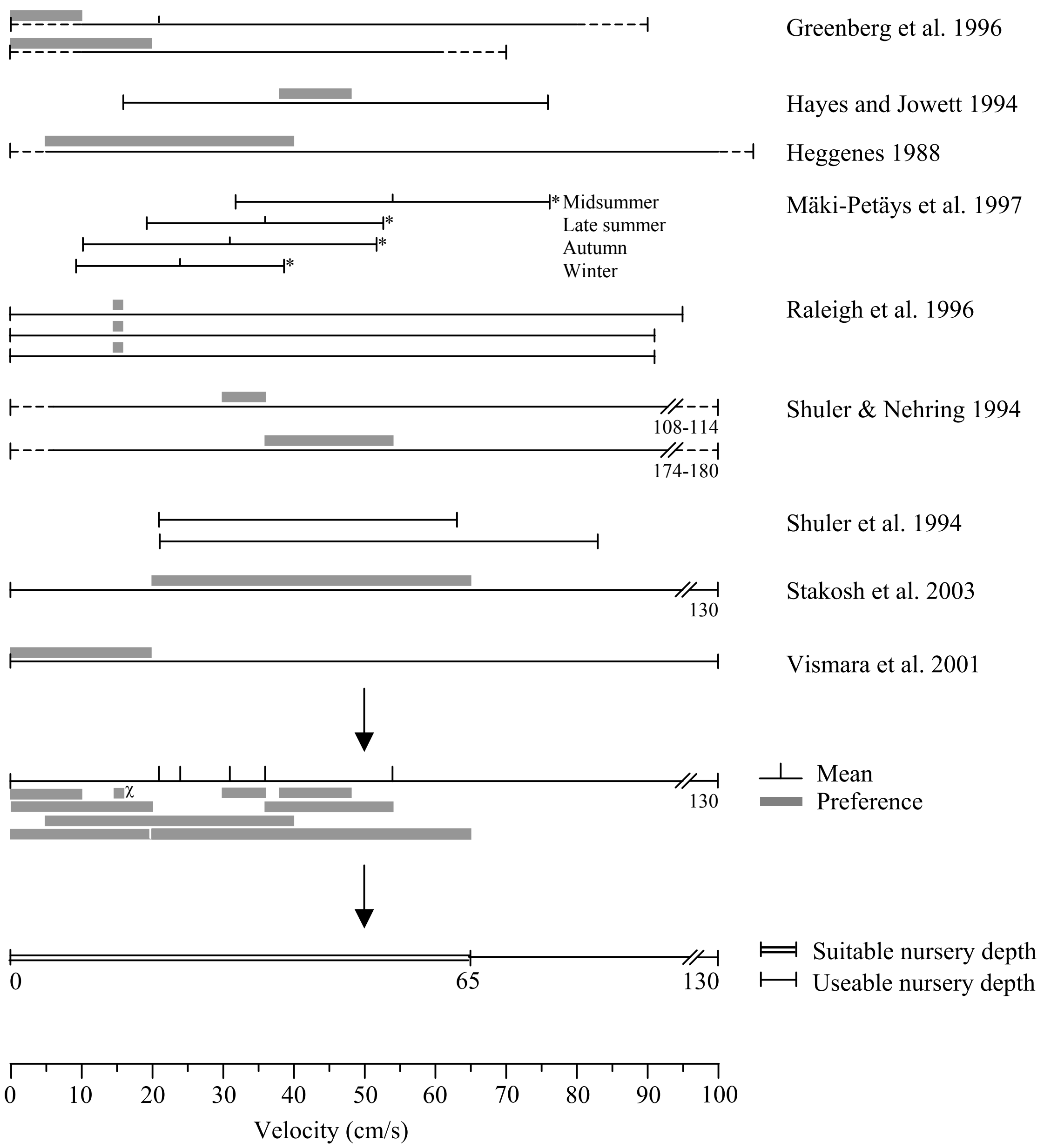
(71.1 cm/s). High water column velocities were previously observed for emergent fry occupying nursery habitat sheltering under stones and it was concluded that these individuals were not experiencing these high velocities in their sheltered microhabitats. Rincón and Lobón-Cerviá (1993) observation are higher than the bulk of water column observation and their preference range and the mean reported by Greenberg and his colleagues (1996) will be included in the 'usable' portion of the tolerance rather than the 'suitable' portion. All the measures of central tendency and the preference ranges for measures of substrate water velocities are below 30 cm/s without exception and this value will be the upper limit of the 'suitable' range of the substrate water velocities used by juvenile brown trout. Thus, the suitable range for water column measures of water velocity based on the literature surveyed includes a suitable range from 0 to 55 cm/s and a useable range of 55 to 150 cm/s. The tolerance profile for velocities measured closer to the substrate includes a suitable range of 0 to 30 cm/s and a useable range of 30 to 110 cm/s.

5.3.1.4 Velocity use by non-spawning adults (> 20 cm)

Investigations of adult brown trout have come from a wide range of the species worldwide distribution. These studies have come from Europe (Heggenes 1988a, Greenberg et al. 1996, Mäki-Petäys et al. 1997, Vismara et al. 2001), North America (Baldes and Vincent 1969, Cunjak and Power 1986, Raleigh et al. 1986, Shuler and Nehring 1993, Shuler et al. 1994, Strakosh et al. 2003), and New Zealand (Shirvell and Dungey 1983, Hayes and Jowett 1994). All of the sites observed contained sympatric populations of brown trout with the exception of the work by Baldes and Vincent (1969), Heggenes 1988, Stakosh and his colleagues (2003), and some sites examined by Shirvell and Dungey (1983). The salmonids in sympatry with adult *S. trutta* in the set of literature summarized include brook trout (Cunjak and Power 1986), grayling (Greenberg et al. 1996, Mäki-Petäys et al. 1997, Vismara et al. 2001), and rainbow trout (Shirvell and Dungey 1983, Raleigh et al. 1986, Shuler and Nehring 1993, Shuler et al. 1994). The sampling methodology included underwater observations (Cunjak and Power 1986, Raleigh et al. 1986, Shuler and Nehring 1993, Shuler et al. 1994, Greenberg et al. 1996), electro-fishing (Heggenes 1988a, Mäki-Petäys et al. 1997, Vismara et al. 2001), and surface observations (Baldes and Vincent 1969, Shirvell and Dungey 1983, Hayes and Jowett 1994). Shuler and Nehring (1994) and Shuler and his colleagues employed an angling sample protocol in some instances. All of this work was conducted in natural streams with the exception of that completed by Baldes and Vincent (1969) who used and artificial stream apparatus. The literature used in this synopsis is summarized in Figure 5.5 & 5.6 and reported in more detail in Table 5.4.

There are five different measures of mean column velocity used by adult trout used in the survey (calculated by different authors). These range from 21 cm/s (Greenberg et al. 1996) to 54 cm/s (Mäki-Petäys et al. 1997). There are ten preference ranges expressed by the authors surveyed for mean column velocity: these range from lows of <10 m/sec suggested by Greenberg and his colleagues (1996) working in Sweden to 20 – 65 cm/s suggested by Stakosh and his colleagues (2003) working in the north-eastern United States. The maximum velocity that adult non-spawners were observed using was reported by Shuler and Nehring (1994) and expressed as a frequency histogram. They observed large trout using mean water column velocities as high as 174-180 cm/s in the Rio Grande (USA) although they calculate a preference range of 36-54 cm/s. The lower limit of the 174-180 cm/s category in Shuler and Nehring's study will be used as the

Mean column velocity



* - mean ± SD; χ - three values

Figure 5.5: Procedure used to define the range of mean water column velocities that can be used by non-spawning adult brown trout (*Salmo trutta*) (>20 cm).

Near Bed Velocity

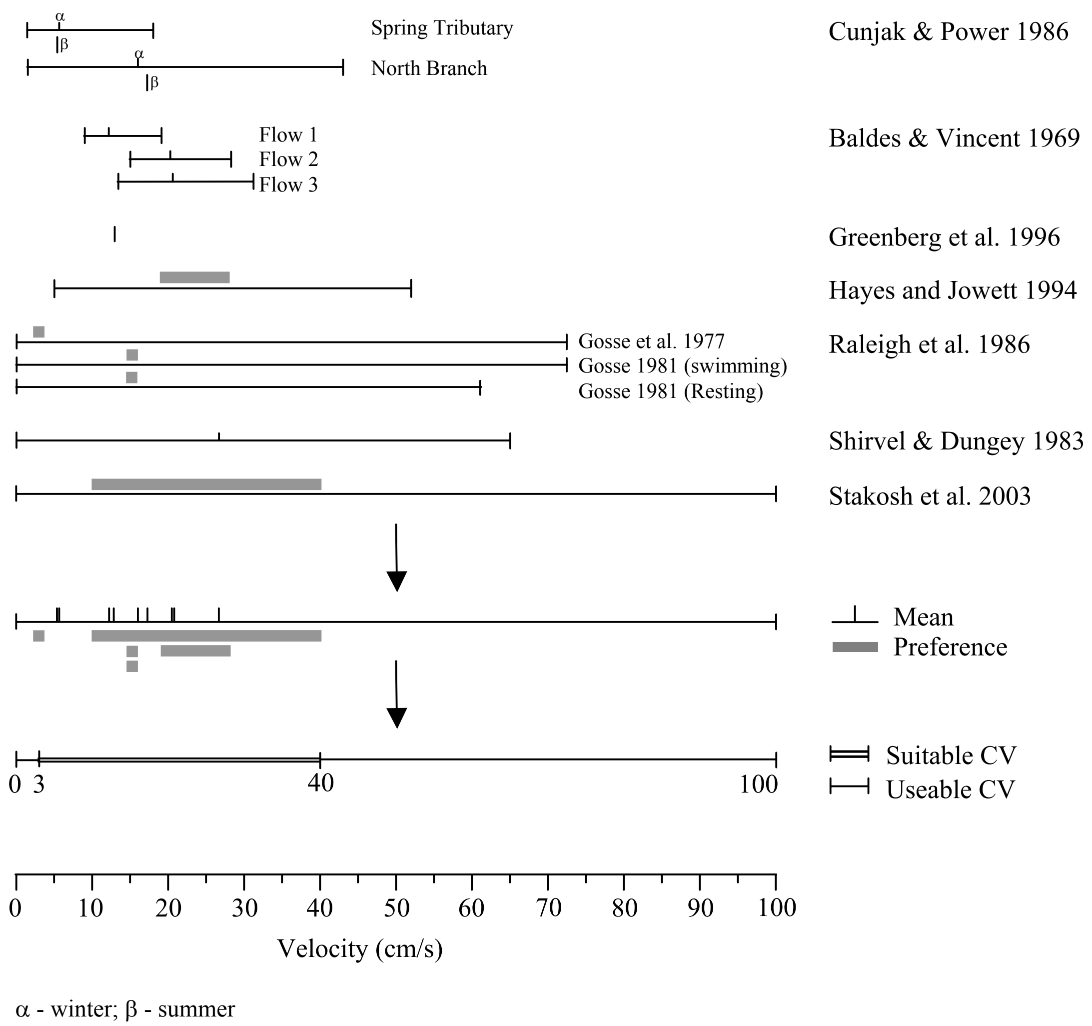


Figure 5.6: Procedure used to define the range of near bed mean column velocities that can be used by non-spawning adult brown trout (*Salmo trutta*) (>20 cm).

Table 5.4: Literature used to define the range of stream velocities that can be used by adult brown trout (*Salmo trutta*) (>20cm).

Source	Water column (WC) velocity † *cm/s*	Near bed velocity (NBV) † *cm/s*	Location of measure	Number of fish sampled	Preferred velocity (FH/SC) *cm/s*	Category range (FH/SC) *cm/s*	Category size (FH/SC) *cm*	Fish size; mean (range) *cm*	Study Location	Allo-sympatric populations	Other species	Survey method
Baldes & Vincent 1969 $^{\gamma}$Flow 1 Flow 2 Flow 3	- - -	$^{\delta}$12.2 (9.1-19.2) $^{\delta}$20.4 (15.2-28.3) $^{\delta}$20.7 (13.1-31.1)	NBV – 12.7cm from bottom	108	-	-	-	20.3$^{\alpha}$	USA (Colorado)	Allopatric	-	Surface (artificial stream)
Cunjak & Power 1986 Spring Trib. North Branch	-	$^{\alpha}$5.7 [1.5-18.0];$^{\beta}$5.4 $^{\alpha}$16.0 [1.5-43.0];$^{\beta}$17.2	NBV-focal point	16 36	-	-	-	20.3 (12-40) 21.9 (12-50)	Canada (Ontario)	Sympatric	Brook Trout	Underwater (winter)
Greenberg et al. 1996 Diurnal dive Dusk dive Stone dives	21.0 - -	12.9 - -	WC – 0.6 depth NBV – snout position	111 55 -	<10 <20 -	0-10 to 80-90 0-10 to 60-70 -	10 10 -	12-35	Sweden (Norrland)	Sympatric	Grayling	Underwater
Hayes and Jowett 1994 Mean (WC) Focal point	[16-76] -	- [5-52]	WC – 0.4 depth (0.2/0.8 depth > 1m)	151	38-48 19-28	-	-	45-65	New Zealand	-	-	Surface
Heggenes 1988	-	-	WC- 0.6 depth	11	5-40	0-5 to 100-105	5	24.7 (0.23 SE) n=19	Norway (southeast)	Allopatric	-	Electro-fishing
Mäki-Petäys et al. 1997 Midsummer Late Summer Autumn Winter	54±4.9 36±4.8 31±5.2 24±4.6	-	WC – 0.6 depth	20 12 16 10	-	-	-	16-25	Finland (northern)	Sympatric	Grayling	Electro-fishing

Table 5.4: Continued.

Source	Water column (WC) velocity † *cm/s*	Near bed velocity (NBV) † *cm/s*	Location of measure	Number of fish sampled	Preferred velocity (FH/SC) *cm/s*	Category range (FH/SC) *cm/s*	Category size (FH/SC) *cm*	Fish size; mean (range) *cm*	Study Location	Allo-sympatric populations	Other species	Survey method
Raleigh et al. 1986 Gosse et al. 1977 Gosse 1981(swim) Gosse 1981 (rest)	-	-	WC-0.6 depth NBV – snout velocity	352 225 222	$^{\varepsilon}$15.2; $^{\phi}$3.0 $^{\varepsilon}$15.2;$^{\phi}$15.2 $^{\varepsilon}$15.2;$^{\phi}$15.2	$^{\varepsilon}$0-95.5; 0-$^{\phi}$73.2 $^{\varepsilon}$0-91.4; 0-$^{\phi}$73.2 $^{\varepsilon}$0-91.4; 0-$^{\phi}$61.0	-	> 9.4 - -	USA (Utah)	-	Rainbow trout	Underwater
Shirvel & Dungey 1983	-	26.7±11.5 [0-65]	NBV – snout velocity	140	-	-	-	30-60	New Zealand	Allopatric Sympatric	Rainbow trout	Surface
Shuler and Nehring 1994 Rio Grande South Platte	-	-	WC- 0.6 depth	208 -	30-36 36-54	0-6 to 108-114 0-6 to 174-180	6 6	≥20 ≥20	USA (Colorado)	Sympatric	Rainbow trout	Underwater; Angling
Shuler et al. 1994 Day Night	[21-63] [21-83]	-	WC – 0.6 depth	208 104	-	-	-	≥20	USA (Colorodo)	Sympatric	Rainbow trout	Underwater Surface (angling)
Stakosh et al. 2003	-	-	WC – 0.6 depth (0.2/0.8 depth > .75m) NBV- snout velocity	144	(WC) 20-65 (NBV) 10-40	(WC) 0-130 (NBV) 0-100	-	≥ 17	USA (Conn)	Allopatric	-	Underwater
Vismara et al. 2001	-	-	WC – 0.6 depth (0.2/0.8 depth > .80cm)	213	< 20	0-100	-	>22	Spain	Sympatric	Grayling	Electro-fishing

† mean±SD (SE) [range]; α - winter; β- summer; γ Flow 1 (0.37 cm/s),Flow 2 (1.10 cm/s), Flow 3 (2.07 cm/s); δ – mode; ε - mean column velocity; ϕ snout velocity

upper limit of the tolerance profile for measures of mean column velocity used by adult non-spawning trout. All the measures of central tendency and preference ranges (or values) fell between 0 and 65 cm/s and this range will be used as the 'suitable' velocities in the tolerance profile. Mean velocities ranging from 65 to 174 cm/s will be considered 'useable' by large trout.

Mean measures of near bed velocity were more numerous than mean water column estimates; however, there were fewer expressions of trout preference. There were nine measures of central tendency ranging from a mean of 5.4 cm/s during the summer in Spring Tributary, Canada (Cunjak and Power 1986) to 26.7 cm/s observed by Shirvell and Dungey (1983) in New Zealand. There were five preference velocities (or ranges) for near bed velocity, ranging from 3 cm/s (Gosse et al. 1977 see: Raleigh et al 1986) to 10 to 40 cm/s reported by Stakosh et al. 2003. All values of both preference and direct estimates of central tendency (mean or mode) fall between 3 and 40 cm/s. This range will be used as the 'suitable' range of substrate water velocities for the tolerance profile of adult non-spawning brown trout.

The near bed water velocities observed being utilized by large brown trout include a range of 0 to 65 cm/s reported by Shirvell and Dungey 1983 and 0 to 100 cm/s observed by Stakosh et al. 2003 which was reported as a suitability curve. It is important to remember that with suitability curves, velocities (or depths, substrate measures etc.) corresponding to low suitability values are considered 'not-usable' for use by a given species or age class. However, they do correspond to observations of individuals using these velocities (or other microhabitat parameter) and can be used to help define the 'useable' portion of a tolerance profile. Thus the useable portion of the tolerance profile for measures of substrate water velocity for large brown trout (>20 cm will range from 0 to 100 cm/s). Within this range suitable segment will be defined as ranging from 3 to 40 cm/s.

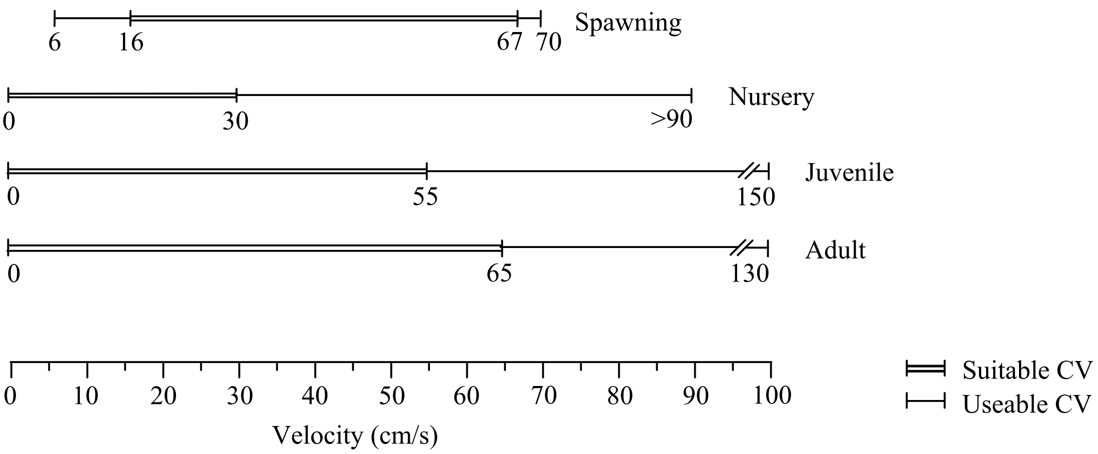
5.2.1.5 Summary of tolerance profiles

The tolerance profiles created for all four life-stages of brown trout for measures of both mean column and near bed velocity have been compiled and are illustrated in Figure 5.7. Similar trends are apparent in both summaries. With the exception of adult near bed velocities, where the minimum suitable velocity is 3 cm/s, brown trout at all age classes and for both types of measures have suitability ranges beginning at of 0 cm/s; the range of suitable velocities available to brown trout increases as fish grow; and the range of water velocity suitable for spawning has a higher minimum and maximum than the tolerance profiles at other life stages. The maximum velocity measured was highest in the juvenile tolerance profiles for both sets. However, this probably reflects availability rather than choice (Heggenes 1988c, Mäki-Petäys et al. 1997, Liebig et al. 2001, Heggenes et al. 2002).

This result of this summary is not unexpected.. Numerous studies have noted a preference for low velocities at all non-spawning life stages (Karlstrom 1977, Ottaway and Forrest 1983, Bachman 1984, Heggenes 1988c, Fausch 1993, Bremset and Berg 1999, Bremset 2000, Dare and Hubert 2003). From a hydraulic perspective, slower water velocities are generally observed in the water column with increased depth (Cunjak and Power 1986, Rincon and Lobon-Cervia 1993, Heggenes and Dokk 2001, Heggenes et al. 2002) and the variability in the range of values also decreases

A) Mean Column Velocity

(0.6 depth)



B) Substrate Water Velocity
(Focal or snout velocity)

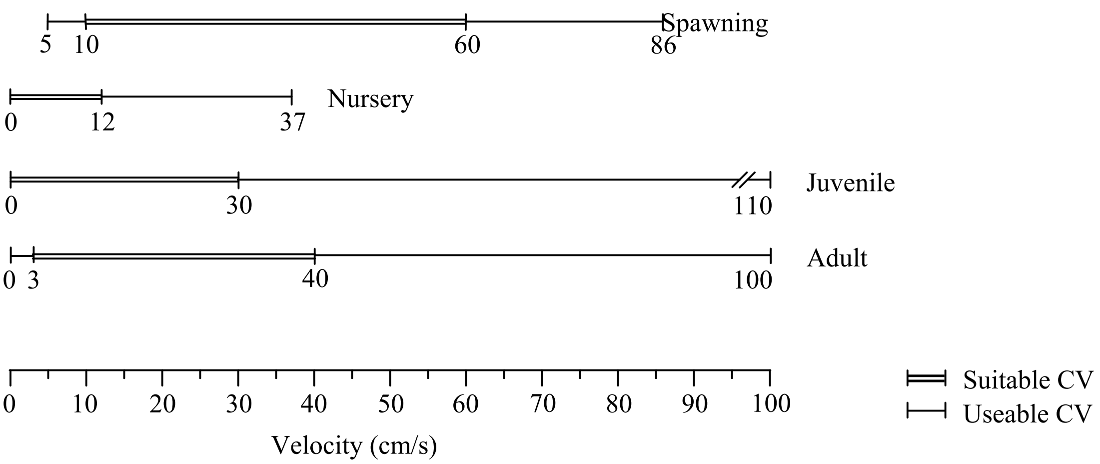


Figure 5.7: Summary of tolerance profiles by age class [including spawning; nursery (≤7 cm); juvenile (>7 –20 cm); and adult non-spawners (> 20 cm)] for measures of A) mean column velocity and B) near bed water velocity used by brown trout (*Salmo trutta*).

(Heggenes 2002, Heggenes et al. 2002). Larger trout are more suited to withstand higher velocities and subsequently have a higher range of microhabitat available for use (Bohlin 1977, Cunjak and Power 1986, Hayes 1991, Shuler et al. 1994, Greenberg et al. 1996, Kocik and Taylor 1996). Spawning trout require higher velocities than those found at random in a stream (Essington et al. 1998) indicating they are actively seeking swiftly flowing areas to spawn. As well, workers have suggested there is a minimum velocity that is suitable for spawning (Huet 1962, Crisp and Carling 1989) which is supported by this summary.

5.3.2 Habitat preference

The profiles constructed in Section 5.2 were then applied to the data gathered from each quadrat measured in the reaches studied. The proportion of available habitat for each study reach at each life stage is summarized in the following sections.

5.3.2.1 Spawning velocity

The proportion of suitable, useable, and not-usable water velocity for both the total and wetted streambed that is available at each site was compared using a chi-square test. This was done to compare the seven data sets (March, Burnhouse, and Bin Bun upstream and downstream; plus a before and after spate comparison at Bin Burn-downstream) to see if the proportion of suitable, useable, not-useable, and dry quadrats differed statistically in the total and wetted streambed. The results of these analyses are presented in Tables 5.5 and 5.6. The chi-square value for both the total (X^2 = 493.4, df = 18, p = 0.000) and the wetted (X^2 = 307.8, df = 12, p = 0.000) area of the streambed indicate that the relative proportion of the velocities are not independent of reach (i.e. the proportion of habitat classifications vary from site to site). The proportion of water velocity suitability classifications for each site is outlined in Table 5.7. In the total streambed area the proportion of suitable velocity for spawning varied from 8.5% at Burnhouse Burn-downstream to 49.3% at March Burn-upstream. Useable water velocities ranged from 5.8% at Bin Burn-downstream (post spate) to 40.3% and March Burn-downstream. The lowest amount of velocity that is not-useable for spawning is

Table 5.5: Results for chi-square test comparing the proportions of quadrats that contained water velocities that were considered suitable, useable, not-usable and dry for spawning brown trout for both the total and wetted streambed at all study reaches.

Life Stage	Streambed	X^2 value	DF	Cells < 5†	p-value	X^2 (Likelihood)*
Spawn	Total	493.4	18	0	0.000	no
	Wetted	307.8	12	0	0.000	no
Nursery	Total	317.2	18	0	0.000	no
	Wetted	122.0	12	0	0.000	no
Juvenile	Total	306.4	18	7	0.000	yes
	Wetted	121.0	12	7	0.000	yes
Adult	Total	275.6	18	7	0.000	yes
	Wetted	90.2	12	7	0.000	yes

† Number of cells with expected values less than 5; * Used likelihood ratio chi-square

Table 5.6 Results for chi-square test comparing the proportions of quadrats that contained water velocities that were considered suitable, useable, not-usable and dry for spawning brown trout for both the total and wetted streambed at Bin Burn (downstream) before and after the spate.

Life Stage	Streambed	X^2 value	DF	Cells < 5†	p-value	X^2 (Likelihood)*
Spawn	Total	61.0	3	0	0.000	no
	Wetted	3.2	2	0	0.202	no
Nursery	Total	77.0	3	0	0.000	no
	Wetted	17.1	2	0	0.000	no
Juvenile	Total	61.5	3	2	0.000	yes
	Wetted	3.8	2	2	0.149	yes
Adult	Total	64.6	3	2	0.000	yes
	Wetted	6.9	2	2	0.031	yes

† Number of cells with expected values less than 5; * Used likelihood ratio chi-square

Table 5.7: Summary of the proportion of velocities available for the four life stages of brown trout for all study reaches including both total and the wetted portions of the streambed

		Spawn Total (%)	Wetted (%)	Nursery Total (%)	Wetted (%)	Juvenile Total (%)	Wetted (%)	Adult Total (%)	Wetted (%)
March Burn Upstream	Suitable	49.3	51.8	42.7	44.9	63.9	67.2	81.8	86.0
	Useable	26.3	27.7	52.4	55.1	31.2	32.8	13.3	14.0
	Not Useable	19.5	20.5	0.0	0.0	0.0	0.0	0.0	0.0
	Dry	4.9	-	4.9	-	4.9	-	4.9	-
March Burn Downstream	Suitable	16.7	27.8	47.9	79.8	55.1	91.9	56.1	93.4
	Useable	40.3	67.1	10.4	17.2	4.9	8.1	3.3	5.6
	Not Useable	3.1	5.1	1.8	3.0	0.0	0.0	0.6	1.0
	Dry	39.9	-	39.9	-	40.0	-	40.0	-
Burnhouse Burn Upstream	Suitable	13.3	34.6	33.6	87.1	38.6	100.0	38.6	100.0
	Useable	8.8	22.7	5.0	12.9	0.0	0.0	0.0	0.0
	Not Useable	16.5	42.7	0.0	0.0	0.0	0.0	0.0	0.0
	Dry	61.4	-	61.4	-	61.4		61.4	-
Burnhouse Burn Downstream	Suitable	8.5	21.4	36.7	92.5	39.6	100.0	39.6	100.0
	Useable	9.5	24.1	3.0	7.5	0.0	0.0	0.0	0.0
	Not Useable	21.6	54.5	0.0	0.0	0.0	0.0	0.0	0.0
	Dry	60.4	-	60.3	-	60.4	-	60.4	-
Bin Burn Upstream	Suitable	21.8	29.1	44.9	59.8	54.0	72.0	57.3	76.4
	Useable	13.9	18.5	18.1	24.2	20.4	27.2	14.2	19.0
	Not Useable	39.3	52.4	12.0	16.0	0.6	0.8	3.5	4.6
	Dry	25.0	-	25.0	-	25.0	-	25.0	-
Bin Burn Downstream (Pre-spate)	Suitable	31.7	47.4	43.6	65.3	55.8	83.5	58.9	88.2
	Useable	12.9	19.3	19.8	29.7	11.0	16.5	7.9	11.8
	Not Useable	22.2	33.3	3.4	5.0	0.0	0.0	0.0	0.0
	Dry	33.2	-	33.2	-	33.2	-	33.2	-
Bin Burn Downstream (Post-spate)	Suitable	19.5	46.9	27.6	66.4	36.5	87.8	39.0	93.6
	Useable	5.8	14.0	13.9	33.3	5.1	12.2	2.6	6.4
	Not Useable	16.3	39.1	0.1	0.3	0.0	0.0	0.0	0.0
	Dry	58.4	-	58.4	-	58.4	-	58.4	-

was found at March Burn downstream (3.1%) and the highest proportion was found at Bin Burn-upstream (39.3%). Dry streambed ranged from 4.9% at March Burn-upstream to 61.4% at Burnhouse Burn-upstream.

In the wetted portion of the streambed the highest proportion of suitable spawning velocity was found at March Burn-upstream (51.8%) and the lowest was found at Burnhouse Burn-downstream (21.4%). Velocities classified as useable for spawning varied from 14.0% to 67.1% at Bin Burn-downstream (post-spate) and March Burn-downstream, respectively. Quadrats with velocities that could not be used for spawning varied from 5.1% to 54.5% (March Burn-downstream and Burnhouse Burn-downstream) of the wetted streambeds in the reaches examined. Histograms that illustrate the proportions of velocities that are suitable, useable, and not-suitable for spawning as well as the proportion of dry streambed are provided in Figure 5.8 (A-G).

As previously mentioned in Chapter 4, both the discharge and the surface area within the study reaches varies from site to site (see Table 4.8, Chapter 4), thus the area available with acceptable velocities for spawning can be calculated by multiplying the proportion of suitable and useable habitat by the wetted surface area at each reach. This was done and the results are presented in Table 5.8. The Burnhouse Burn sites, up- and downstream, had the smallest amount of total area with adequate velocities (6.6 m^2 and 11.3 m^2, respectively) while Bin Burn-downstream (post spate) had the largest (54.9 m^2). A larger surface area with adequate spawning velocities was created after the spate and stream reconstruction at the downstream Bin Burn site (from 39.6 m^2 to 54.9 m^2). The March Burn sites and Bin Burn upstream had intermediate values. The chi-square

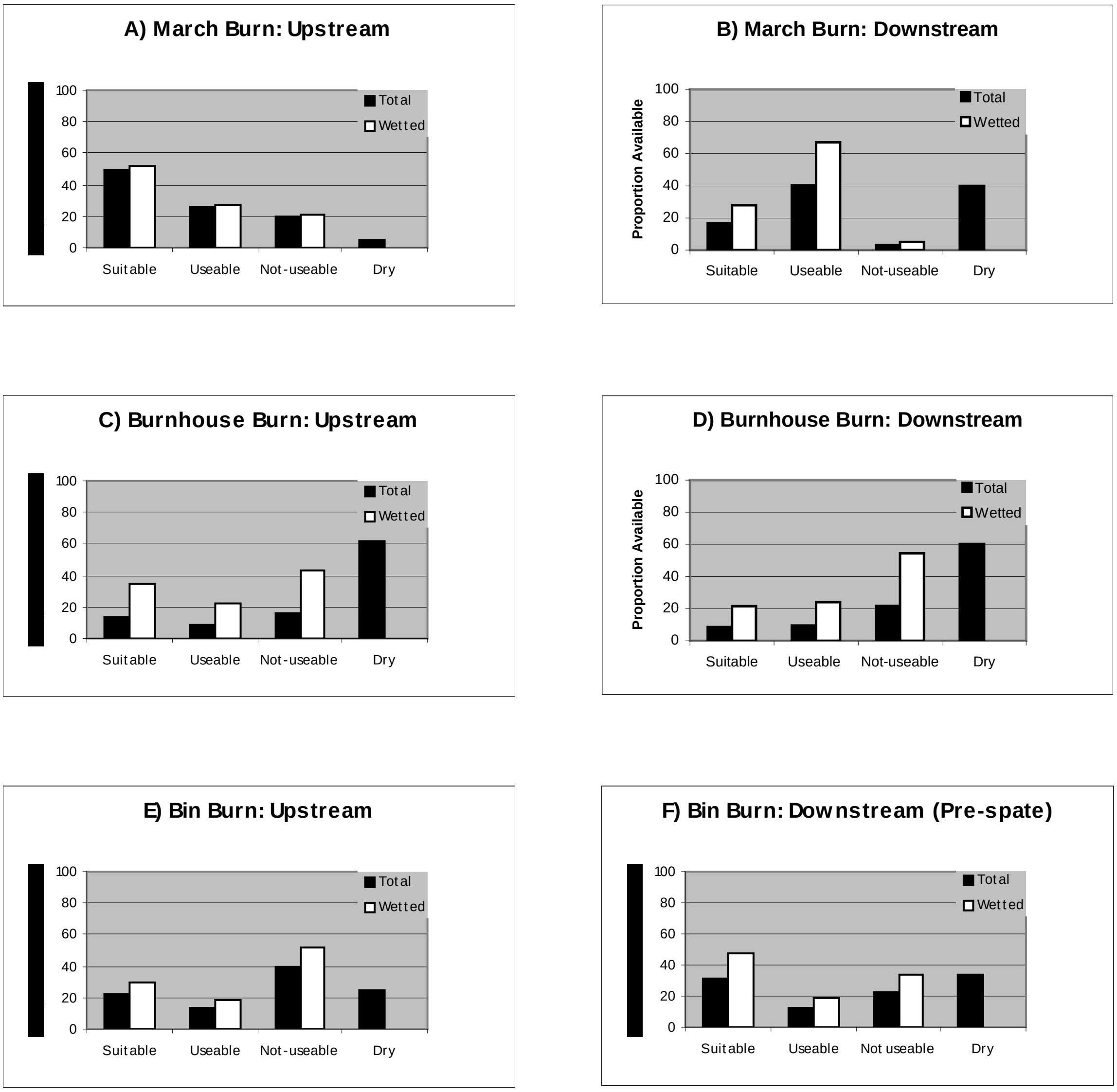


Figure 5.8 (A-G): Histograms of the proportion of velocity available for spawning trout at all study sites classified as suitable, useable, not-suitable and dry for both the total and wetted streambed.

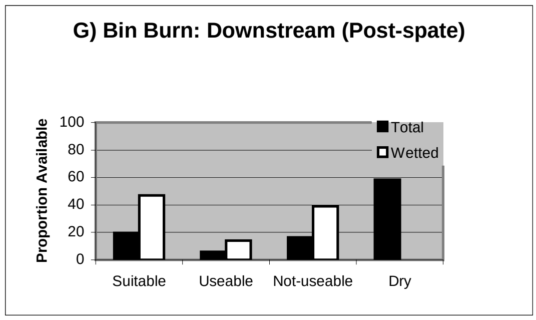


Table 5.8: The total area that has accessible water velocities for each study reach for age class. Total accessible area calculated by multiplying total wetted area by the sum of the proportion of the streambed that was either suitable or useable.

Age Class	Site	Total Wetted Area (m^3)	Proportion of wetted area; suitable or useable (%)	Total area suitable or useable (m^2)
Spawning	March Burn-us†	12.0	79.5	9.5
	March Burn-ds‡	48.8	94.9	46.3
	Burnhouse Burn-us	11.6	57.3	6.6
	Burnhouse Burn-ds	24.9	45.5	11.3
	Bin Burn-us	32.2	47.6	15.3
	Bin Burn-ds (pre spate)	59.4	66.7	39.6
	Bin Burn-ds (post spate)	90.1	61.0	54.9
Nursery	March Burn-us	12.0	100.0	12.0
	March Burn-ds	48.8	97.0	47.3
	Burnhouse Burn-us	11.6	100.0	11.6
	Burnhouse Burn-ds	24.9	100.0	24.9
	Bin Burn-us	32.2	84.0	27.1
	Bin Burn-ds (pre spate)	59.4	95.0	56.4
	Bin Burn-ds (post spate)	90.1	99.7	89.8
Juvenile	March Burn-us	12.0	100.0	12.0
	March Burn-ds	48.8	100.0	48.8
	Burnhouse Burn-us	11.6	100.0	11.6
	Burnhouse Burn-ds	24.9	100.0	24.9
	Bin Burn-us	32.2	99.2	32.0
	Bin Burn-ds (pre spate)	59.4	100.0	59.4
	Bin Burn-ds (post spate)	90.1	100.0	90.1
Adult	March Burn-us	12.0	100.0	12.0
	March Burn-ds	48.8	99.0	48.3
	Burnhouse Burn-us	11.6	100.0	11.6
	Burnhouse Burn-ds	24.9	100.0	24.9
	Bin Burn-us	32.2	95.3	30.2
	Bin Burn-ds (pre spate)	59.4	100.0	59.4
	Bin Burn-ds (post spate)	90.1	100.0	90.1

† upstream; ‡ downstream

test that compared the relative proportion of the suitability classification before and after the spate (Table 5.6) indicates that these proportion are different when examining the total streambed (X^2 = 61.0, df = 3, p = 0.000) but not statistically different when examining the wetted streambed alone (X^2 = 3.2, df = 2, p = 0.000). Although the area available. with velocities increased after the spate the relative proportion of suitable, useable, and not-useable quadrats did not change in the wetted streambed

5.3.2.2 Nursery water velocity (fish length ≤ 7 cm)

The chi-squared analysis of the relative proportion of the suitability classifications based on the distribution of velocity for trout ≤ 7 cm in length revealed that these proportions differ, statistically for the total and wetted surfaces between the seven data sets examined ($X^2 = 317.2$, df = 18, p = 0.000; $X^2 = 122.0$, df = 12, p = 0.000 for the total and wetted streambeds respectively; Table 5.5). The relative proportion of suitable, useable, not-useable, and dry streambed is outlined in Table 5.7 and illustrated in Figure 5.9 (A-G). Like spawning depth these streams seemed to have a larger proportion of the velocities available that were suitable for young trout. When examining the total streambed suitable habitat varied from 27.6% to 47.9% at Bin Burn-downstream (post spate) and March Burn downstream, respectively. Velocities classified as useable ranged from 3.0% at Burnhouse Burn-downstream to 52.4% at March Burn-upstream. The proportion of quadrats with velocities not-useable as nursery habitat ranged from 0% or close to zero at the upstream sites of March and Burnhouse Burns and the downstream sites at Burnhouse and Bin Burns (post-spate) to 12.0% at Bin Burn-upstream.

When considering the wetted area of these streams the vast majority of the quadrats measured had velocities suitable or useable as nursery habitat for brown trout. Suitable velocities ranged from 92.5% at Burnhouse Burn-downstream to 44.9% at March Burn-

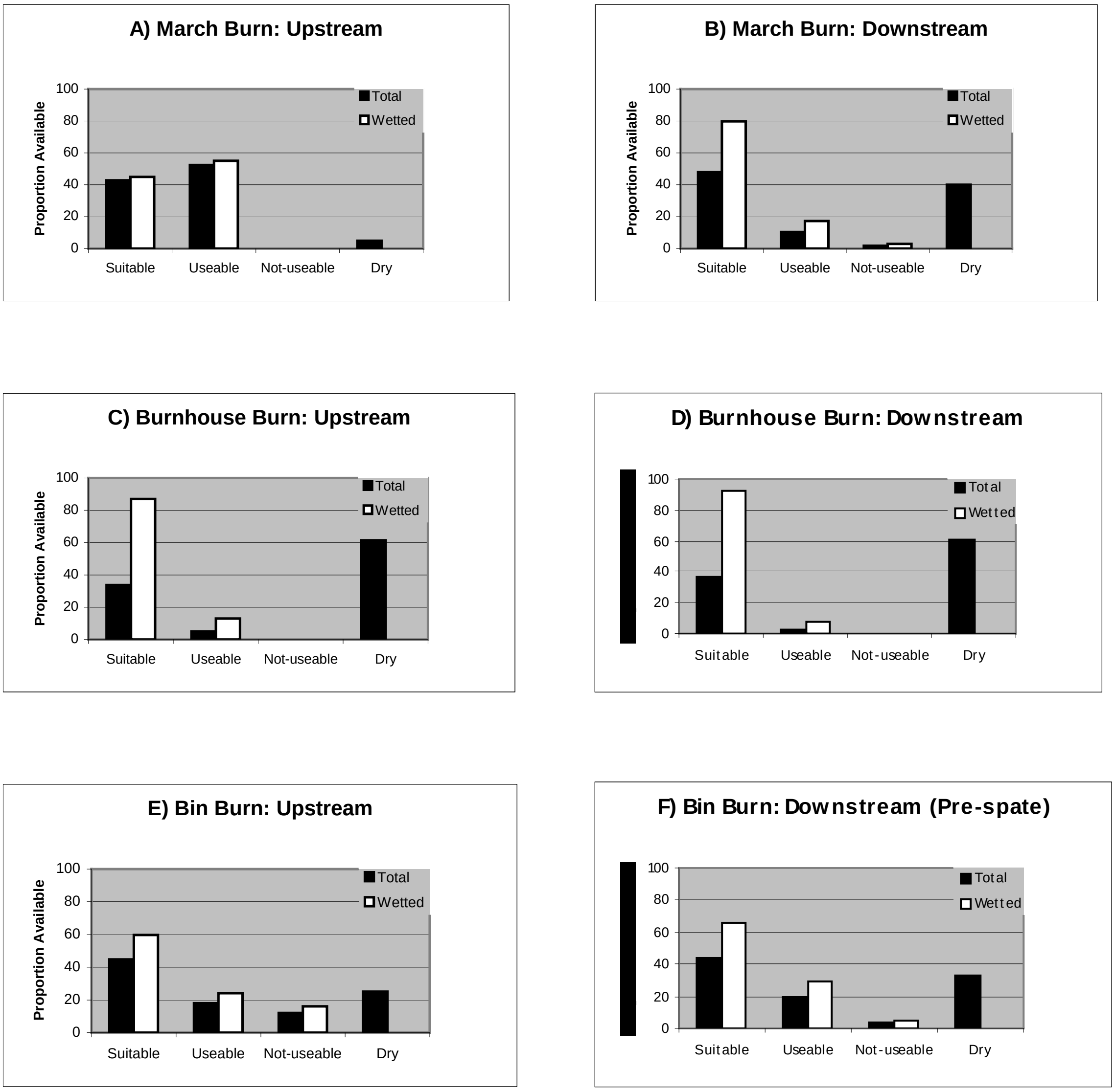
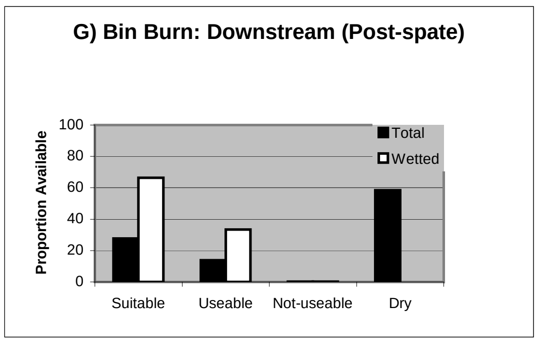


Figure 5.9 (A-G): Histograms of the proportion of measures of velocity available to young trout (≤ 7 cm) at all study sites classified as suitable, useable, not-suitable and dry for both the total and wetted streambed.



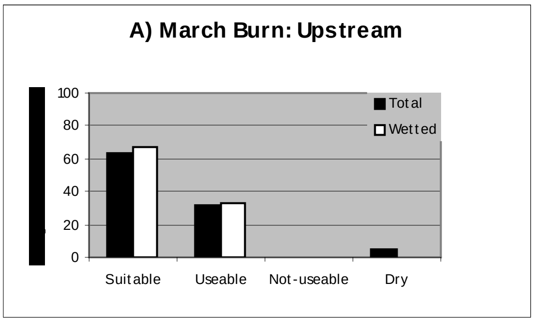
upstream. The highest proportion of suitable habitat was found at March Burn-upstream (55.1%) and the least at Burnhouse Burn-downstream (7.5%). The proportion of quadrats with velocities 'not-useable' varied from 0% (March and Burnhouse Burn upstream; Burnhouse Burn-downstream) to 16% (Bin Burn-upstream). The proportion of dry streambed is the same as previously noted. The total area available with adequate water velocities (proportion of suitable and useable summer multiplied by the wetted area of the reach) for nursery brown trout followed a similar patter as seen for spawning water velocities (Table 5.8). The smallest total surface area with adequate velocities was found at the Burnhouse Burn sites (11.6 m^2 and 24.9 m^2; upstream and downstream respectively), and the largest at Bin Burn downstream (56.4m^2 and 89.9 $m^{2;}$ pre- and post-spate respectively). Intermediate values were found at the March Burn sites and the upstream Bin Burn site.

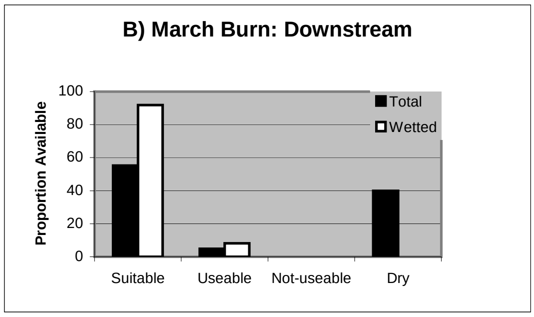
Statistically, the chi-square test indicates that the proportion of the quadrats classified according to the usability criteria based on the tolerance of trout ≤ 7 cm long is different between sites for both the total (X^2 = 77.0, df = 3, p = 0.000) and the wetted streambed (X^2 = 17.1, df = 2, p = 0.000; Table 5.6). As previously mentioned, the discharge at the time of sampling this site pre- and post-spate was similar (Table 4.8) however, the total area accessible, based on the water velocity criteria, increased from 55.7 m^2 to 89.9m^2 (Table 5.8). The majority of the quadrats available were classified as suitable for this age class both before and after the spate and in the total and wetted streambed (Table 5.7).

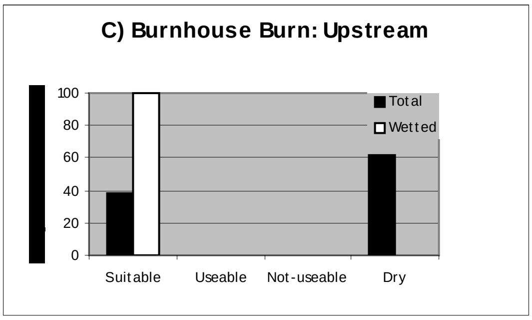
5.3.2.3 Juvenile water velocity (fish length: >7 to 20 cm)

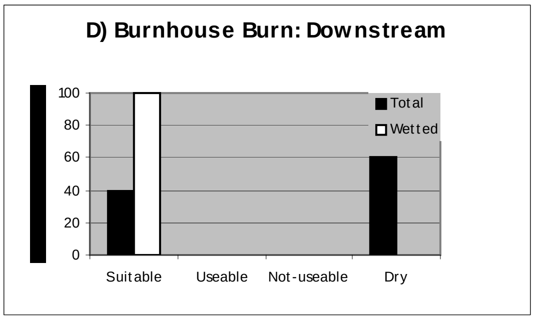
The chi-square test reveals that statistically there is a relationship between site and the velocities available to juvenile brown trout (length of >7 to 20 cm), classified based on the three suitability categories and including dry streambed when examining the total streambed. The chi-square values for both the total and wetted portions of the streambed were X^2 = 306.4, df = 18, p = 0.000 and X^2 = 121.0, df = 12, p = 0.000, respectively [Table 5.5 and Figure 5.10(A-G)]. Almost all of the velocities observed were classified as either suitable or useable by juvenile trout in these reaches (Table 5.7). In the total streambed suitable velocities ranged from 36.5% at the Bin Burn-downstream (post-spate) site to 63.9% at March Burn-upstream. The greatest proportion of useable velocities was found at March Burn-upstream (31.2%) and there was no useable habitat at either the upstream or downstream sites on Burnhouse Burn. Quadrats classified with water velocities considered not-useable by juvenile trout were scarce. In fact all sites had no quadrats with this classification with the exception of Bin Burn-upstream, which had 0.6% of the water velocity measures, being categorized not-useable.

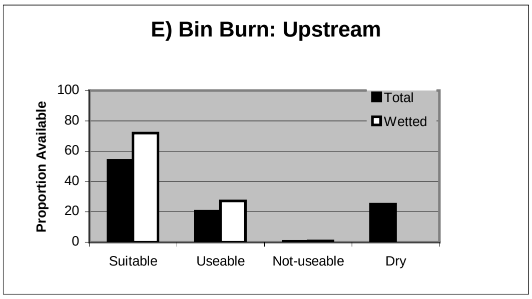
The relative proportions of suitable, useable, and not-useable velocities are more pronounced when examining the wetted streambed alone. All the quadrats were classified as suitable in both the up- and downstream sites on Burnhouse Burn. The smallest proportion of suitable velocities was found at March Burn-upstream (67.2%). Clearly, none of the quadrats at the Burnhouse Burn sites are classes 'useable' while 32.8% of March Burn-upstream bears this classification. The remaining sites have intermediate proportions of useable velocities. With the exception of Bin Burn-downstream (0.8%) there were no quadrats classified as not-useable.

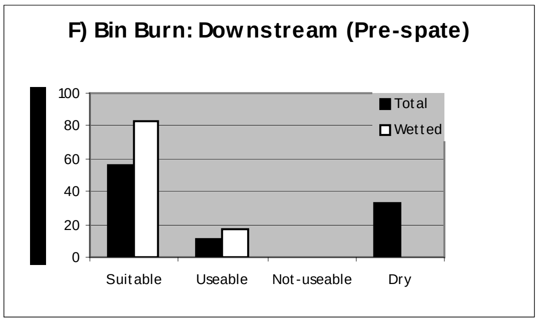












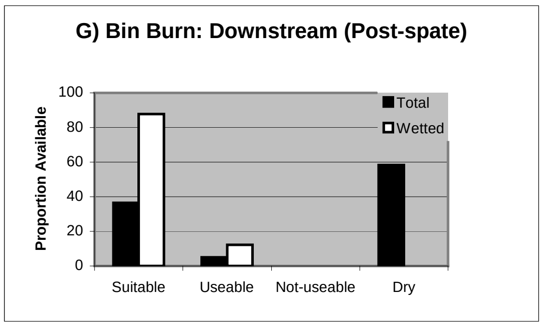


Figure 5.10 (A-G): Histograms of the proportion of measures of velocity available to juvenile trout (> 7 cm to 20 cm) at all study sites classified as suitable, useable, not-usable and dry for both the total and wetted streambed.

The chi-squared analysis suggests that the proportions of the water velocities classified based on the suitability criteria for juvenile trout are different when examining the total streambed (X^2 = 61.5, df = 3, p = 0.000) however, these proportions are not different when looking at wetted streambed (X^2 = 3.8, df = 2, p = 0.149). The proportion of accessible quadrates (quadrats with velocities classed as suitable or useable) were 66.8% and 100% for the total and wetted streambed before the spate and 41.6 and 100% after the spate. The accessible (sum of suitable and useable) area before the spate was 59.4 m^2 and 90.1 m^2, after. Although the relative proportion of accessible water velocities decreased after the spate the total surface area available increased.

5.3.2.4 Adult non-spawners water velocity (> 20 cm)

The chi-square test comparing the relative proportions of the suitability criteria with sites revealed that there is a relationship between site and water velocity available to brown trout. The chi-squared values for both the total and wetted streambed were as X^2 = 275.6 (df = 18, p = 0.000) and X^2 = 90.2 (df = 12, p = 0.000), respectively (Table 5.5). The trend noted in the juvenile trout summary towards an increasingly higher proportion water velocities being classified as suitable continues with the larger fish [Table 5.7 and Figure 5.11(A-G)]. In fact only 3.5% of the quadrats surveyed at Bin Burn-upstream and 0.6% at March Burn-downstream were classed as not-useable when examining the entire streambed. There were no quadrats classified as not-useable at the other sites when looking at the total streambed. Suitable velocities ranged from 38.6% at Burnhouse Burn-upstream to 81.8% at March Burn-upstream. Quadrats classified as useable varied from 0% at both Burnhouse Burn sites to 14.2% at Bin Burn-upstream.

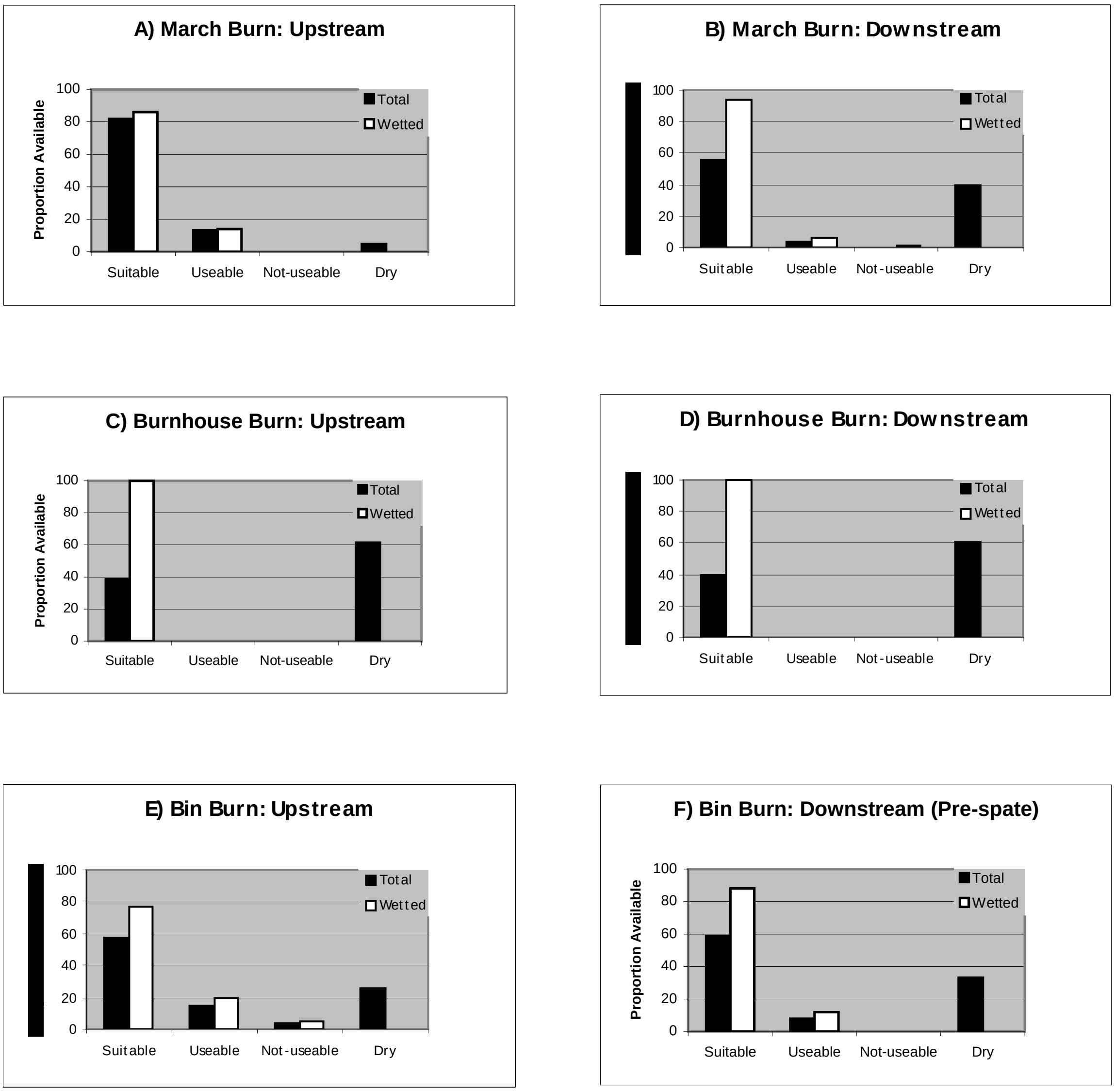
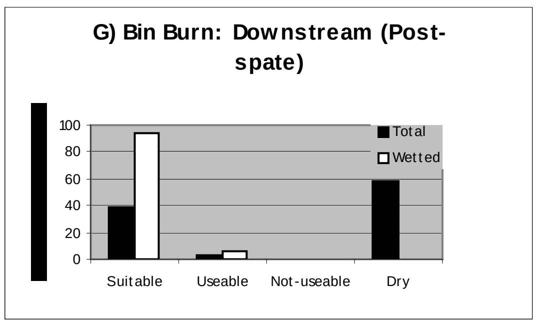


Figure 5.11 (A-G): Histograms of the proportion of measures of velocity available to adult trout (> 20 cm) at all study sites classified as suitable, useable, not-unusable and dry for both the total and wetted streambed.



When examining the wetted portion of the study reaches suitable water velocities for adult trout were observed in 74.4% (Bin Burn-upstream) to 100.0% (Burnhouse Burn up- and downstream) of the quadrats surveyed. Useable water velocities were found in 0% (both Burnhouse burn sites) to 19.0% of the quadrats. Very infrequently were water velocities encountered that were not-useable by adult trout. All water velocities measured at March Burn-upstream, both Burnhouse Burn sites, and Bin Burn-downstream both before and after the spate were considered useable or suitable. Only 1% of the study reach at March Burn-downstream and 4.6% of the wetted area surveyed at Bin Burn-upstream was considered not-usable.

The area available with adequate water velocities for adult trout (proportion of suitable and useable summer multiplied by the wetted area of the reach) was essentially the same as that observed as that observed for the juvenile classifications. The Burnhouse Burn sites (up- and downstream) had the lowest accessible area (11.6 m^2 and 24.9 m^2, respectively) the downstream Bin Burn sites had the highest (59.4 m^2 and 90.1 m^2, pre- and post spate respectively and March Burn and Bin Burn-upstream had intermediate values (Table 5.8)

Lastly, the chi-squared analysis revealed that the proportions of habitat classified based on the suitability criteria are different at Bin Burn-downstream before and after the spate. The Chi-squared value comparing the entire streambeds is $X^2 = 64.6$ (df = 3, p = 0.000) and the chi-squared value for the wetted portion of the streambeds is $X^2 = 6.9$ (df = 2, p = 0.031). The proportion of the streambeds accessible (suitable plus useable) before the spate for the total and wetted streambed were 66.8% and 100.0% (total and wetted, respectively) and 41.6% and 100.0% after the spate (Table 5.7). Although the proportion of accessible habitat in the total streambed decreased after the spate the surface area actually available to adult trout increased from 59.4 m^2 to 90.1 m^2 (Table 5.8).

5.3.3 Habitat maps

A graphical representation of each stream and the habitat available for spawning (Figure 5.12), nursery (Figure 5.13), juvenile (Figure 5.14) and adult size classes (Figure 5.15), was produced. The boundary between the stream channel and the streamside is clearly differentiated as the streamside segments are coloured black. The wetted area, or stream course within the stream channel, can be distinguished from the dry streambed as shaded and clear quadrats, respectively. Using these coding a basic description of each stream was produced.

Once again, habitat maps provide a good representation of the habitat available in the study reaches and a sequential review of these figures clearly demonstrates trends in habitat availability as fish age, from the perspective of water velocity. Figure 5.12 illustrates that the best spawning water velocities are scattered throughout the wetted portion of the streambed but tend not to be associated with the pool area in the upstream portion of the reach. This confirms what would be expected, as trout are less likely to spawn in slow moving deeper waters of a pool. The best water velocities for spawning trout seemed to be more evenly distributed throughout the downstream Burnhouse Burn reach. These figures also confirm the trends noted in the tables and histograms: that the water velocities present in these streams seems to be well suited for all age classes but particularly so as fish age. Further the juvenile and adult age classes are similar in that the majority of the quadrats surveyed were suitable for the larger trout and bulk of the

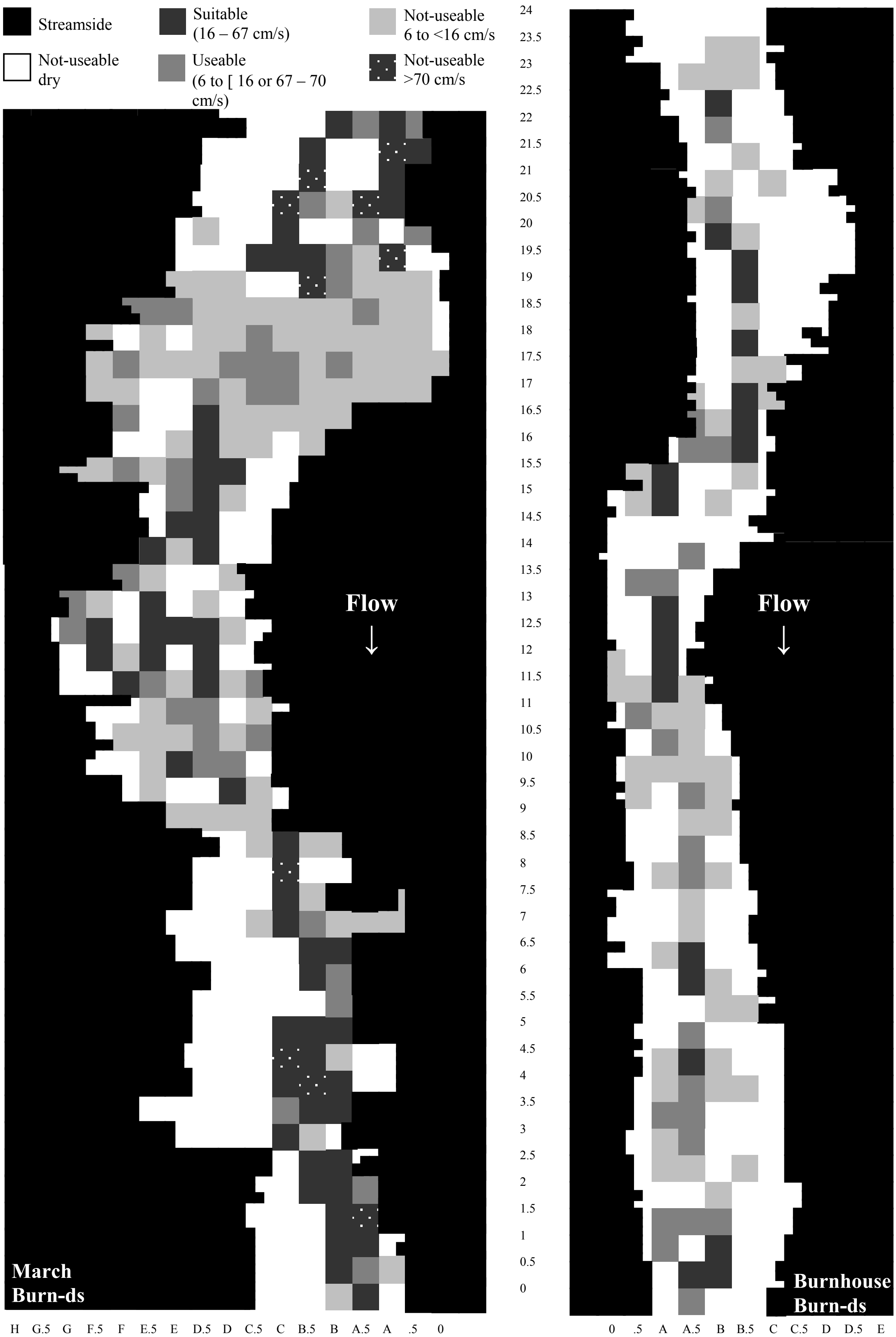


Figure 5.12: Water velocity available for spawning brown trout (*Salmo trutta*) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on velocity requirements.

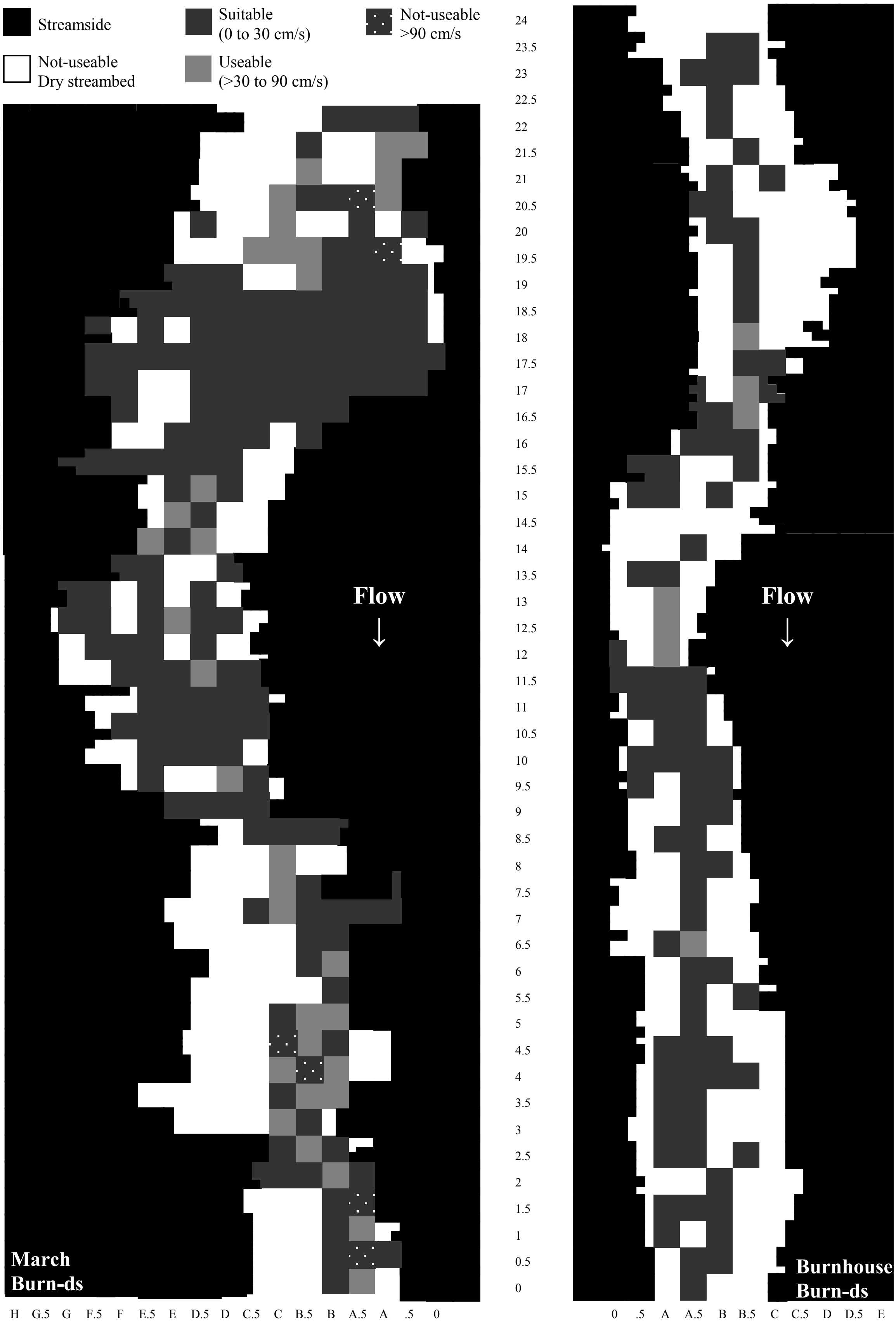


Figure 5.13: Water velocity available for brown trout fry (*Salmo trutta*; length ≤ 7 cm) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on velocity requirements.

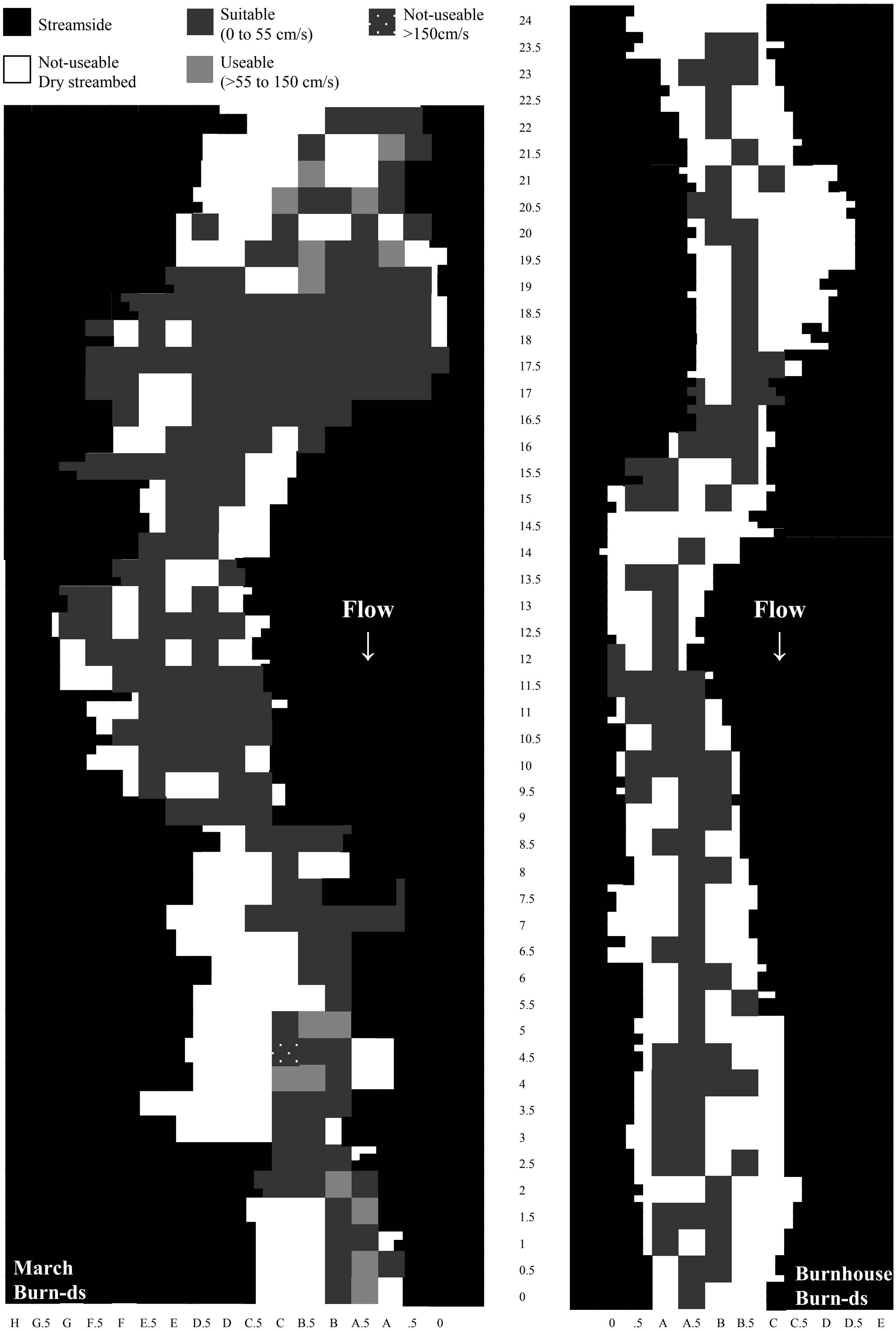


Figure 5.14: Water velocity available for juvenile brown trout (*Salmo trutta*; length >7 to 20 cm) in March ($Q = 0.341\ m^3/sec$) and Burnhouse Burn ($Q = 0.0058\ m^3/sec$) based on velocity requirements.

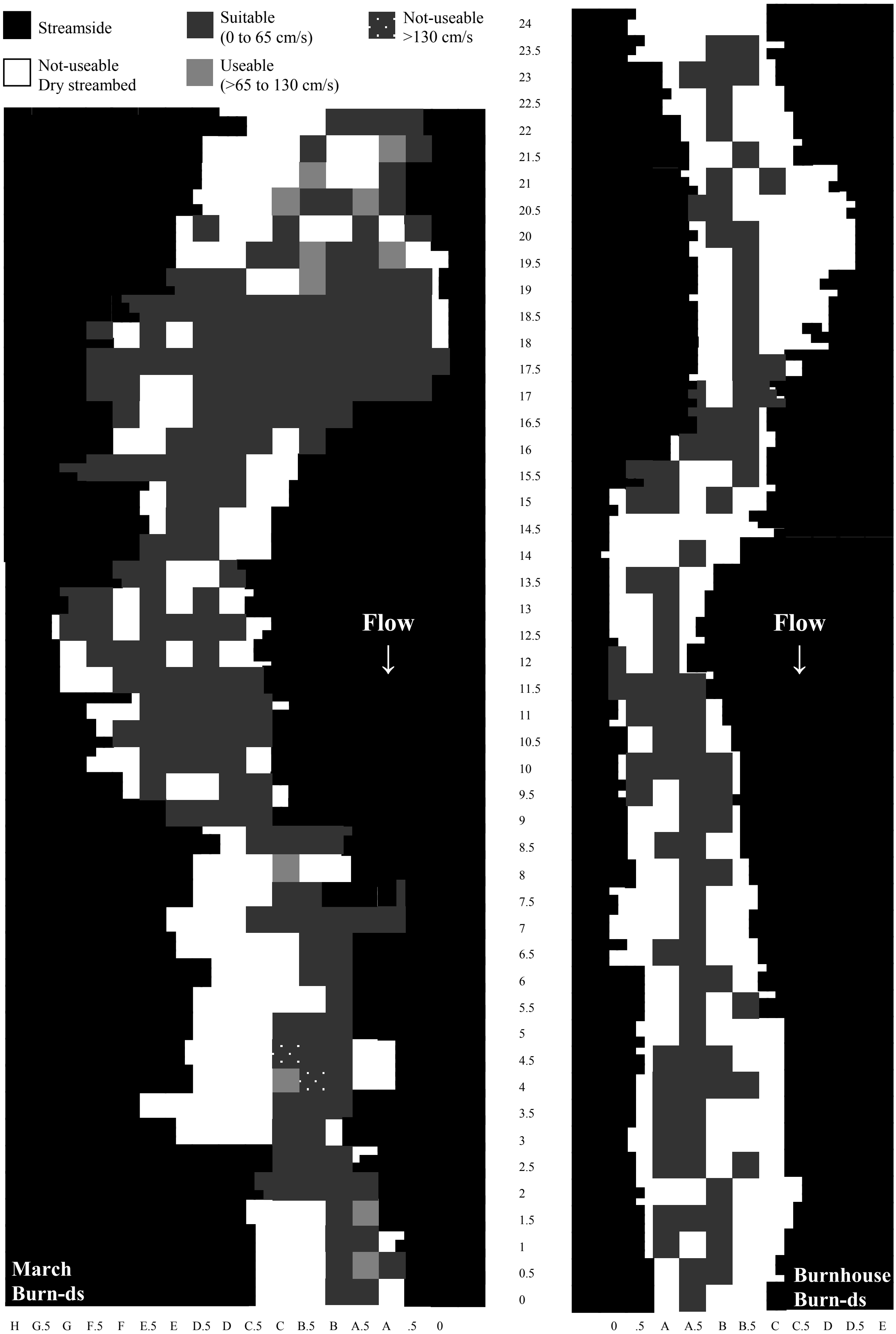


Figure 5.15: Water velocity available for adult non-spawning brown trout (*Salmo trutta*; length >20 cm) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on velocity requirements.

remaining quadrats were useable. Very little if any of the water velocity observed in the study reaches could not be used by juvenile or adult trout.

5.4 Discussion

The water velocity recorded in burns examined in the Carron Valley in the late summer of 2002, are well suited for resident brown trout of all age classes as well as adult spawners. The specific physical characteristics of each stream differ, and the relative proportions of the suitability criteria are statistically different, yet similar conclusions can be drawn regarding the habitat that is available for use by brown trout. These conclusions were based on the construction of size-class specific tolerance profiles complied from a wide range of literature sources that can be applied 'universally' to any lotic system that formally or currently host's populations of *Salmo trutta*.

Interestingly, from the perspective of water velocity, the proportion of the wetted area available for use increases with the size of the resident trout. This is the inverse of the trend that was noted for the analysis of depth in Chapter 4. As well, it is contrary to what would be expected based on the fish surveys conducted in these streams. As previously discussed (Chapter 4) the bulk of the brown trout observed were small and could be classified as fry (≤ 7 cm in length) or small juveniles. The resident fish observed had a mean lengths ranging from 3.7 cm to 6.5 cm in Burnhouse Burn, 9.0 cm to 12.0 cm in March Burn, and 4.8 cm to 12.5 cm in Bin Burn. As well a spawning run was observed in early-December 2002 in Burnhouse Burn-downstream, which contained the least amount of useable spawning velocity during the late summer survey.

Although the dry portion of the streams ranged from approximately 5% to 60% during the survey conducted in July and August of 2002 a greater proportion of the streambed may have been wet during early December when trout were observed spawning. The discharge data outlined in the Appendix (Figure A.1 to A.6) indicates that these streams have a highly variable discharge and that the proportion of the wetted area available in December is most likely different, and possibly greater, than that observed in the late summer. However, any conclusions regarding habitat availability in the winter are purely speculative without a detailed habitat assessment. However, the fish survey conducted in December of 2002 (Table 4.10) resulted in the discovery of adult spawners in the accessible burns (Burnhouse and March Burn-downstream).

The figures sin the Appendix (A.1 to A.6) indicates highly variable discharge often occurring in the winter months. These high discharge periods may coincide with low ambient temperatures. Temperature are important to stream dwelling fish as they cannot maintain their position in the water column above certain velocities (Tetzlaff et al. 2005b) and their susceptibility to displacement is thought to be due in part to its size (body length) and the ambient water temperature (Graham et al. 1996). The relationship between body length and temperature for Atlantic salmon has been examined in flume experiments and had been described mathematically as the critical displacement velocity (CDV) (Graham et al. 1996, Tetzlaff et al. 2005b). The equation for this relationship is as follows:

$$CDV = CDV_{BL} * L/100 \text{ with } CDV_{BL} = 4.14 \log T + 1.74 \text{ (Atlantic salmon)}$$

Where CDV_{BL} is expressed in body lengths per second, T = water temperature (^{o}C) and L is fish body length (cm).

CDV represents the maximum sustained velocity against which a fish can hold position. When the stream velocity exceeds the CDV, the opportunity to feed is likely to be inhibited which can result in decreased growth rates and weight loss. Alternately discharges that exceed CDV may result in fish being swept downstream. The absolute value of the displacement value is dependent on fish size and water temperature. Larger fish are generally able to withstand greater velocities than smaller ones and CDVs are lowered for all age classes as temperature falls (Tetzlaff et al. 2005a, Tetzlaff et al. 2005b). Temperatures were not recorded during this study. However, the calculation of CDV would be very useful, particularly during periods of high discharge (high water velocities) so that areas of velocity refuge might be identified.

It seems counter intuitive that large fish are more suited to the available velocities in such small streams, particularly in light of the fish survey data that indicates that these streams do not possess populations of resident adults or large juveniles. It is important to point out that the tolerance profiles display a range of velocities that can be used by brown trout. The streams aren't necessarily better suited for larger fish; rather the larger fish can use a broader range of the water velocities that are present in the streams studied. It should also be repeated that salmonids select microhabitat based on multiple factors (Dare and Hubert 2000) not just water velocity, therefore, the lack of large resident trout in these streams is more likely due to the restrictions imposed by other environmental requirements (e.g. water depth). A more thorough discussion of interactions between physical parameters and trout microhabitat selection and preference will be undertaken in Chapter 7.

The broad tolerance profiles of larger trout, relative to the smaller size classes, implies that smaller trout are more sensitive or more vulnerable to limitations in appropriate velocities. Trout fry have a narrower tolerance profile than older larger fish thus are capable of utilizing a smaller portion of the velocity available in any given stream. This observation, coupled with the competitive exclusion that restricts access of small trout to deeper more energetically profitable areas indicates that smaller fish have less access to the preferential habitat than larger conspecifics. This suggests that the physical environment is a hostile place for young trout. There is some support in the literature for this supposition (Mortensen 1977, Egglishaw and Shackley 1980). Young trout are often located in stream margins and other shallow and slow moving environments (Karlstrom 1977) sometimes described as 'marginal' (less suitable) habitat (Eklöv et al. 1999). As well, immature salmonids, including those at the swim-up-stage, are vulnerable to downstream displacement resulting from high flows (Ottaway and Forrest 1983, Heggenes and Traaen 1988) especially if they are combined with low temperatures (see above). Finally, Bachman (1984) reminds us that preference for site selection is made based on energetic considerations [see also Bremset and Berg (1999)]. He points out that larger trout occupy smaller preferable home ranges forcing younger smaller conspecifics into less desirable foraging areas resulting in greater movements (larger home ranges), which are energetically costly.

Once again, as discussed in Chapter 4, the tolerance profiles created are limited in that data from the full range of brown trout's worldwide distribution could not be obtained. Further work is necessary to fill these gaps and the specific boundaries in the tolerance profiles may change. However, I believe that the fundamental structure of the profiles will be consistent as the set of data used in their construction grow. Again, it is unlikely, that with additional information either of these streams will be re-classified from nursery streams to ones that should support large resident trout. There are common issues that will become apparent with the development of the substrate composition tolerance profiles that are similar to issues that have arisen both in the discussion of velocity and stream depth. Examples include inconsistent sampling methodologies, quantitative assessment of data, the use of quadrats, the method chosen to display the data and assess habitat availability and a discussion on habitat selection based on multiple physical factors at varying discharges is warranted. These common issues as well as the development and integration of these physical variables will be discussed in Chapter 7and 8.

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6.0 Substrate

6.1 Introduction

Substrate is one of the four most frequently studied physical habitat parameters that is thought to influence the distribution of brown trout (*Salmo trutta*) living in riverine ecosystems. As previously mentioned, the other three habitat variables of importance are water depth (discussed in Chapter 4), water velocity (discussed in Chapter 5), and cover (Heggenes 1988c). The relative importance placed on these four habitat components is inconsistent. In a range of studies, workers have suggested conflictingly that depth (Bohlin 1977, Egglishaw and Shackley 1982, Kennedy and Strange 1982, 1986, Heggenes 1988a, b), water velocity (Shirvell and Dungey 1983, Bachman 1984, Gatz et al. 1987), cover (Baldes and Vincent 1969, Mortensen 1977, Fausch and White 1981), substrate (Karlstrom 1977, Gatz et al. 1987) or a combination of these variables (Karlstrom 1977, Bagliniere and Champigneulle 1982, Gatz et al. 1987) as being the most important habitat feature to influence the distribution and abundance of salmonids. Regardless, the availability of adequately sized bottom substrate, particularly at crucial and vulnerable periods in the life cycle of brown trout, such as spawning and for young-of-the-year, is essential for the maintenance of viable populations of trout and other salmonids (Mäki-Petäys et al. 1999).

Streambed substrate functions mainly as a shelter both from high water velocities and as cover from predators (Jenkins 1969, Heggenes 1988c, Heggenes et al. 1996). The position within a stream occupied by salmonids is based on the suitability and availability of appropriate substrate, as well as other physical, chemical, and biological conditions, and is selected on the microhabitat scale (Baldes and Vincent 1969). The microhabitat chosen varies with photoperiod, season, fish age, and with the level of inter and intra-specific competition (Bohlin 1977, Cunjak and Power 1986, Hayes 1987, Heggenes 1988d, Kocik and Taylor 1996, Elso and Greenberg 2001, Vismara et al. 2001). Based on the interplay between these factors, the substrate accessible for use by trout may be classified based on its suitability both at the various life stages and for specific purposes such as spawning. By defining a range of suitable or preferred streambed substrate for brown trout, as a subset of the complete range available, reference conditions can be defined which can be used as the basis of environmental-quality monitoring and assessment.

The objectives of this chapter are 1) the development of size-class specific tolerance profiles for streambed substrate based on published observations of *Salmo Trutta* habitat use and 2) the application of these profiles to the streambed substrate observed at the six sites (expressed in seven data-sets). This analysis will be conducted in order to determine the effectiveness of the technique and to demonstrate how the methodology can be used to assess the streambed substrate component of the physical habitat available to brown trout in running waters. The remainder of the introduction will outline the requirements that brown trout have for substrate in streams.

6.1.1 Streambed Substrate Requirements: Overview

A common generalization is that brown trout seem to prefer coarse substrates such as cobble and are less abundant in finer bottom substrates (silt, sand, and fine gravel) or smooth bedrock (see Figure 6.1 as a key to substrate sizes) (Heggenes 1988c, Haury et al. 1999, Heggenes et al. 1999, Armstrong et al. 2003). More specific substrate particle sizes are required by brown trout when spawning. When excavating spawning sites

Substratum type	Size class (mm)
Organic fine materials	
Organic coarse materials	
Clay, silt	0.004-0.06
Sand	0.07-2
Fine gravel	2.1-8
Gravel	8.1-16
Small pebble	16.1-32
Pebble	32.1-64
Small cobble	64.1-128
Cobble	128.1-256
Large cobble	256.1-384
Boulder	384.1-512
Large boulder	> 512.1
Rough bedrock	
Smooth bedrock	

Figure 6.1: Modified Wentworth scale after Bain et al. 1985 used for stream substrate particle size classification. Redrawn from Table II in Heggenes and Saltveit 1990.

(redds) females utilize streambed areas with stone or gravel bottoms. There is some variation in the size of substratum used as larger females will spawn on coarser substrates and bury their eggs deeper (Klemetsen et al. 2003). The creation of redds involves the creation of a pit in the substrate within which eggs are deposited. This is done repeatedly in a upstream succession with excavated material loosened in the construction of upstream pits covering eggs deposited in the downstream depressions (Crisp 1993). This process removes fine materials (silt and sand) which have been associated with decreased survival of eggs and alevins as this material reduces stream-flow within the redd, resulting in the build-up of toxic metabolites and asphyxiation due to decreased oxygen levels (Grost and Hubert 1991, Crisp 1996). Fine sediment has also been associated with embedding stream substratum, limiting the ability of spawners to dislodge materials during the excavation process (Zeh and Dönni 1994) as well as restricting the movement of alevins during 'swim-up' (Crisp 1993). The base of the excavated pits have been shown to contain relatively large substrates which are thought to increase porosity and enhance survival of offspring (Barlaup et al. 1994). The ideal site for spawning has been described as the tail end of a pool as it merges into a riffle; though, this has been contested (Ottaway et al. 1981). Overly large substrate can also be problematic as it allows access to predators, (Rubin et al. 2004) and females may have difficulty excavating a redd when large, immovable substrate is present.

When the eggs have hatched and the bulk of the yolk sac has been consumed young trout swim-up through the gravel and establish feeding territories in the vicinity of the spawning area (Klemetsen et al. 2003). For newly emerged and older resident trout substrate has three main functions. The first of these is to act as cover or shelter from predation. The spaces or interstices between rocks, cobbles, and boulders can be used as hiding places for young-of-the-year and juvenile trout so long as they are of adequate size for fish to utilize (Bachman 1984). Secondly, interstices and a heterogeneous or rough streambed provided by larger particles sizes can provide a refuge during high or extreme velocities. This is particularly important during the young stages of salmonid development as fry are susceptible to downstream displacement (Ottaway and Forrest 1983, Heggenes and Traaen 1988, Pender and Kwak 2002). During periods of low or moderate flow interstices and large stones can provide profitable focal positions as they often offer low velocity areas in close proximity to high flows which carry drifting prey items allowing for net energy maximization (Bachman 1984, Fausch 1984, Hayes and Jowett 1994). Finally, as trout are territorial (Raleigh et al. 1986) large stones, cobbles, or boulders provide a heterogeneous streambed, which provides increased visual isolation between neighbouring fish thereby reducing aggression, increasing the number of potential territories within a given area and presumably increased fry density (Heggenes 1988c).

There is an obvious relationship between trout size and the ability of these fish to use habitat features such as interstices and substrate particles. All trout can use small streambed substrates (silt, sand, and gravel) (Heggenes 1996), however, many authors report that trout within any age classes prefer the coarsest substrates available (Armstrong et al. 2003). Clearly, as fish grow larger there ability to fit into interstices and isolate themselves behind large stones and rocks decreases and their requirements for larger substrates increases (Greenberg et al. 1996). As well, trout exhibit diurnal and seasonal changes in their use of streambed substrate. The activity of salmonids is reduced as temperature fall resulting in restricted mobility and increased susceptibility to predation (Armstrong et al. 2003) (see Section 5.4, CDV). As autumnal temperatures decrease salmonids become increasingly nocturnal hiding in crevices and under stones during the day (Heggenes and Saltveit 1990, Heggenes et al. 1993, Cunjak 1996, Bremset 2000). Mäki-Petäys (1997) found that during the summer trout used substrate classes roughly in relation to their availability while in autumn and wintertime there was a shift in usage towards the larger substrates. Brown trout and other salmonids have been observed burying themselves in the stony substrates of streams during winter presumably to avoid predation, as a mechanism to minimize energy expenditure (Raleigh et al. 1986, Griffith and Smith 1993) and as a thermal refuge as these spaces may be warmed by groundwater intrusions (Mäki-Petäys et al. 1999).

6.2 Results

6.2.1 Tolerance profiles

6.2.1.1 Spawning substrate

The literature used in the constructing of tolerance profiles of the streambed substrate used by brown trout is quite diverse. The particle size used for redd construction has been examined in New Zealand (Shirvell and Dungey 1983), Canada and the United States (Witzel and MacCrimmon 1983, Grost et al. 1990, Beard and Carline 1991, Essington et al. 1998, Pender and Kwak 2002), and Europe (Ottaway et al. 1981, Nihouarn 1983, Fragnoud 1987, Heggberget et al. 1988, Rubin et al. 2004). When reported, all of the studies involved sympatric population of salmonids; however, Witzel and MacCrimmon (1983) examined sites with allopatric populations as well. The other salmonids involved in these investigations include Atlantic salmon (*Salmo salar*) (Ottaway et al. 1981, Heggberget et al. 1988), brook trout (*Salvelinus fontinalis*) (Witzel and MacCrimmon 1983, Grost et al. 1990, Essington et al. 1998), and rainbow trout (*Salmo gairdneri*) (Shirvell and Dungey 1983). The influence of a non-salmonid species Ozark sculpins (*Cottus hypselurus*) on brown trout reproductive success was examined by Pender and Kwak (2002). Particle size of spawning sites were determined using three different methods which included visual assessment (Grost et al. 1990, Essington et al. 1998, Pender and Kwak 2002), core sample analysis (Ottaway et al. 1981, Shirvell and Dungey 1983, Witzel and MacCrimmon 1983, Beard and Carline 1991, Rubin et al. 2004), and direct instream measures (Heggberget et al. 1988). To be as comprehensive as possible, if otherwise unavailable, some of the values reported for spawning depth have been extracted from a literature review by Haury and his colleagues (1999). These include the values reported by Fragnoud (1987) and Nihouarn

(1983). Limited information is available about the specific characteristics of these investigations. The literature used in this synopsis is reported in detail in Table 6.1 and summarized in Figure 6.2.

The type of data used for the assessment of streambed substrate use by *S. trutta* differs from that used in the assessment of water depth and velocity in that information reported is an estimation a dominant or mean particle size of a composite of material rather than a discrete measure at a point in time. Visual assessments report the dominant substrate type while core samples processed using gravimetric analysis will report a statistic that describes the central particle size. It is important to remember when reviewing this information that substrate being described most likely contains a wide range of particles even if some size classes are present in small proportions. Using the various methods the dominant or central particles sizes in the redds observed range from sand to large boulders. The low values reported come from particle size analysis (core samples) and were reported by Rubin and his colleagues (2004) in a stream in Switzerland. Although uncommon, spawning in sandy substrates is not unprecedented as Crisp and Carling (1989) report occasionally observing trout spawn in coarse sand substrates. The largest substrate, large boulders (≅ 30 cm), was observed at about 1% of the spawning sites visited and is quite unusual. Generally the largest dominate substrate useable for spawning is cobble, again confirmed by Crisp and Carling (1989), which has a diameter from 12 to 25 cm; the upper limit being related to the size of the female.

The standardization of the data generates bands of suitability based on the substrate sizes outline in the modified Wentworth scale (Heggenes and Saltveit 1990). The full range of spawning substrates is rarely reported. Typically, in visual assessments, a

Table 6.1: Literature used to define the range of substrate sizes that can be used by brown trout (*Salmo trutta*) to spawn.

Source	Substrate size *cm* Diameter; mean±SD (SE) [range]	No. of redds or spawners	Preferred Substrate *cm*	Category range *cm*	Assessment type *cm*	Fish size; mean ±SD (range) *cm*	Study Location	Allo - sympatric Popn's	Other species	Notes
Beard & Carline 1991	-	226	0.4 to 6.4	-	Core: 8 sieve sizes	20->30	USA Pennsylvania	Allopatric	-	87% of weight of substrate in 'preferred' range
Essington et al. 1998 Subsection 1 Subsection 2	-	108	0.3-2 0.3-2	-	Visual 3 categories	-	USA (Minnesota)	Sympatric	Brook trout	81% & 59% of substrate in 'preferred' range in SS1 and SS2 r
Fragnoud 1987	0.2-6.4	620	-	-	-	30	France (Eastern)	-	-	Cited in Haury et al. 1999
Heggberget et al. 1988 large rivers small rivers	6.6±1.9 8.1±2.7	36 125	-	-	Diameter of 10 stones (dominant size class)	0.5 kg (Alta) 0.9 kg (Gaula) 2.1 kg (Driva) 0.8 kg (Eira)	Norway	Sympatric	Atlantic Salmon	Large (Alta, Gaula & Driva pooled); Small (Eira)
Grost et al. 1990	-	80	2.6 to 7.5	(0.7-2.5 to >30)	Visual: 5 categories	20-40	USA (Wyoming)	Sympatric	Brook Trout	60% of total substrate in 'preferred' range
Nihouarn 1983 Width (7-21m) Width (1.5-3m)	2-5 0.2-2	38 58	-	-	-	-	France (Eastern)	-	-	Cited in Haury et al. 1999
Ottaway et al. 1981 Gr't Eggleshope Beck	6.5 (3.8)*	24	-	-	Core: grain size at 1/3 phi (φ) unit intervals	(25.6-34.5)	England (northern)	Sympatric	Atlantic salmon	* S.D. estimated from S.E.
Pender and Kwak 2002 Beaver Bull Shoals 2	$6.4\text{-}13.0^{\alpha}$; $1.6\text{-}6.4^{\beta}$ $6.4\text{-}13.0^{\alpha}$; $1.6\text{-}6.4^{\beta}$	8 15	-	-	Visual: Modified Wentworth after Bovee & Milhous 1978	35-43	USA (Missouri, Arkansas)	Sympatric	Sculpins	α redd pit β redd tailspill
Rubin et al. 2004 GMD MD	1.9±0.7 [1.0-3.1] GMD 1.8±0.5 [1.1-2.8]MD	15 15	-	<0.1 to 3-12	Core: 8 categories	(18.5 - 89.0)	Sweden (Gotland)	-	-	GMD: geometric mean diameter MD: median diameter
Shirvell and Dungey 1983	1.4±0.6 [0.5-2.8]	140	-	<0.05 to > 3.2	Core: 8 categories	41 (32-55)	New Zealand	Sympatric	Rainbow trout	Analysis: Phi units and methods of moments

Table 6.1: Continued.

Source	**Substrate size *cm*** Diameter; mean±SD (SE) [range]	**No. of redds or spawners**	**Preferred Substrate *cm***	**Category range *cm***	**Assessment type *cm***	**Fish size; mean ±SD (range) *cm***	**Study Location**	**Allo - sympatric Populations**	**Other species**	**Notes**
Witzel and MacCrimmon 1983 GMD MD	 0.69±0.28 [GMD] 0.99±0.48 [MD]	 47 47	 0.08-3.2 [δ] -	-	Core: 14 categories	18.0-54.5 cm	Canada (Ontario)	Allopatric & Sympatric	Brook Trout	δ: middle 70% of the total weight Used mean±SD for preferred range

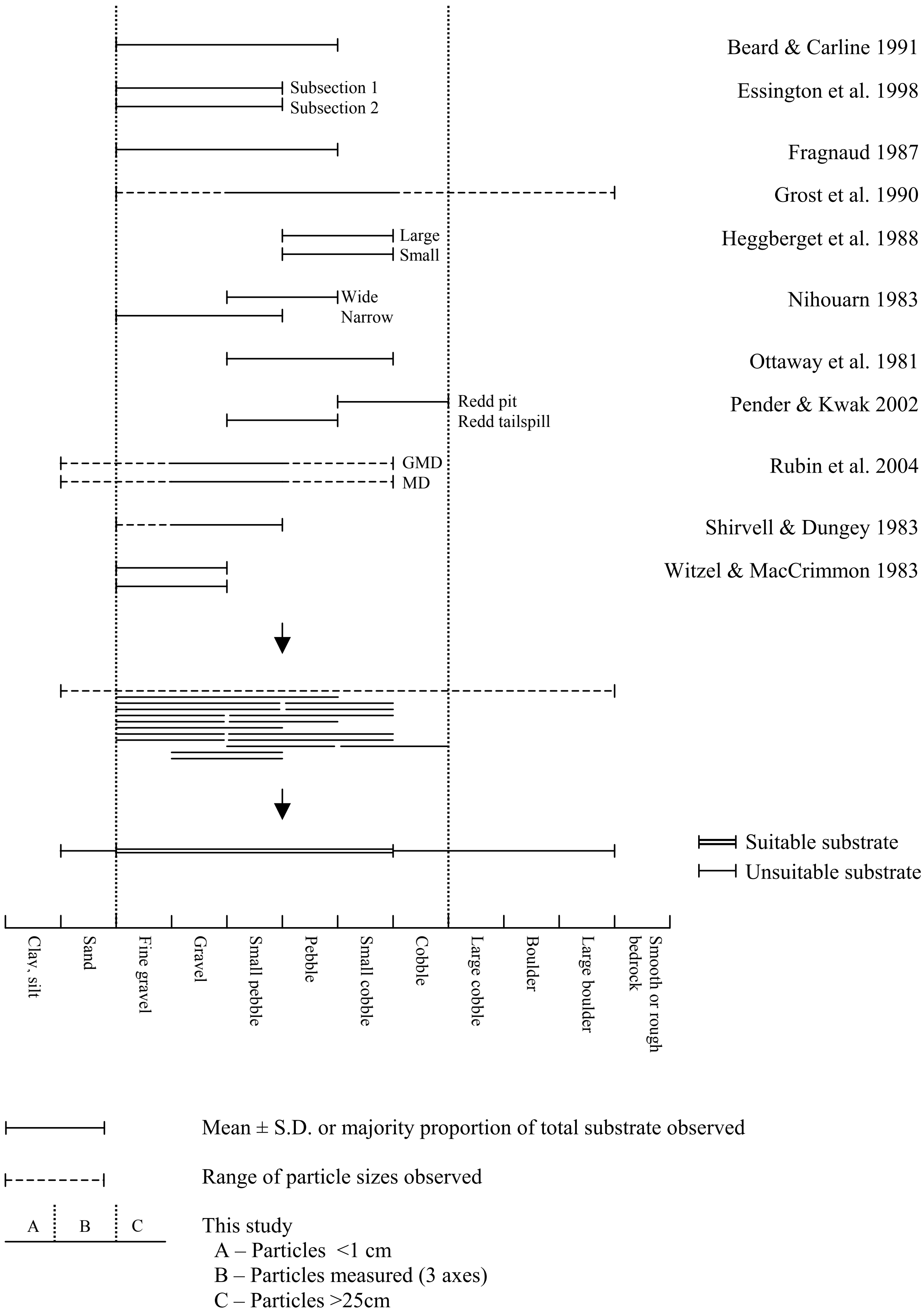


Figure 6.2: Procedure used to define the range of substrate particle sizes that can be used by brown trout (*Salmo trutta*) to spawn.

central proportion is determined that represents the dominant substrate type seen. With the gravimetric analysis (core samples) a central particles size is measured and the variance is estimated and reported as a standard deviation or standard error of the mean (see Table 6.1). In this instance, the mean plus or minus one standard deviation represents 68% of the substrate size used by trout to spawn. These bands of central tendency are used to distinguish the suitable and unsuitable regions within the tolerance profile. An examination of Figure 6.2 reveals that the vast majority of trout observed used spawning substrates that ranged in size from fine gravel to small cobble (particle sizes 0.21 to 12.8 cm). This range will be used as the suitable region within the tolerance profile. Pender and Kwak (2002) reports that at the spawning pit the substrate size range falls outside of this suitable range. The substrate dominating in the redd pit is classified in their study as 'small cobble' and range from 6.4 to 13.0 cm. This size range is reclassified as small cobble to cobble in the scales used in this summary. The upper limit of 13 cm falls towards the lower end of the cobble range in the standard scale. As well, the redd pit is more likely to contain larger stones that cannot be dislodged by the spawning female. This filtering process may result in substrate compositions larger than would be seen in either pre-spawning substrates or in the tailspill of the redd. For these reasons the suitable region of the tolerance profile was not extended to include the upper portion of Pender and Kwak's (2002) redd pitt substrate band.

In summary, the tolerance profile that will be used for brown trout spawning substrate is subdivided into two bands. The central band represents the suitable substrate that can be used for spawning and ranges from fine gravel (0.21-0.8 cm) to small cobble (6.41-12.8 cm). The second band (excluding the central suitable band) includes sand (0.0061-0.2 cm) and cobble (12.81-25.6) to large boulder (38.41-51.2) substrates and is considered unsuitable for spawning. All other substrates such as clay, silt, and bedrock cannot be used at all for spawning.

6.2.1.2 Nursery substrate (fish length ≤ 7 cm)

The distribution of the studies that have examined and reported on the use of substrate by brown trout fry is much narrower than that seen in the spawning substrate summary. These studies are restricted to studies conducted in northern Europe (Greenberg et al. 1996, Mäki-Petäys et al. 1997, Heggenes et al. 2002) and the United States (Harris et al. 1992, Hubert et al. 1994, Kocik and Taylor 1996, LaVoie and Hubert 1996, Pender and Kwak 2002). All studies involved sympatic populations of brown trout with the co-occurring salmonid species include Atlantic salmon (*Salmo salar*) (Heggenes et al. 2002), brook trout (*Salvelinus fontinalis*) (Harris et al. 1992, Hubert et al. 1994, Kocik and Taylor 1996, LaVoie and Hubert 1996), grayling (*Thymallus thymallus*) (Greenberg et al. 1996, Mäki-Petäys et al. 1997), and rainbow trout (*Salmo gairdneri*) (Kocik and Taylor 1996). Microhabitat position was determined using underwater (Greenberg et al. 1996, Kocik and Taylor 1996, Heggenes et al. 2002), surface (Harris et al. 1992, Hubert et al. 1994, LaVoie and Hubert 1996), and electro-fishing (LaVoie and Hubert 1996, Mäki-Petäys et al. 1997, Pender and Kwak 2002) methodologies. All investigation assessed substrate type using visual estimates and modified Wentworth classification systems. The literature used in this synopsis is summarized in Table 6.2 and the construction of the nursery tolerance profile for substrate is illustrated in Figure 6.3.

Small trout (≤ 7cm) have been reported using the full range of substrate particle sizes available [0.0004 to > 51.21 (bedrock)] (Greenberg et al. 1996, Heggenes et al. 2002). The smallest utalized substrate is sand (0.0061 –0.2 cm) (Kocik and Taylor 1996),

Table 6.2: Literature used to define the range of substrate sizes that can be used by brown trout (*Salmo trutta*) fry (≤ 7 cm).

Source	Number of fish sampled	Preferred Substrate (FH/SC)‡ *cm*	Category range (FH/SC) *cm*	Classification scale (FH/SC) *cm*	Fish size; mean ±SD (range) *cm*	Study Location	Allo - sympatric Populations	Other species	Survey method	Notes
Greenberg et al. 1996 Diurnal Dive Dusk Dive Stone Dive	 88 26 122	 1.6-6.4 - 6.4-38.4	 0.0004 to bedrock 0.0004 to bedrock 0.0004 to > 38.4	Modified Wentworth after Heggenes & Saltveit 1990	(2.5-6)	Sweden (Norrland)	Sympatric	Grayling	Underwater	
Harris et al. 1992 June July	 50 50	 <0.5 6-7	 <0.5 to >58 <0.5 to >58	After Bovee 1982	 2.6 3.9	USA (Wyoming)	Sympatric	Brook trout	Surface	
Heggenes et al. 2002	261	3.2 –25.6 (snout position)	0.0004 to smooth bedrock (>51.2)	Modified Wentworth	<7	England (southwest)	Sympatric	Atlantic Salmon	Underwater	
Hubert et al. 1994 June: day June: night July: day & night	 150 150 300	 <0.5 <0.5 -	 <0.5 to >58 <0.5 to 58 <0.5 to >58	Modified Wentworth after Bovee 1982	(2.1-3.4) June (2.3-4.8) July	USA (Wyoming)	Sympatric	Brook trout	Surface	
Kocik & Taylor 1996	65	0.0004 to 0.19	2.4×10^{-5} to 6.39	Modified Wentworth after Bovee 1986	5.3	USA (Michigan)	Sympatric	Rainbow /Brook trout	Underwater	
LaVoie & Hubert 1996 Backwaters Margins of pools Margins of riffles	 22 132 170	 <0.5 <0.5 <0.5-2.5	 <0.5 to 30.5 <0.5 to 22.8 <0.5 to 22.8	Modified Wentworth after Bovee 1986	5.1^{α} 5.6^{β}	USA (Wyoming)	Sympatric	Brook charr	Surface; electro-fishing	median total length for August (α) & September (β)
Mäki-Petäys et al. 1997 Midsummer Late summer Autumn Winter	 33 43 26 20	 6.4-12.8 12.8-25.6 12.8-25.6 12.8-25.6	 3.2 to >25.6 1.6 to >25.6 3.2 to >25.6 3.2 to >25.6	Wentworth modified from Heggenes 1988	4-9	Finland (northeastern)	Sympatric	Grayling	Electro-fishing	
Pender & Kwak 2002 Beaver Bull Shoals 1 Bull Shoals 2	 24 96 34	 $0.006\text{-}0.1^{\chi}$ $6.4\text{-}13.0^{\chi}$ $0.006\text{-}0.1^{\chi}$	-	Modified Wentworth after Bovee & Milhous 1978	-	USA (Missouri, Arkansas)	Sympatric	Sculpins	Electro-fishing	χ mode

‡ Frequency Histogram, Suitability Curve

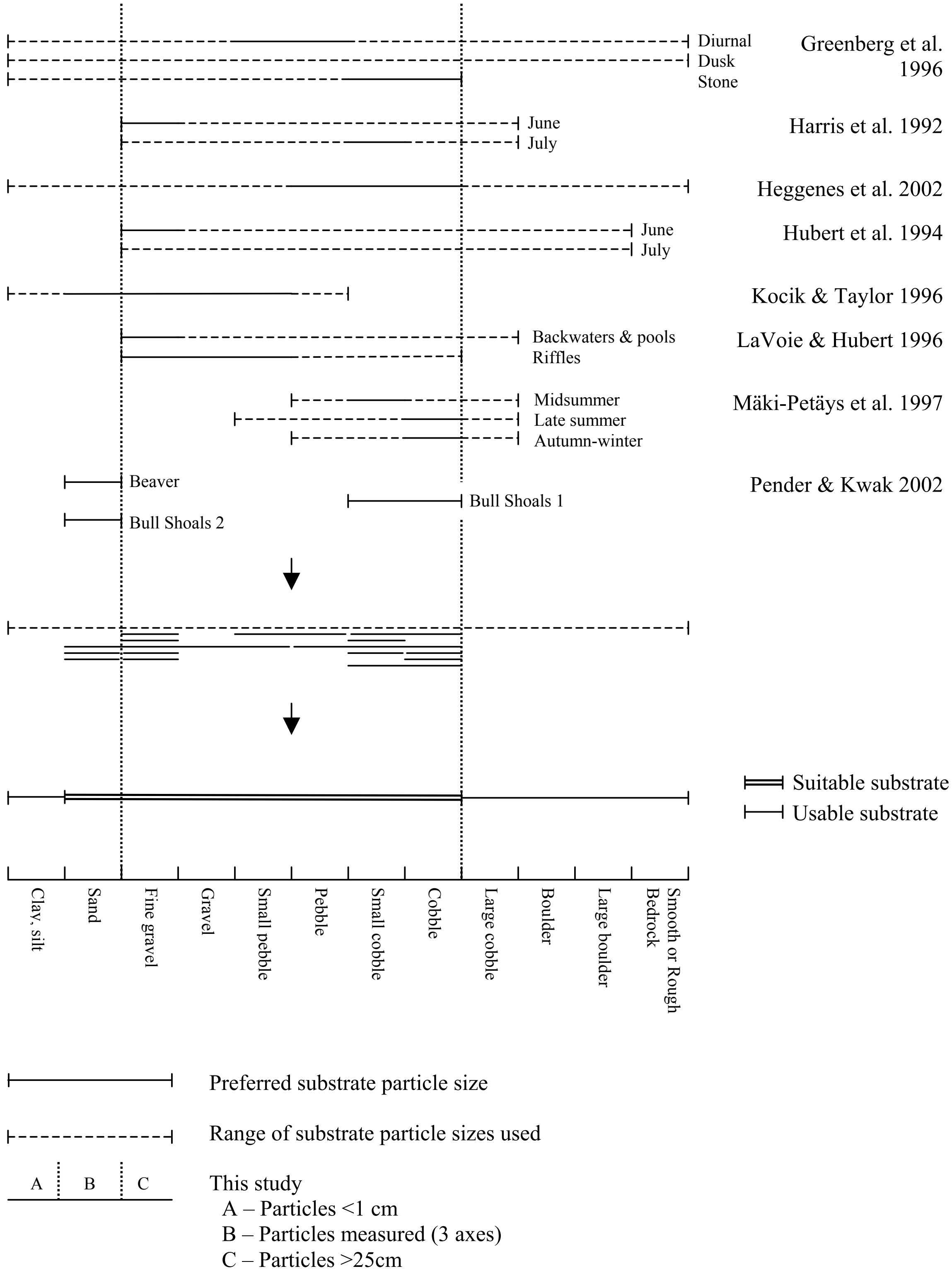


Figure 6.3: Procedure used to define the range of substrate particle sizes that can be used by brown trout (*Salmo trutta*) fry (≤ 7 cm).

while Pender and Kwak (2002) reports the mode of the particle size distribution used by small trout to be within this range at two sites. The maximum preferred substrate size is cobble (12.81-25.6 cm) in a number of studies including Greenberg et al. (1996), Mäki-Petäys et al. (1997), Heggenes et al. (2002), and Pender and Kwak (mode; 2002). Thus the preferred or suitable substrate ranges from sand to cobble while clay-silt, and larger substrates (large cobble to bedrock) are still useable by young trout. This range will be used as the tolerance profile for substrate utilization for brown trout fry (Figure 6.3).

Interestingly, the preference bands seem to form two clusters one for sand and fine gravel sized and another at the opposite end of the 'suitable' region of the tolerance profile consisting of small cobble and medium sized cobble. The reason for this bi-modal distribution is difficult to determine from this survey. However, some workers have noted a change in position of small trout from day to night (Heggenes et al. 1993, Roussel et al. 1999) particularly at colder temperatures (Heggenes and Saltveit 1990, Heggenes 1996) utilizing coarse substrates during the day and fine substrates at night (Elso and Greenberg 2001, Saltveit et al. 2001).

6.2.1.3 Juvenile substrate (fish length: >7 to 20 cm)

Studies concerning juvenile brown trout substrate usage were largely based in Europe, the bulk of these in northern Europe (Heggenes 1988c, Heggenes and Saltveit 1990, Greenberg et al. 1996, Mäki-Petäys et al. 1997, Bremset 2000, Heggenes and Dokk 2001, Heggenes et al. 2002). This summary also includes studies from central and southern Europe (Rincon and Lobon-Cervia 1993, Roussel et al. 1999, Vismara et al. 2001), and the United States (Raleigh et al. 1986, Kocik and Taylor 1996). The majority of these investigations involved sympatic populations of brown trout with the co-occurring salmonid species include Arctic charr (*Salvelinus fontinalis*) (Heggenes and Dokk 2001), Atlantic salmon (*Salmo salar*) (Heggenes and Saltveit 1990, Bremset 2000, Heggenes and Dokk 2001, Heggenes et al. 2002), brook trout (*Salvelinus fontinalis*) (Kocik and Taylor 1996), grayling (*Thymallus thymallus*) (Greenberg et al. 1996, Mäki-Petäys et al. 1997, Vismara et al. 2001), rainbow trout (*Salmo gairdneri*) (Raleigh et al. 1986, Kocik and Taylor 1996), and cutthroat trout (*Salmo clarki*). There were allopatric populations examined as well (Heggenes 1988c, Rincon and Lobon-Cervia 1993, Roussel et al. 1999). All of the habitat surveys employed underwater survey techniques with the exception of the studies by Heggenes (1988c), Mäki-Petäys et al. (1997) and Vismara et al. (2001). All investigation assessed substrate type using visual estimates and modified Wentworth classification systems. The literature used in this synopsis is summarized in Table 6.3 and the construction of the nursery tolerance profile for substrate is illustrated in Figure 6.4.

All the substrate classes present in rivers (clay, silt to bedrock) have been reported being used by juvenile S. *trutta*. Individual studies report broad usage of the substrate with individual investigators reporting full spectrum usage (Greenberg et al. 1996, Heggenes et al. 2002). The preferred habitat used by juvenile trout is broad as well. Raleigh et al. (1986), Kocik and Taylor (Kocik and Taylor 1996), and Roussel et al. (1999) report a preference for fine substrate (clay, silt) while Heggenes (1988c), Greenberg et al. (1996) and Vismara et al. (2001) suggest a preference for large boulders. As well, Greenberg et al. (1996) working at dusk have calculated a preference range ranging from boulders to bedrock; although

Table 6.3: Literature used to define the range of substrate sizes that can be used by juvenile brown trout (*Salmo trutta*) (>7 – 20cm).

Source	Number of fish sampled	Preferred Substrate (FH/SC) † *cm*	Category range (FH/SC) *cm*	Category size (FH/SC) *cm*	Fish size: mean ±SD (range) *cm*	Study Location	Allo - sympatric Populations	Other species	Survey method
Bremset 2000 R. Todalselva Aug-Sept Nov R. Vindøla Aug-Nov	 135 9 115	 none 0.2-1.0 >26.4	 <0.2 to >26.4 <0.2 to >26.4 <0.2 to >26.4	Modified Wentworth after Jowett et al 1991	Parr (89%) YOY (11%)	Norway (Central)	Sympatric	Atlantic Salmon	Underwater
Greenberg et al. 1996 Diurnal Dive Dusk Dive Stone Dive	 55 83 32	 >38.4 >38.4 to bedrock 6.4 to >38.4	 1.6 to bedrock 0.0004 to bedrock 6.4 to >38.4	Modified Wentworth after Heggenes & Saltveit 1990	(7-11)	Sweden (Norrland)	Sympatric	Grayling	Underwater
Heggenes 1988	130	6.4 to > 51.2	0.8 to smooth bedrock	Modified Wentworth after Bain et al. 1985	17.0 (12.1-27.5)	Norway (southeast)	Allopatric	-	Electro-fishing
Heggenes & Dokk 2001 Summer Winter	 120	 6.4 – 25.6 6.4 – 25.6	 0.007 to > 51.2 0.007 to smooth bedrock	Modified Wentworth	13±10.6 (5-16)	Norway (southwest)	Sympatric	Atlantic Salmon Arctic Charr	Underwater
Heggenes & Saltveit 1990	306	25.6 – 51.2	0.0004 to > 51.2	Modified Wentworth after Bain et al. 1985	9 to 13 (range of mean: sections 1 to 5)	Norway (western)	Sympatric	Atlantic Salmon	Underwater
Heggenes et al. 2002	769	3.2 –25.6	0.0004 to smooth bedrock (>51.2)	Modified Wentworth	≥7	England (southwest)	Sympatric	Atlantic Salmon	Underwater
Kocik & Taylor 1996	154	0.0004 to 0.19	2.4×10^{-5} to 12.8	Modified Wentworth after Bovee 1986	8.8 (Oct) 9.3 (Feb)	USA (Michigan)	Sympatric	Rainbow trout	Underwater
Mäki-Petäys et al. 1997 Midsummer Late summer Autumn Winter	 22 27 17 24	(median) 12.8-25.6 12.8-25.6 12.8-25.6 12.8-25.6	(range) 3.2 to >25.6 3.2 to >25.6 3.2 to >25.6 6.4 to >25.6	Wentworth modified from Heggenes 1988	10-15	Finland (north-eastern)	Sympatric	Grayling	Electro-fishing

Table 6.3: Continued.

Source	Number of fish sampled	Preferred Substrate (FH/SC)‡ *cm*	Category range (FH/SC) *cm*	Category size (FH/SC) *cm*	Fish size; mean ±SD (range) *cm*	Study Location	Allo - sympatric Populations	Other species	Survey method
Raleigh et al. 1986									
Moyle et al. 1983	194	0.2-25.0	0.0062 to bedrock	After Bovee 1982	5.1-11.9	USA (Utah)	Sympatric	Rainbow, cutthroat trout	Underwater
Gosse et al. 1977	239	<0.0062-25	<0.0062 to bedrock		15.0-23.1				
Gosse 1981	327	0.2-25.0	<0.0062 to bedrock		-				
Rincón and Lobón-Cerviá 1993									
July 1986	54	0.2-2.5	Silt to boulders (>30cm)$^{\alpha}$	After Grossman & Freeman 1987	12.6	Spain (Valdés, Asturias)	Allopatric	-	Underwater
October 1986	58	0.2-2.5			13.1				
January 1987	39	0.2-2.5			14.1				
May 1987	42	0.2-2.5			13.4				
Roussel et al. 1999						France (Brittany)	Allopatric	-	Underwater
Day	43	1.6-6.4	0.006 to 50.0	Wentworth scale	10-20				
Night	52	<0.006 (silt)	< 0.006 to 50.						
Vismara et al. 2001	315	0.2-6.2 & 25-400	0.2 to 400	PHABSIM substrate code	(12-22)	Spain (Valtellina Valley)	Sympatric	Grayling	Electro-fishing

† Frequency Histogram, Suitability Curve; α – maximum particle diameter;

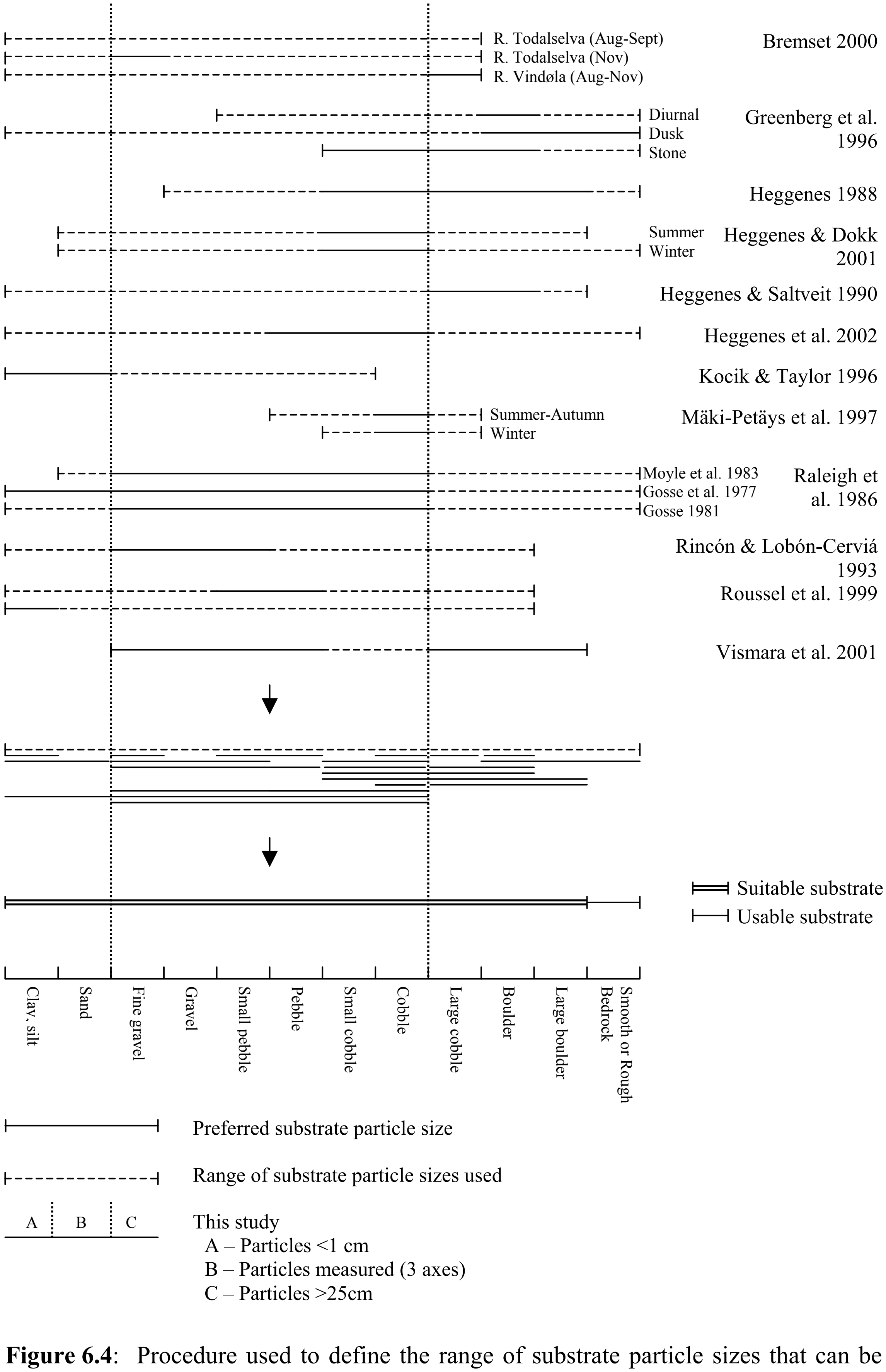


Figure 6.4: Procedure used to define the range of substrate particle sizes that can be used by juvenile brown trout (*Salmo trutta*) (>7 – 20cm).

they describe this preference as being weak. The diversity of streambed substrate usage is reflected in the tolerance profile. The suitable range, based on the preference ranges observed, includes all substrates from clay, silt (0.0004-0.006 cm) to large boulders (38.41-51.2 cm). Because the preference for bedrock was considered weak by Greenberg and his colleagues (1996) it was exclude from the suitable portion of the profile.; however, bedrock is considered useable by juvenile trout.

6.2.1.4 Substrate used by non-spawning adults (> 20 cm)

There are only six studies that report substrate microhabitat used by adult non-spawning *Salmo trutta*. The studies were conducted in Scandinavia (Heggenes 1988c, Greenberg et al. 1996, Mäki-Petäys et al. 1997), the United States (Raleigh et al. 1986, Strakosh et al. 2003), and Spain (Vismara et al. 2001). With the exception of the investigations conducted by Strakosh et al. (2003) and Heggenes (1988) involved sympatric populations of brown trout with the co-occurring species including grayling (*Thymallus thymallus*) (Greenberg et al. 1996, Mäki-Petäys et al. 1997, Vismara et al. 2001), rainbow (*Salmo gairdneri*) and cutthroat trout (*Salmo clarki*) (Raleigh et al. 1986). Half of the studies were conducted using underwater surveys (Raleigh et al. 1986, Greenberg et al. 1996, Strakosh et al. 2003) and the remaining were surveys used electro-fishing methodologies (Heggenes 1988c, Mäki-Petäys et al. 1997, Vismara et al. 2001). All investigation assessed substrate type using visual estimates and modified Wentworth classification systems. The literature used in this synopsis is summarized in Table 6.4 and the construction of the nursery tolerance profile for substrate is illustrated in Figure 6.5.

The full range of streambed substrate usage has been observed by the authors in this

Table 6.4: Literature used to define the range of substrate sizes that can be used by adult brown trout (*Salmo trutta*) (>20cm).

Source	Number of fish sampled	Preferred Substrate (FH/SC) † *cm*	Category range (FH/SC) *cm*	Category size (FH/SC) *cm*	Fish size; mean ±SD (range) *cm*	Study Location	Allo - sympatric Populations	Other species	Survey method
Greenberg et al. 1996 Diurnal Dive Dusk Dive Stone Dive	 111 55 -	 0.2-1.6 >38.4 to bedrock -	 0.0004 to bedrock 0.0004 to bedrock -	Modified Wentworth after Heggenes & Saltveit 1990	(12-35)	Sweden (Norrland)	Sympatric	Grayling	Underwater
Heggenes 1988	11	25.6 to >51.2	0.8 to 51.2	Modified Wentworth after Bain et al. 1985	24.7 (0.23 SE) n=19 [11.2-43.3]	Norway (southeast)	Allopatric	-	Electro-fishing
Mäki-Petäys et al. 1997 Midsummer Late summer Autumn Winter	 20 12 16 10	(median) 6.4-12.8 12.8-25.6 12.8-25.6 12.8-25.6	(range) 3.2 to >25.6 6.4 to >25.6 6.4 to >25.6 12.8 to >25.6	Wentworth modified from Heggenes 1988	16-25	Finland (north-eastern)	Sympatric	Grayling	Electro-fishing
Raleigh et al. 1986 Gosse et al. 1977 Gosse 1981 (swim) Gosse 1981 (rest)	 352 225 222	 <0.0062-0.2; 6.4-25.0 0.2-25.0 <0.0062-0.2	 <0.0062 to bedrock <0.0062 to bedrock <0.0062 to bedrock	After Bovee 1982	> 23.9	USA (Utah)	Sympatric	Rainbow, cutthroat trout	Underwater
Stakosh et al. 2003	144	> 6.4 - 25.6	<0.2 – irregular bedrock	Modified Wentworth after Bain et al 1985	≥ 17	USA (Conn)	Allopatric	-	Underwater
Vismara et al. 2001	213	0.2-6.2 & 25-400	0.2 to 400	PHABSIM substrate code	(> 22)	Spain (Valtellina Valley)	Sympatric	Grayling	Electro-fishing

† Frequency Histogram, Suitability Curve;

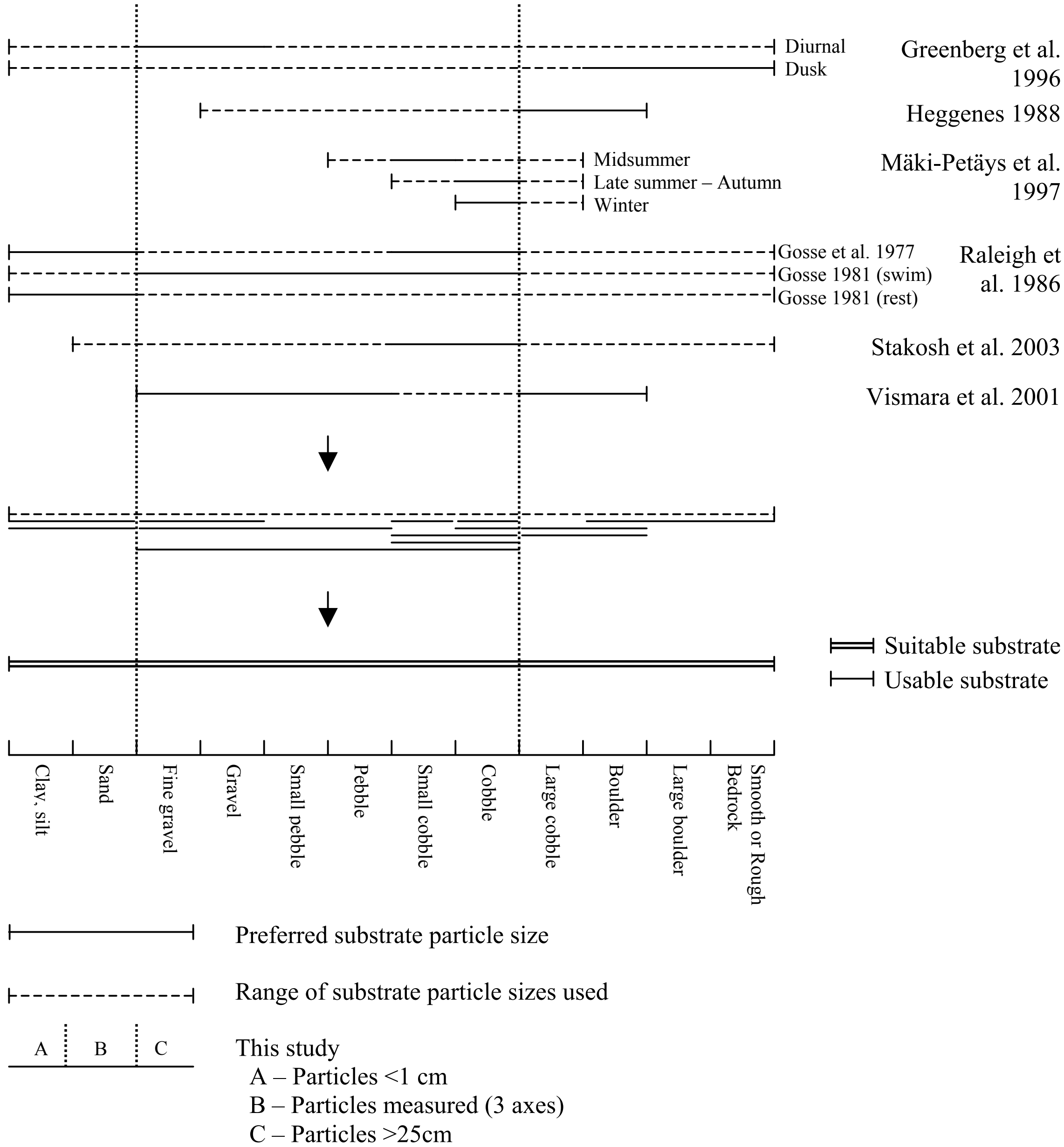


Figure 6.5: Procedure used to define the range of substrate particle sizes that can be used by brown trout (*Salmo trutta*) Adults (>20cm).

summary. As well there are expressed preferences that cover the full range substrate habitat available. This includes the preference by trout for boulder and bedrock at dusk in a Greenberg et al. (1996) survey in a Swedish river. In the juvenile summary this preference was dismissed as weak and excluded from the suitable substrate range. However, in this instance it will be included. There are fewer studies that have recorded adult habitat usage than any other age class and more data may reveal a stronger link between adult stream position and the largest substrate. There has been a link noted between fish size and cobble size (i.e. the larger fish are associated with larger cobble) (Witzel and MacCrimmon 1983, Greenberg et al. 1996, Heggenes et al. 2002), as trout use the crevices as cover from high velocities and predator avoidance (Raleigh et al. 1986, Saltveit et al. 2001, Pender and Kwak 2002). The larger fish will require larger spaces provided by the largest streambed particles. Cover in this form may be provided by fissures and overhangs provided in rough bedrock substrates so this classification will be included in the suitable range for adult non-spawning brown trout. All substrate classes available are included in this suitable range indicating that adult residents do not seem to have a strong preference for streambed substrate.

6.2.1.5 Summary of tolerance profiles

Brown trout have a broad tolerance for streambed substrates (Figure 6.6). During all stages of life, with the exception of spawning, all age classes can use the full range of inorganic substrates available. Spawning trout have the strictest substrate requirements demanding particles sizes ranging from fine gravel to small cobble substrates. The suitable range of substrates increases with age with fry favouring sand to cobble

Summary of Tolerance Profiles

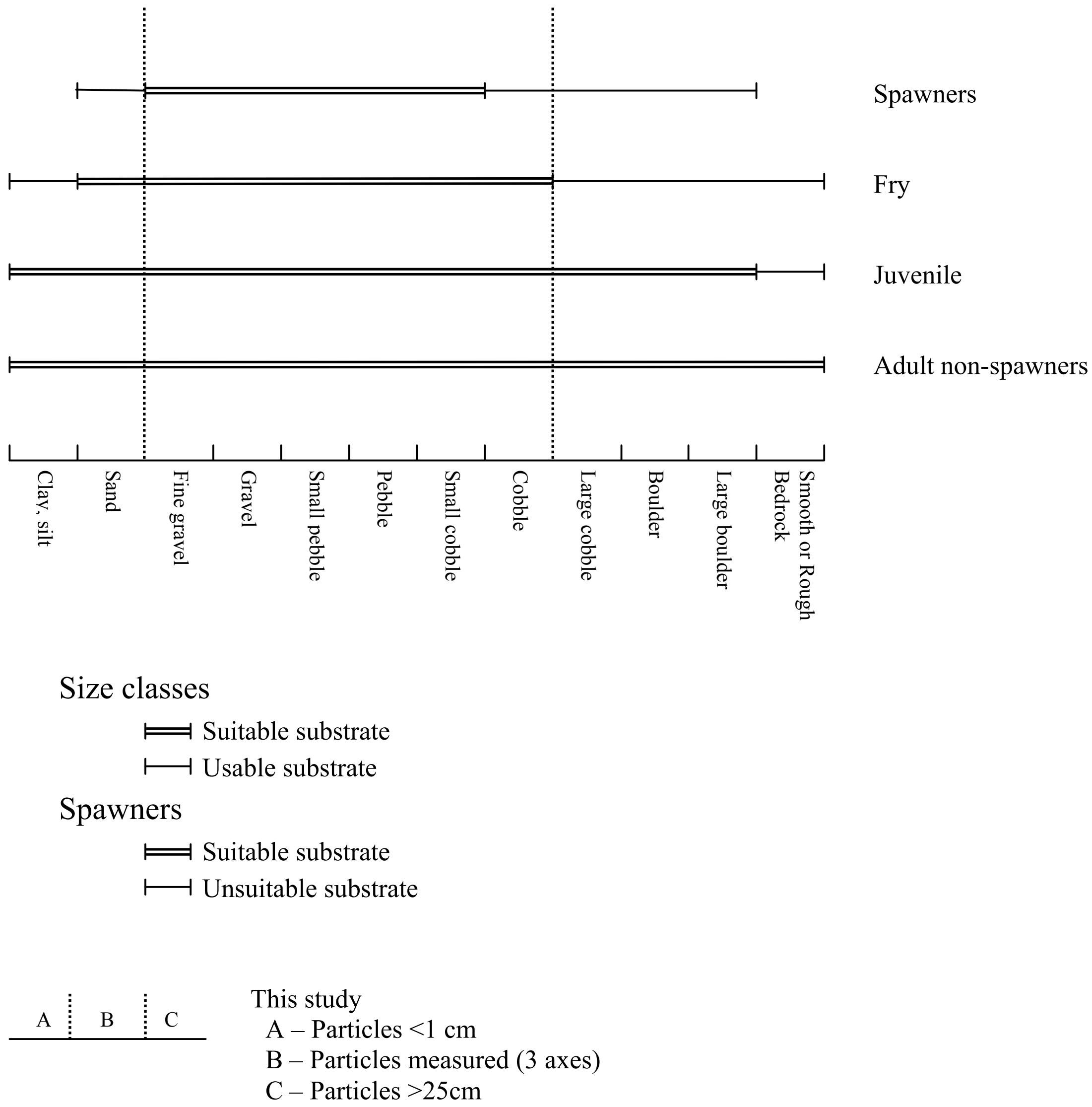


Figure 6.6: Summary of tolerance profiles by age class [including spawning; nursery/fry (≤7 cm); juvenile (>7 –20 cm); and adult non-spawners (> 20 cm)] for measures of substrate particle sizes used by brown trout (*Salmo trutta*).

substrates, juveniles choosing clay/silt to large boulders, and adult trout found across the entire distribution indication they have no strong presence for streambed particle size.

The broad range of streambed substrate particles used by resident life stages of *S. trutta* may be due to a number of factors. As previously mentioned, substrate is used as cover both from predation and as velocity refugia. There are often non-substrate forms of cover available for fish such as undercut banks, macrophytes, deep pools, or overhanging vegetation at stream edges (Raleigh et al. 1986, Griffith and Smith 1993, Cunjak 1996). These features can be used as if appropriate substrate is not available, thus, they may be associated with substrates such as sand or bedrock if alternate forms of cover can be found at those locations. As well, we have already noted diurnal shifts in trout stream position (nursery substrate: Section 6.2.1.2). Additionally, trout change their stream location with season (Cunjak 1996, Mäki-Petäys et al. 1997). By pooling this data, as I have done in this summary, the full range of habitat usage is presented. The summer-winter or day-night examined separately would probably yield narrower tolerances.

Streambed substrate is one of the four most important criteria used by non-spawning brown trout for habitat selection but it has been thought of as one of the least important of these variables. For example, Greenberg et al. (1996) note a weak preference for substrate. Other factors (velocity, depth) may take precedence over substrate when choosing stream position resulting in more flexible or non-specific particle size associations. As well, trout often use large upstream boulders as velocity refuges (Bachman 1984). The substrate that they are immediately associated with (focal point substrate) may not reflect their true selection criteria.

6.2.2 Habitat assessment

The profiles constructed in Section 6.2.1 when applied to the data gathered from each quadrat measured in the reaches studied. The proportion of available streambed substrate for each study reach at each life stage is summarized in the following sections.

6.2.2.1 Spawning substrate

Stream substrate could either be used (suitable) or not used (not-suitable) for spawning: there was no intermediate or marginal 'useable' category. The proportion of suitable and not-suitable substrate that was available at each site (seven datasets) was compared using a chi-squared test. The results of this test and comparisons of pre- and post-spate at Bin Burn are presented in Table 6.5 and 6.6. The comparison of the substrate suitability for spawning between the seven data sets revealed that statistically there are differences between these data sets (i.e. streambed substrate is dependent on site: X^2 = 337.0, df = 6, p = 0.000). The proportion of water velocity suitability classifications for each site is outlined in Table 6.7. The streambed substrate suitable for spawning ranged from 11.4% to 81.6% (March Burn-downstream and March Burn-upstream, respectively). Not surprisingly, the amount of streambed substrate that was considered not-useable by spawning trout ranged from 18.4% to 88.6% observed at March Burn-upstream and downstream, respectively. Histograms that illustrate the proportion of suitable and not-useable spawning substrates at all sites for all seven data sets are presented in Figure 6.7 (A-G).

The total area available with acceptable substrate for spawning was calculated by multiplying the proportion of suitable substrate by the total surface area available in each reach. Again, only the total streambed was analysed for substrate, not total and

Table 6.5: Results for chi-square test comparing the proportions of quadrats that contained substrate that was considered suitable, useable, and not-useable, (suitable and unsuitable only for spawning) by brown trout at all study reaches and life stages.

Life Stage	Streambed	X^2 value	DF	Cells < 5†	p-value	X^2 (Likelihood)*
Spawn	Total	337.0	6	0	0.000	no
Nursery	Total	585.6	6	0	0.000	no
Juvenile	Total	585.6	6	0	0.000	no
Adult	Total	-	-	-	-	-

† Number of cells with expected values less than 5; * Used likelihood ratio chi-square

Table 6.6: Results for chi-square test comparing the proportions of quadrats that contained substrate that was considered suitable, useable, and not-useable (suitable and unsuitable only for spawning) by brown trout at Bin Burn before and after the spate at all life stages.

Life Stage	Streambed	X^2 value	DF	Cells < 5†	p-value	X^2 (Likelihood)*
Spawn	Total	55.7	1	0	0.000	no
Nursery	Total	220.4	1	0	0.000	no
Juvenile	Total	220.4	1	0	0.000	no
Adult	Total	-	-	-	-	-

† Number of cells with expected values less than 5; * Used likelihood ratio chi-square

wetted as in the previous chapters, as measures of substrate can be taken in the absence of water. The total area available with acceptable substrate for the four life stages and seven data sets is presented in Table 6.8. The smallest available area for spawning was found at Burnhouse Burn-upstream (5.5 m^2), the next smallest at March Burn-downstream (9.4 m^2) and the third smallest at Bin Burn-upstream (12.3 m^2). The largest streambed area with appropriate spawning substrate was observed at Bin Burn-downstream after the spate (139.9 m^2). March burn-upstream had the second largest area (48.1m^2), which was quite a bit smaller than Bin-Burn-downstream. The

Table 6.7: Summary of the proportion of stream substrate available for the four life stages of brown trout for all study reaches for the total streambed (including wetted and dry portions).

		Spawning Total (%)	Nursery Total (%)	Juvenile Total (%)	Adult Total (%)
March Burn Upstream	Suitable Useable Not-useable	81.6 - 18.4	100.0 0.0 0.0	100.0 0.0 0.0	100.0 0.0 0.0
March Burn Downstream	Suitable Useable Not-useable	11.4 - 88.6	19.1 80.9 0.0	19.1 80.9 0.0	100.0 0.0 0.0
Burnhouse Burn Upstream	Suitable Useable Not-useable	45.6 - 56.4	48.5 51.5 0.0	48.5 51.5 0.0	100.0 0.0 0.0
Burnhouse Burn Downstream	Suitable Useable Not-useable	48.8 - 51.3	54.6 45.4 0.0	54.6 45.4 0.0	100.0 0.0 0.0
Bin Burn Upstream	Suitable Useable Not-useable	28.5 - 71.5	39.4 60.6 0.0	39.4 60.6 0.0	100.0 0.0 0.0
Bin Burn Downstream (Pre-spate)	Suitable Useable Not-useable	39.3 - 60.1	42.3 57.7 0.0	42.3 57.7 0.0	100.0 0.0 0.0
Bin Burn Downstream (Post-spate)	Suitable Useable Not-useable	64.7 - 35.3	84.6 15.4 0.0	84.6 15.4 0.0	100.0 0.0 0.0

remaining sites had intermediate values. A great deal more substrate was available after the spate at Bin Burn than before. The surface area increased from 35.0 m^2 to 139.0 m^2. The test that compares the relative proportion of suitable and not-suitable between the substrate available for spawning before and after the spate indicates that, statistically, the proportions of habitat available are also different ($X^2 = 55.7$, df = 1, p = 0.000).

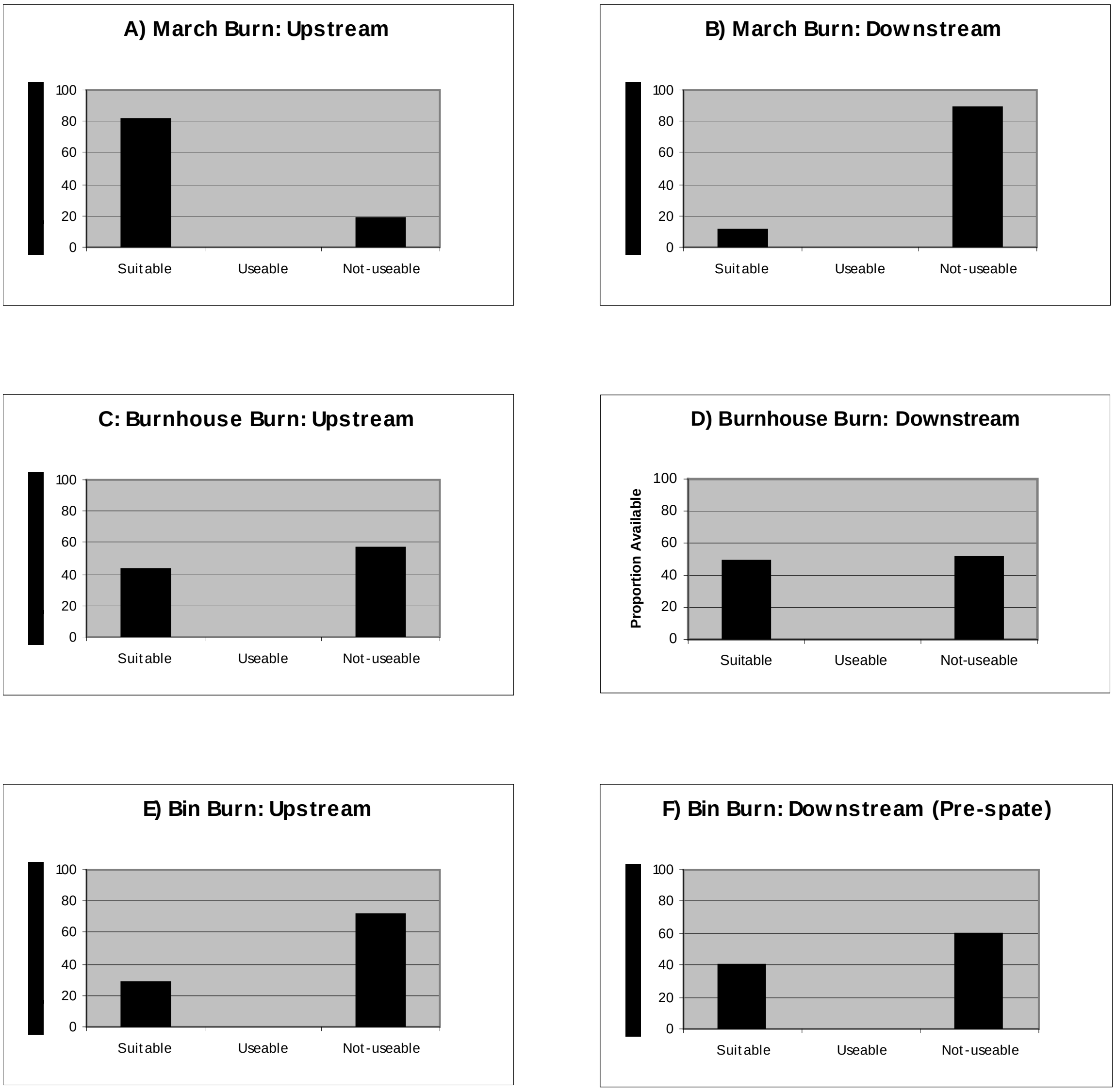


Figure 6.7 (A-G): Histogram of the proportion of streambed substrate available for spawning trout at all study sites classified as suitable and not-suitable for the total streambed (dry and wetted portions).

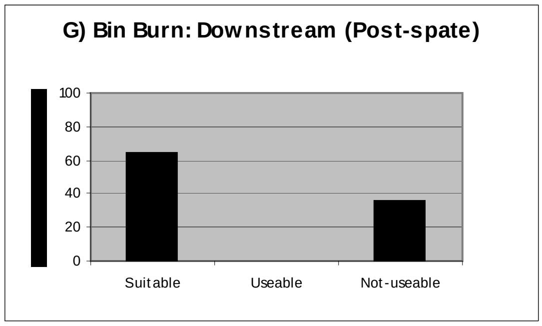


Table 6.8: The total area that has acceptable stream-substrate at each study reach for each age class. Total accessible area calculated by multiplying the total streambed area by the sum of the proportion of the streambed that was either suitable or useable.

Age Class	Site	Total Area (m^3)	Proportion of wetted area; suitable or useable (%)	Total area suitable or useable (m^2)
Spawning	March Burn-us†	58.9	81.6	48.1
	March Burn-ds‡	82.5	11.4	9.4
	Burnhouse Burn-us	12.3	43.6	5.5
	Burnhouse Burn-ds	30.1	48.8	14.7
	Bin Burn-us	43.0	28.5	12.3
	Bin Burn-ds (pre spate)	87.7	40.0	35.0
	Bin Burn-ds (post spate)	216.3	64.7	139.9
Nursery	March Burn-us	58.9	100.0	58.9
	March Burn-ds	82.5	100.0	82.5
	Burnhouse Burn-us	12.3	100.0	12.3
	Burnhouse Burn-ds	30.1	100.0	30.1
	Bin Burn-us	43.0	100.0	43.0
	Bin Burn-ds (pre spate)	87.7	100.0	87.7
	Bin Burn-ds (post spate)	216.3	100.0	216.3
Juvenile	March Burn-us	58.9	100.0	58.9
	March Burn-ds	82.5	100.0	82.5
	Burnhouse Burn-us	12.3	100.0	12.3
	Burnhouse Burn-ds	30.1	100.0	30.1
	Bin Burn-us	43.0	100.0	43.0
	Bin Burn-ds (pre spate)	87.7	100.0	87.7
	Bin Burn-ds (post spate)	216.3	100.0	216.3
Adult	March Burn-us	58.9	100.0	58.9
	March Burn-ds	82.5	100.0	82.5
	Burnhouse Burn-us	12.3	100.0	12.3
	Burnhouse Burn-ds	30.1	100.0	30.1
	Bin Burn-us	43.0	100.0	43.0
	Bin Burn-ds (pre spate)	87.7	100.0	87.7
	Bin Burn-ds (post spate)	216.3	100.0	216.3

† upstream; ‡ downstream

6.2.2.2 Nursery (fish length ≤ 7 cm) and juvenile substrate (fish length: >7 to 20cm)

The survey of habitat available for the remaining life-stages assessed habitat as suitable, useable, and not-suitable. Further, the tolerance profiles developed from the literature, when compared to the habitat available, are the same for the nursery and the juvenile life stages as the survey method used could not distinguish between the size categories that were larger in size than large cobble (or smaller than sand). The chi-squared test that examined the relative proportion of streambed substrate classified into these categories between the seven datasets for both the nursery and juvenile life stages indicates that the substrate suitability classifications are dependent on site ($X^2 = 585.6$, df = 6, p = 0.000). The relative portions of suitable, useable, and not-suitable substrate observed at the six sites is outlined in Table 6.7 and presented in Figure 6.8 (A-G) and 6.9 (A-G). None of the substrate available was considered not-useable within any of the seven data sets. Substrate observed to be useable for the young and intermediate life stages ranged from 0% (March Burn-upstream) to 80.9% at March Burn-downstream. Again, like the spawning data, these trends are reversed for suitable substrate with the greatest amount at March Burn-upstream (100.0%) and the least amount available at March Burn-downstream (19.1%).

As there is no streambed substrate that is not-useable, the entire area of the study reaches are acceptable for use by the smaller and intermediate size classes of trout. Therefore, the streambed classified as acceptable for these size classes of fish is essentially the size of the reach (Figure 6.8). Burnhouse Burn is the smallest reach and thus has the least amount of substrate available (12.3 m^2). The largest reach is the reconstructed section of Bin Burn that was created after the spate and thus has the greatest amount of streambed substrate available for small and intermediate sized trout. This site, post spate, is considerable larger than it was before the flood event. Prior to the disturbance only 87.7 m^2 was available which is similar to that found at March Burn-downstream (82.5 m^2). Further, the chi-squared test reveals that the relative proportion of suitable, useable and not-useable substrate is significantly different before

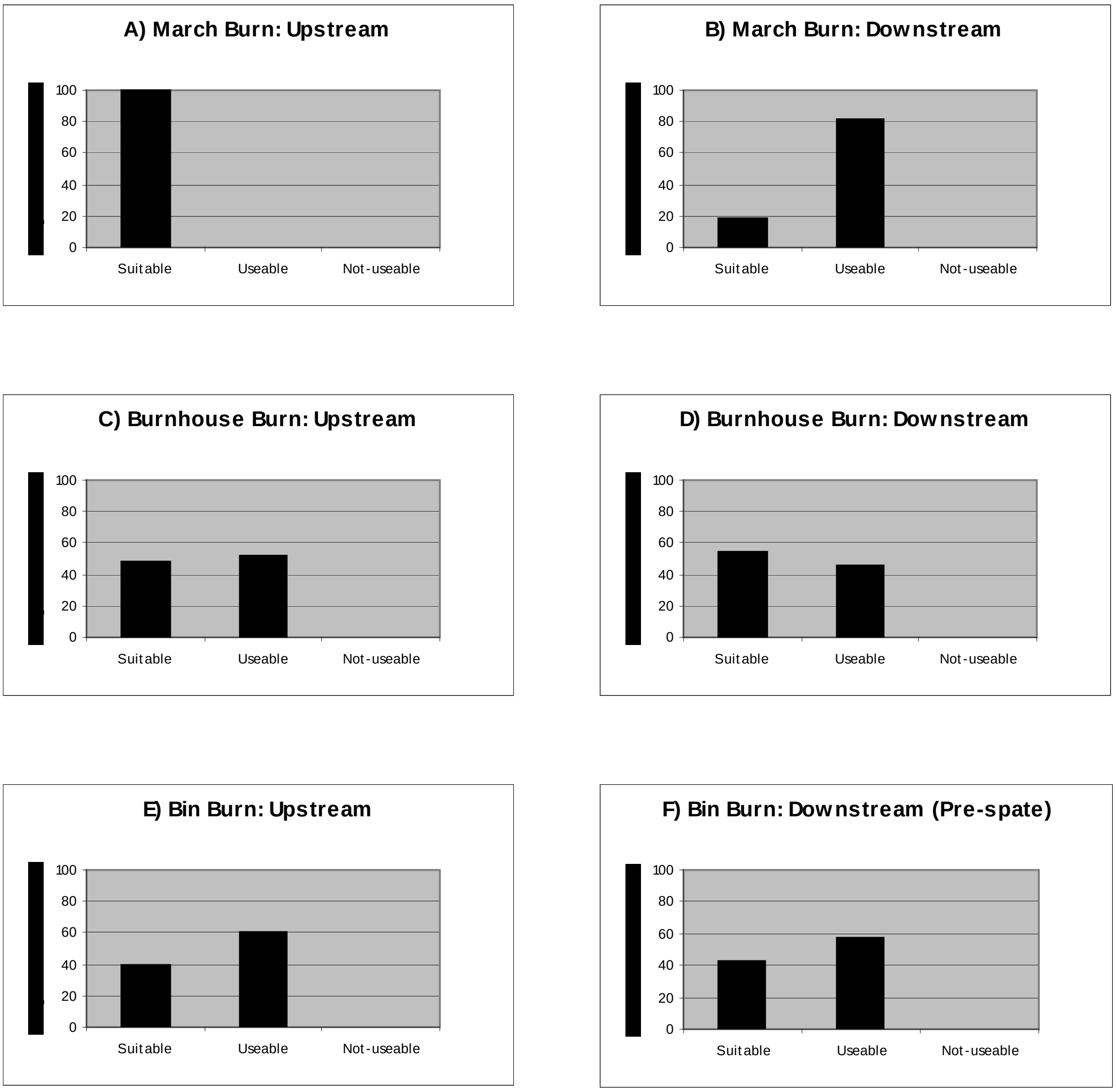
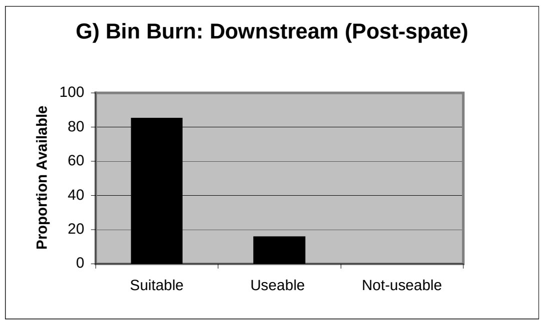
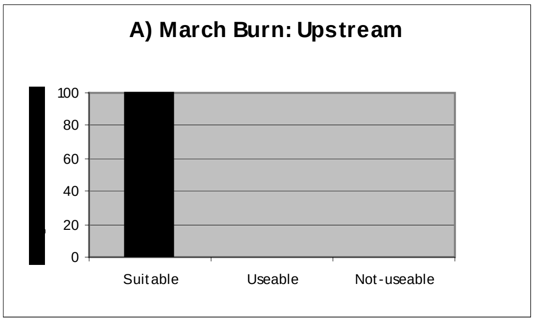
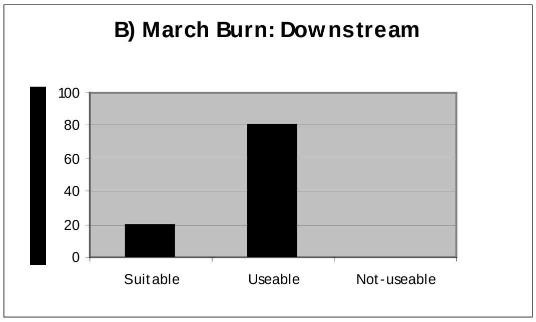
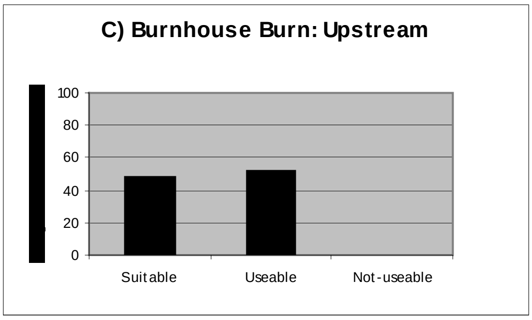


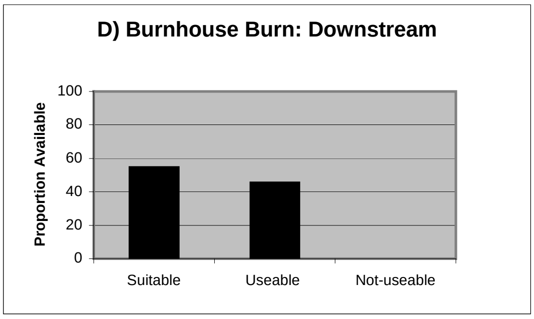
Figure 6.8 (A-G): Histogram of the proportion of streambed substrate available to young trout (≤ 7 cm) at all study sites classified as suitable, useable, and not-suitable for the total streambed (dry and wetted portions).

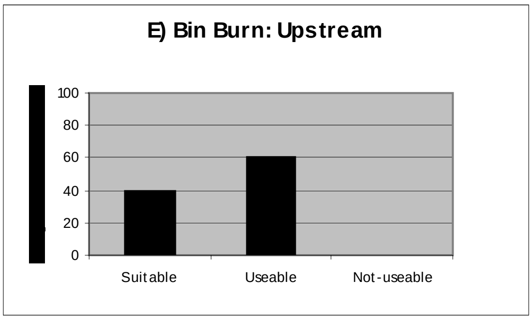












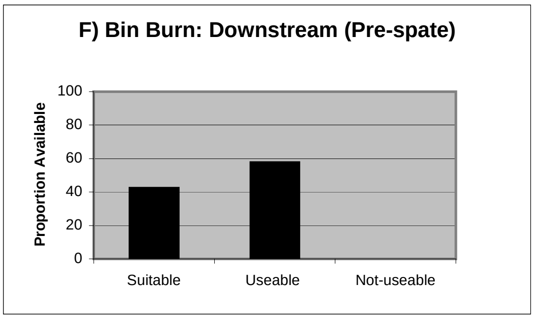
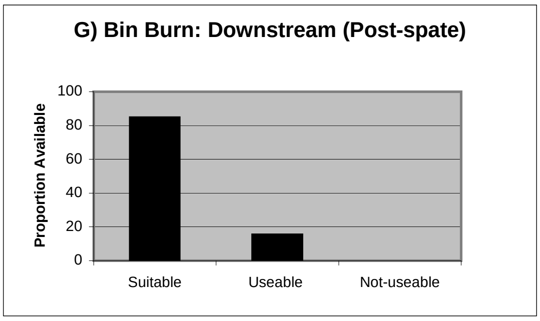


Figure 6.9 (A-G): Histogram of the proportion of streambed substrate available to juvenile trout (> 7 cm to 20 cm) at all study sites classified as suitable, useable, and not-suitable for the total streambed (dry and wetted portions).



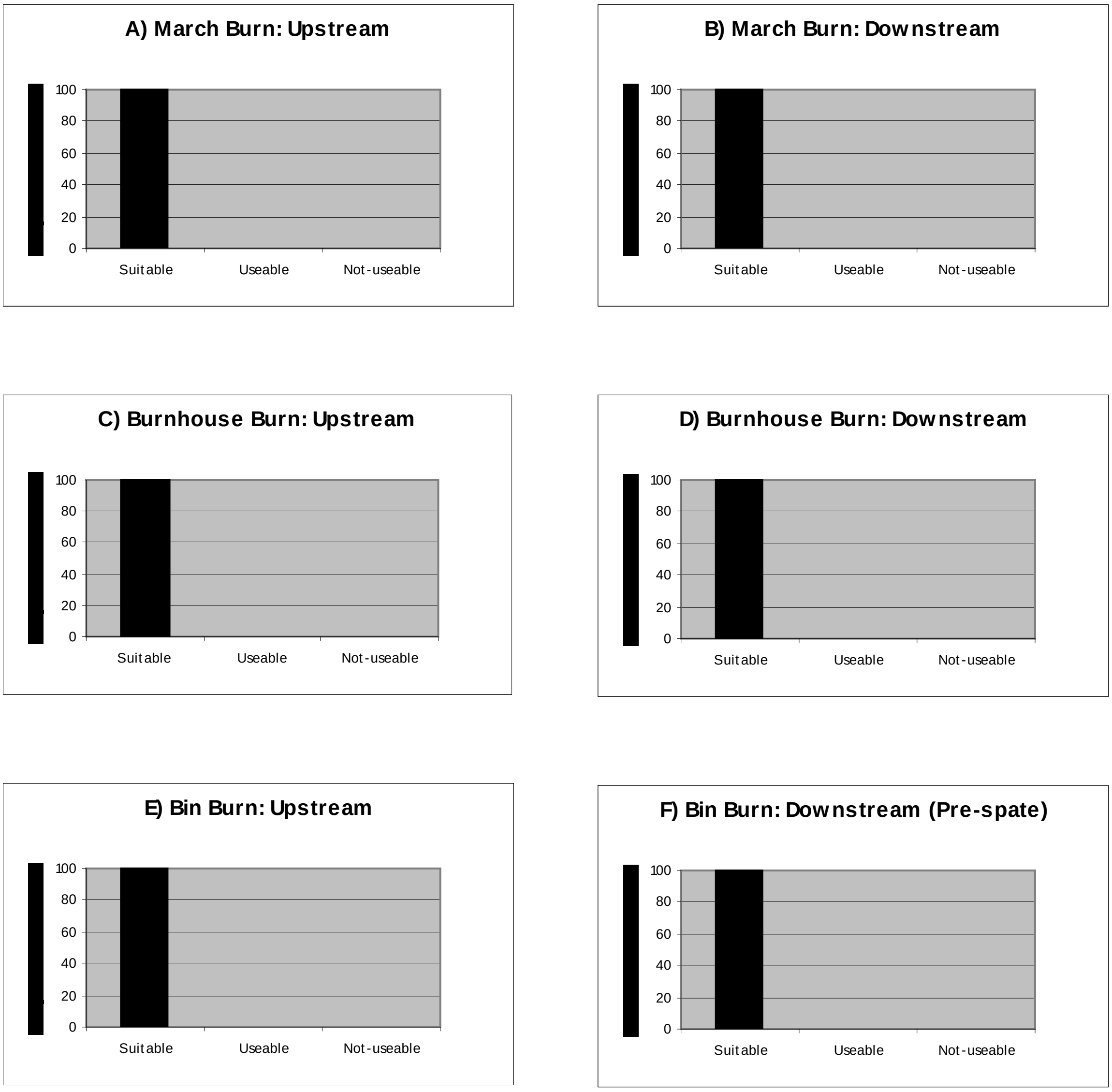
and after the spate (X^2 = 220.4, df = 1, p = 0.000). As would be expected, upstream sites have less area available for resident nursery and growing trout than the downstream sites.

6.2.2.3 Adult non-spawners substrate (> 20 cm)

It is clear from the construction of the tolerance profiles that adult resident brown trout do not discriminate substrate size when selecting microhabitat. All substrate size classes are considered suitable (i.e. no differences between sites) [Table 6.7 and Figure 6.10 (A-G)]. Thus, 100% of the substrate at all sites is considered suitable and like nursery and juvenile the total area available for use by this age-class is comprised of the entire area of the reach.

6.2.3 Habitat maps

A graphical representation of each stream and the habitat available for spawning (Figure 6.11), nursery (Figure 6.12), juvenile (Figure 6.13) and adult size classes (Figure 6.14), was produced. Figure 6.10 illustrates the location of streambed substrate that has been considered suitable or not-suitable for spawning by brown trout. The location of spawning habitat based on the analysis of streambed substrate is consistent with that seen for water depth and velocity. At the March Burn site the best available habitat is located just downstream of the pool at the upstream portion of the reach and is scattered more evenly throughout the streambed further downstream. In March Burn the best quality spawning substrate is distributed more evenly throughout the reach. As well, the graphic illustrates the observation that there was more suitably sized spawning substrate in Burnhouse Burn-downstream than March Burn-downstream even though the March



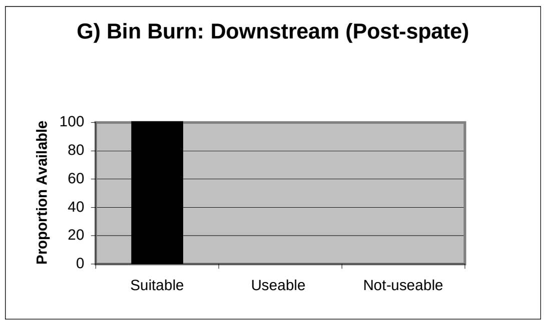


Figure 6.10 (A-G): Histogram of the proportion of streambed substrate available to adult trout (> 20 cm) at all study sites classified as suitable, useable, and not-suitable for the total streambed (dry and wetted portions).

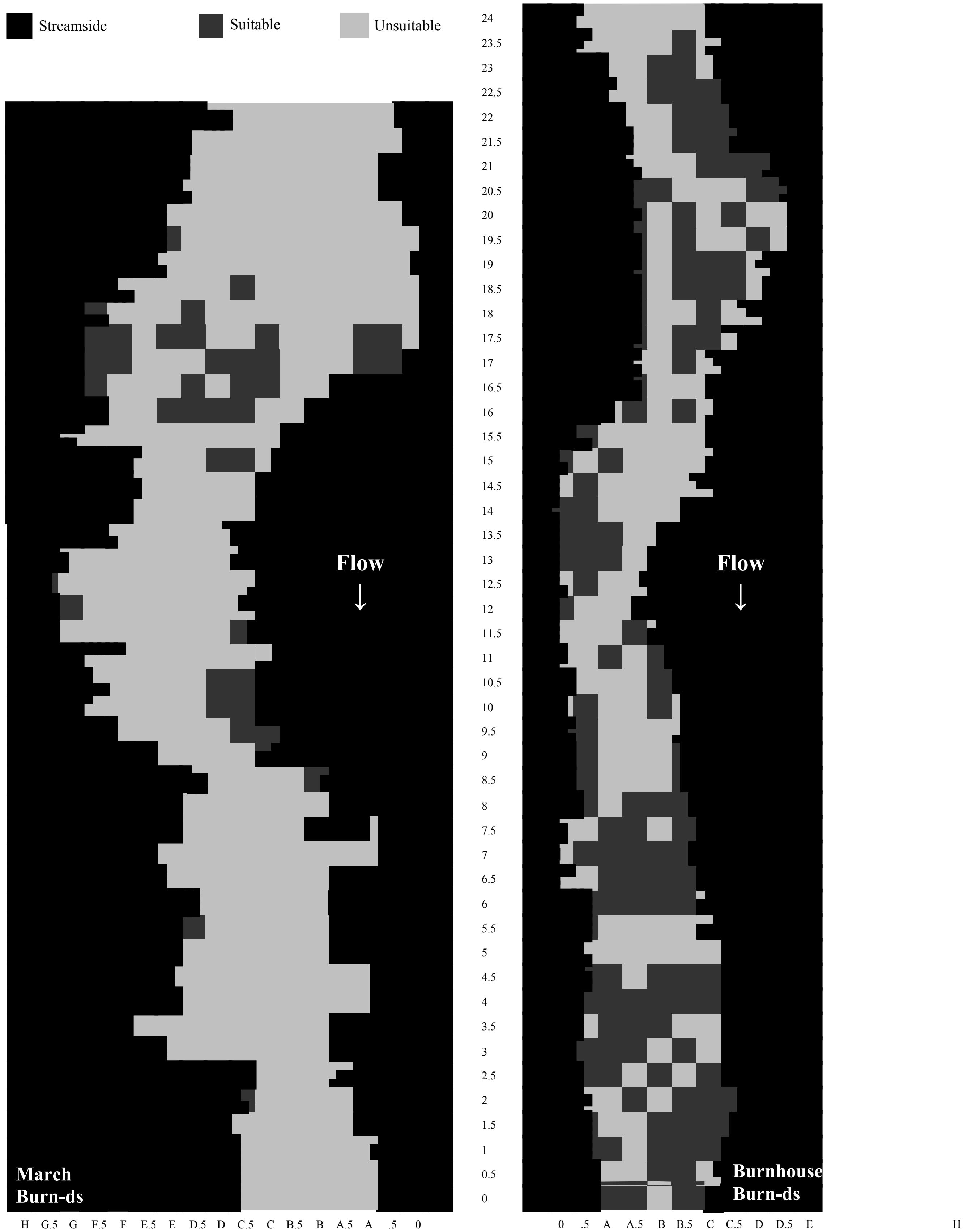


Figure 6.11: Habitat available for spawning brown trout (*Salmo trutta*) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on the substrate requirements.

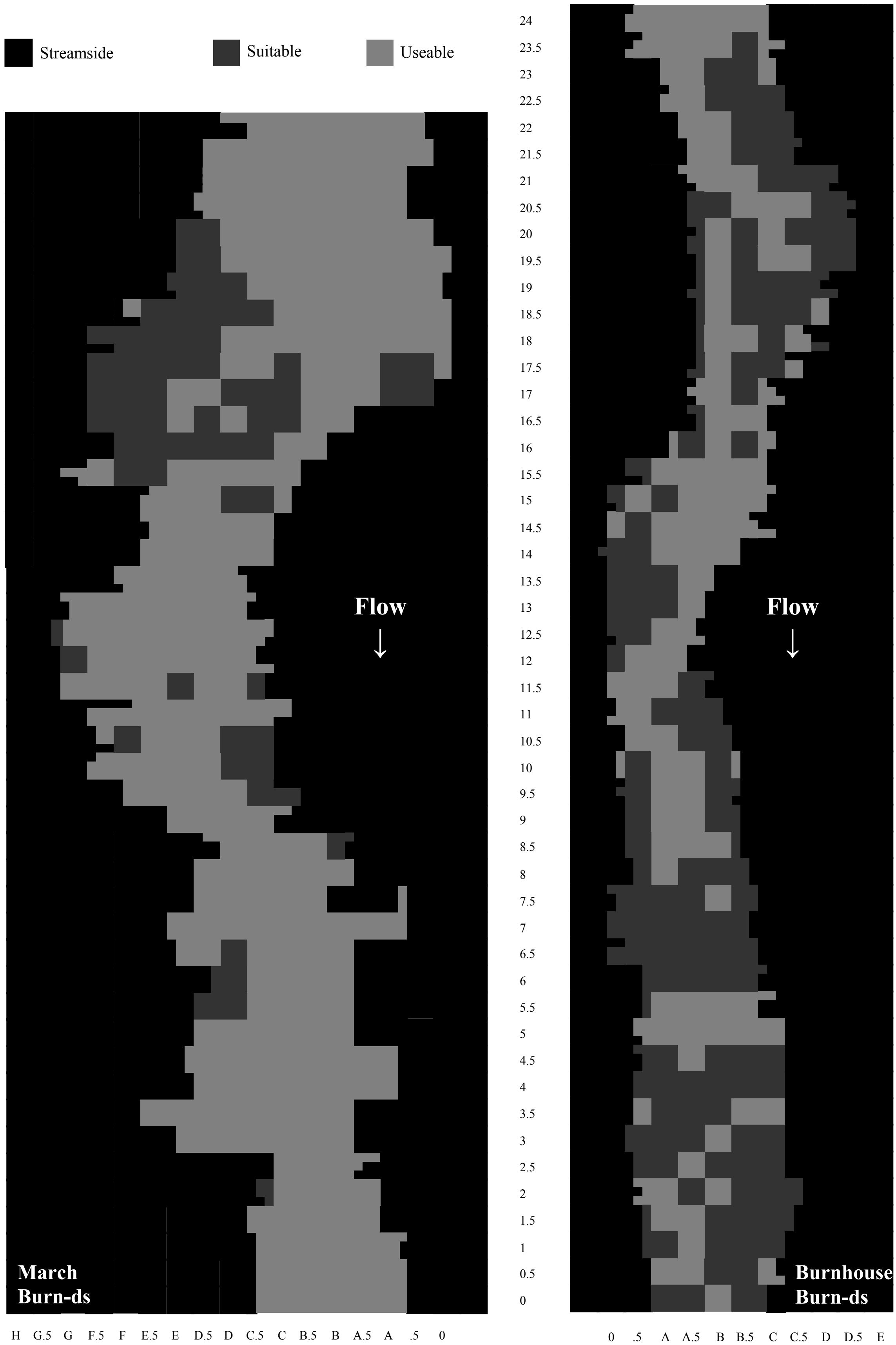


Figure 6.12: Habitat available for brown trout fry (*Salmo trutta*; length ≤ 7cm) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on the substrate requirements.

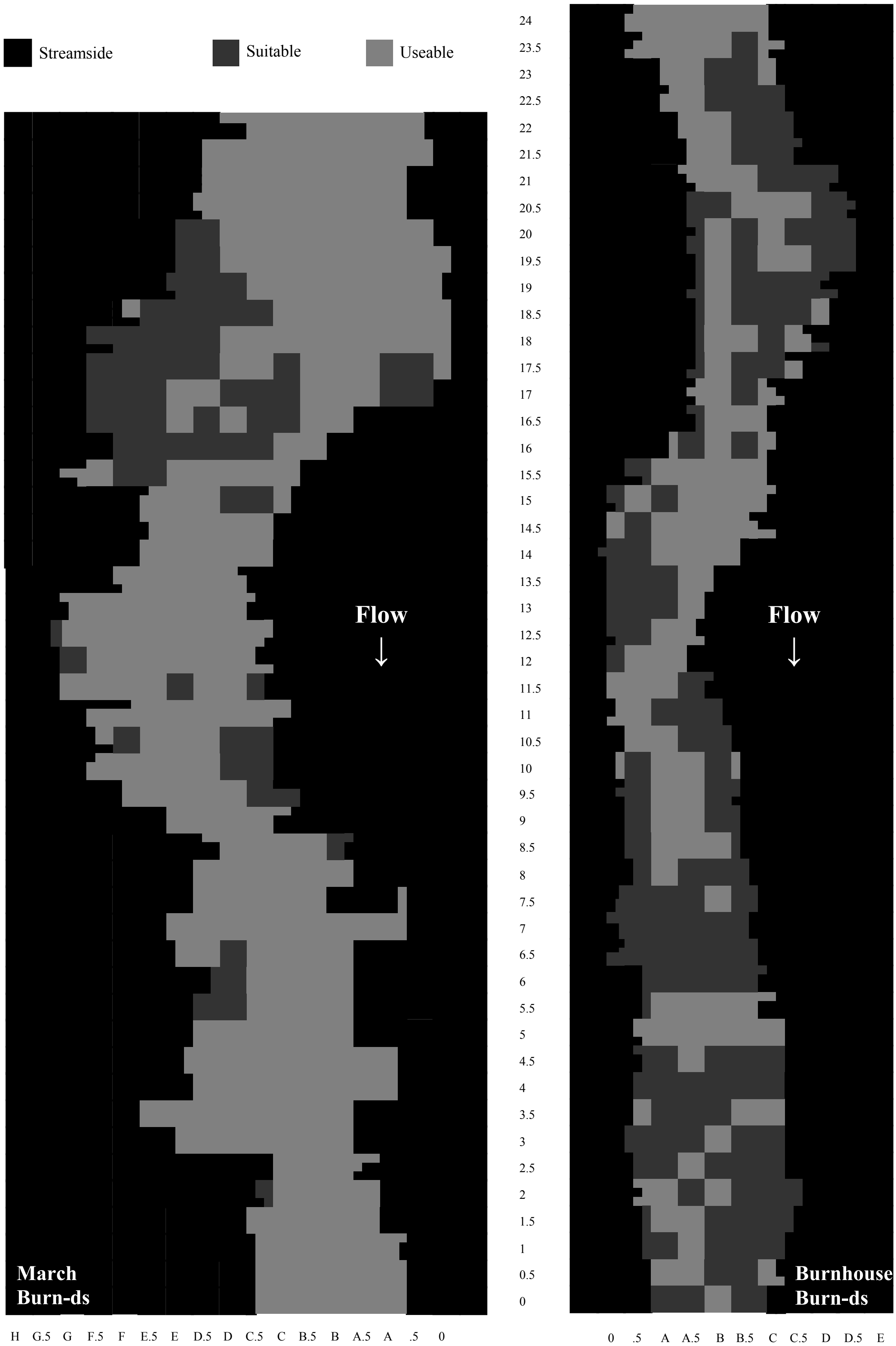


Figure 6.13: Habitat available for juvenile brown trout (*Salmo trutta*; length > 7 to 20 cm) in March ($Q = 0.341$ m^3/sec) and Burnhouse Burn ($Q = 0.0058$ m^3/sec) based on the substrate requirements.

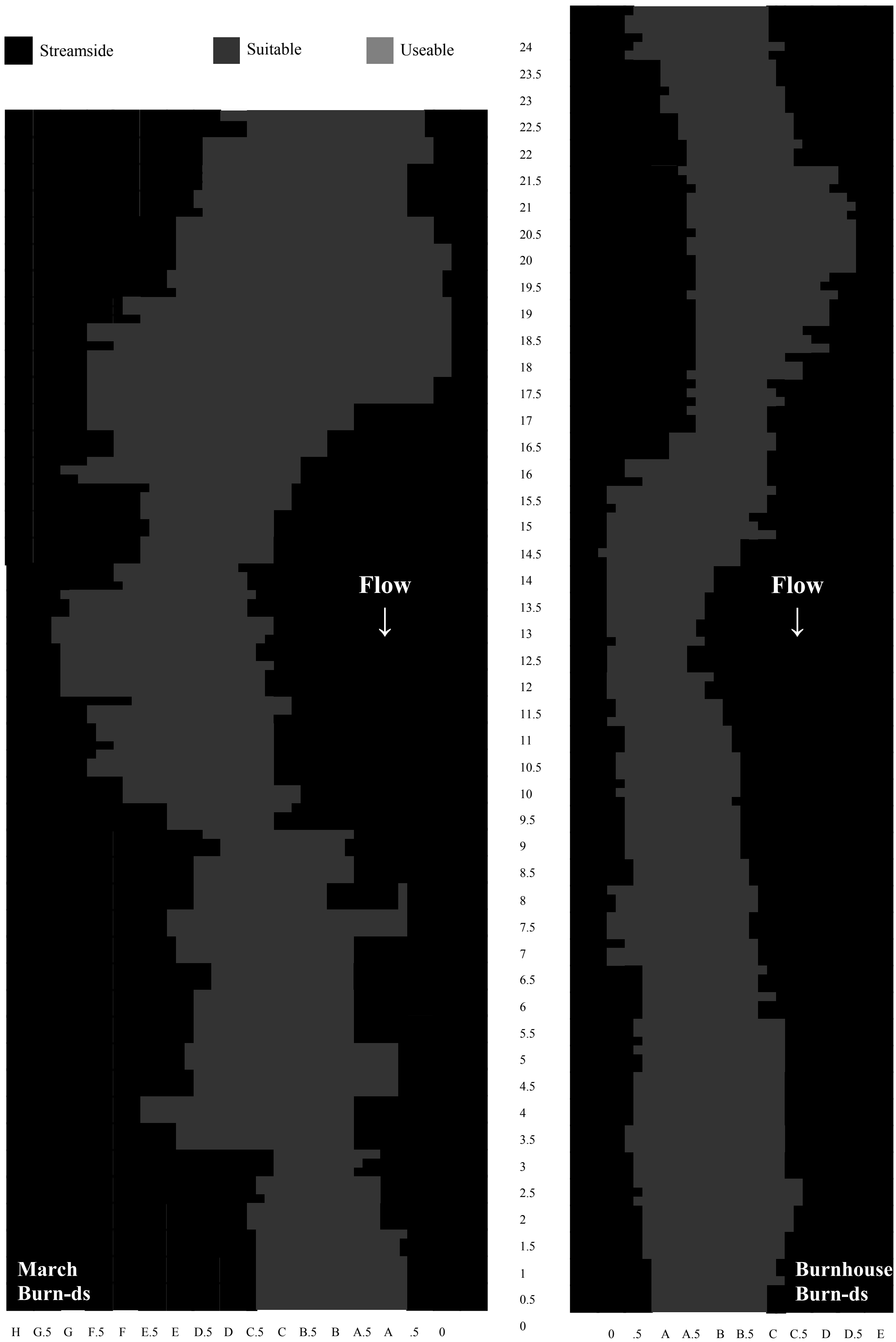


Figure 6.14: Habitat available for brown trout adults (*Salmo trutta*; length > 20 cm) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec) based on the substrate requirements.

Burn site has a larger surface area (Table 6.8). What the survey does not reveal is that the March Burn site was dominated by bedrock while the Burnhouse Burn site had a greater proportion of cobble and gravel (personal observation).

Figures 6.11 through 6.14 also clearly demonstrate how the proportion of substrate classified as suitable increases with age class and that microhabitat choice is not restricted in these streambeds for adult resident trout. Further, the proportions varied from site to site and between the two sites illustrated; however, all of the streambed substrate could be used (i.e. either useable or suitable) by resident trout at all age class.

6.3 Discussion

The creation of the tolerance profiles for the four different life stages of brown trout has allowed me to assess the availability of structural physical habit features provided by the existing streambed substrate in the study streams. This assessment was independent in that a direct comparison using physical reference sites was not necessary. As with stream depth and water velocity the criteria used in this assessment can be applied universally to any stream that currently or has supported populations of *S. trutta*. These habitat assessment criteria can also be used to assess the ability of fluvial environments to support trout that currently do not hold populations of these fish. All of the study reaches are well suited, from the perspective of streambed substrate, for use by all age classes of trout although proportionately there are differences between sites.

The analysis of substrate availability is similar to the assessment of water velocity in that it would appear that the streams are better suited to larger trout although most of the fish found, with the exception of autumn spawners in Burnhouse Burn, were fry and small juveniles (Chapter 4). As well, there was variable amounts substrate available for spawning at all sites. Like the assessment of water velocity usage, this is a reflection of the ability of older trout to use a broader range of physical habitat rather than a result of more habitat available to the older age classes (see Figure 6.6). Similarly with previous analysis (Chapter 5), the greatest restrictions in habitat availability occur with the smallest fish as they have the narrowest range of requirements. However, as discussed previously (Chapters 4 and 5) the younger trout may have broad habitat tolerances but are excluded by intraspecific completion with larger trout. Thus, it would seem that suitable habitat for the youngest cohorts may be a limiting factor to trout populations when all age classes are present. Unlike both depth and water velocity the assessment of streambed substrate cannot distinguish between wet and dry quadrats in the streambed. The availability of substrate particles sizes is recorded for the entire streambed but access to the entire habitat is only available during periods of high flow. As the substrate assessment will be joined with the depth and current velocity substrate for the final assessment (Chapter 7) this information will be regained. As well, it is important to keep in mind that substrate is assessed based on particle categories and is discrete data unlike depth and water velocities which are continuous measures. The categories used vary in size from clay, silt to boulders and bedrock as they are based on a $\log_2$ scale.

The assessment of the streambed substrate available to brown trout using the tolerance profiles was used to assesses the physical habitat available in the study reaches but there are improvements that can be made to the methodology. These improvements are centred on the attempt to quantitatively survey the streambed using the quadrats and by sampling individual stones. To begin, this was extremely time consuming. It took two people approximately one week per study reach to conduct the substrate survey. This would not be cost effective in most stream survey programs. Secondly, accurate measures were limited to particles sizes between 1 and 25 cm in diameter. This reduced the accuracy of substrate size classification for the assessment of fry habitat less than 1 cm and juvenile habitat greater than 25 cm. The visual qualitative measures most commonly used were able to distinguish between particle sizes ranging from clay and silt (0.0004-0.006 cm) to smooth bedrock (Heggenes et al. 1990). These qualitative visual measures and are known to differ, at times statistically significantly, between observers (Wohl et al. 1996). However they are almost universally utilized, are more time effective, and have greater resolution than the method proposed in this chapter.

Although not ideal, the best available substrate assessment methods remain the visual techniques employed by the authors listed in the literature summaries presented in Tables 6.1 to 6.4. They have the highest resolution and can be employed quickly and efficiently. As well, with training the variation between assessors can be reduced and the accuracy of the observation improved (Latulippe et al. 2001). Further, Bain et al. (1985) have developed techniques based on visual assessment and the substrate size scales similar to that outlined in Figure 6.1. In this method the substrate is visually assessed along transects in a stream and a code number is assigned to the substrate class at each interval. A mean and measure of variance (S.D. or S.E.) can be calculated for each transect based on the coded values. This technique could be modified for use with the quadrat system employed in this study. With the training outlined by Latulippe and his colleagues (2001) the variance and inaccuracies between assessors can be reduced. This would result in a semi-quantitative method that would include the full range of substrate present, have higher resolution, and be quicker and more cost effective than the techniques employed in this study.

Typically, the assessment of streambed substrate for the fry, juvenile, and adult size classes' required the visual assessments of surface particles at points within a study reach. Substrate used for spawning differed in that the bulk of the particle size analysis involved taking core samples from redds or areas thought to be used by trout to spawn. This is not possible for the monitoring program outlined in this study, as it would require hundreds (for example: 329 and 235 in March and Burnhouse Burn, respectively) of core samples if all quadrats were to be represented. This is not only expensive both in time and money but would be extremely destructive to the stream. Visual assessments of surface substrates used for spawning are required for monitoring purposes. Only four of the eleven studies used to construct the tolerance profile for spawning substrates used visual or surface measures. The remaining used core samples or unknown sampling methodologies. As monitoring will require visual assessment of surface substrates of spawning habitat more studies are required using surface assessment techniques to ensure that the tolerance profiles are representative of the surface particles sizes used by trout to spawn.

With a few exceptions, the bulk of the studies used to create these tolerance profiles came from northern Europe and North America (Tables 6.1 to 6.4). Populations of brown trout also occur in southern Europe, eastern Asia, and the southern hemisphere (Africa, Australia, and New Zealand). Including more studies from the southern populations of this species could strengthen the tolerance profiles and the subsequent habitat analysis. Most of the investigations involved sympatric populations, which is certainly not unusual for salmonids (Armstrong et al. 2003, Klemetsen et al. 2003). However, allopatric populations do occur and the tolerance profiles could be enhanced by a further understanding of trout habitat use in the absence of interspecific competition. Finally, studies involving non-spawning adult are sparse. Only six studies are included in the creation of this tolerance profile. Again, more studies would improve the confidence in using tolerance profiles to assess physical habitat availability for this age group.

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7.0 Integration

7.1 Introduction

In the previous chapters the physical habitat available for use by brown trout (*Salmo trutta*) was evaluated in a univariate way. However, trout, like other salmonids, will assess the suitability of microhabitat based on the quality of multiple variables which combine to form appropriate habitat (Shirvell and Dungey 1983, Heggenes and Saltveit 1990, Young 1995). I have already mentioned the importance of water depth (Chapter 4), water velocity (Chapter 5), and streambed substrate (Chapter 7) to the abundance and distribution of brown trout. These chapters examine the relative availability of these habitat features for use by brown trout in isolation. In order to complete the habitat assessment in March and Burnhouse Burn, the water depth, velocity and streambed substrate accessible for use needs to be combined.

The objective of this chapter is to amalgamate the information about habitat suitability contained in Chapters 4 to 6 regarding stream-depth, water velocity and streambed substrate to produce a final assessment of physical habitat availability in March and Burnhouse Burn. The analysis will be completed for the four life-history stages of *S. trutta* and will include a discussion of the suitability of each stream for the production and maintenance of sustainable populations of brown trout. Secondly, a secondary integration will be conducted for depth and water velocity by calculating Froude numbers for each quadrat in the wetted section of the stream. The appropriateness for brown trout residence and reproduction will be compared between streams and the validity of this assessment will be discussed. Finally, the strengths and weaknesses of using physical habitat to assess the suitability of stream reaches for brown trout will be outlined.

7.2 Methods

7.2.1 Physical habitat index

In the previous chapters a tolerance profile was constructed, based on a survey of the literature, that outlined a range of habitat that is suitable or useable by brown trout at each of the life stages [spawning, nursery (fry), juvenile and adult]. The one exception is streambed substrate used by trout to spawn, which was classified as suitable or unsuitable. For example, stream depth between 11 and 50 cm was considered suitable, depth between 5 and <11 cm or > 50 and 91 cm were considered useable, and stream depths outside of this range were thought to be unsuitable for spawning by brown trout (Figure 4.2). A profile of the habitat that could be used by trout was created by comparing these tolerance profiles with measures of depth, velocity and substrate in each quadrat within the study reaches. The data collected from the stream survey was compared with the tolerance profiles for each life stage of brown trout and the physical habitat availability in each stream was assessed and summarized. These figures are illustrated in Chapters 4 through 6. In this chapter, the three types of habitat assessment were then combined to produce a final integrated habitat assessment that outline the integrated physical habitat available for brown trout in these streams.

The first step in the integration process was to code the results from the initial habitat assessment. Each quadrat was determined to be suitable, useable, or unsuitable for depth, velocity and substrate. The suitable determination was coded as 2, useable as 1, and unsuitable as 0. This was done for all three habitat-parameters with one exception.

Spawning substrate was considered suitable (2) or unsuitable (0). For each quadrat in both streams a set of three coded numbers was determined. The product of these three numbers was then used to determine if a quadrat was suitable for use by trout and to determine the quality of that habitat. There are four possible results from all the combinations of habitat codes that result in useable habitat. Again the exception is the set involving spawning substrate, which has three possible results. These numbers were multiplied together because it would allow for the elimination of quadrats that were unsuitable for use by trout. For example, if substrate and depth were considered suitable (both coded 2) but the quadrat was too shallow (unsuitable - 0) the quadrate would be considered unsatisfactory (2 x 2 x 0 = 0) in the integrated analysis, as trout could not access the site. The possible outcomes from the coding process are outlined in Table 7.1. A summary of the proportion of quadrats classified by the quality index was created for each dataset and presented as tables and histograms.

Spawning substrate was considered separately because of the nature of salmonid reproduction. Salmon have been observed spawning in substrate outside the range determined by the tolerance profiles outlined in Chapter 6 (Figure 6.3). For example, Crisp (1993) comments that trout have been observed spawning in substrates normally considered too small (sand). Although a spawning female may use substrates at the extreme of the tolerance range the survivorship of the eggs and resulting embryo's may be affected (Rubin et al. 2004). In order to include survivorship into the habitat assessment the margins of the tolerance profile were excluded and habitat was considered only as suitable or unsuitable.

Table 7.1: Possible combination of habitat suitability codes and quality evaluation for sampling quadrats containing habitat useable by brown trout (*Salmo trutta*).

Depth	Water Velocity	Streambed Substrate	Product	Quality
2	2	2	8	Very high
2	2	1	4	High
2	1	2	4	High
1	2	2	4	High
2	1	1	2	Good
1	2	2	2	Good
1	1	2	2	Good
1	1	1	1	Satisfactory
Spawning Substrate				
2	2	2	8	Very high
2	1	2	4	High
1	2	2	4	High
1	1	2	2	Good

Habitat assessment maps were again created using the results of the coding and integration process for the downstream sites of March and Burnhouse Burns. The integrated maps show the relative abundance of habitat that could be used and that which could not for each of the life stages of brown trout and the spatial distribution of this habitat with each reach. Shading was used to illustrate the quality of habitat ranging from satisfactory to very high in quadrats that could be used by trout. Unshaded quadrats represent physical habitat that is unsuitable for use. Statistical analysis used has been outline previously in Chapter 3 (Section 3.6).

7.2.2 Froude Number

Stream physical habitat parameters can also be integrated by calculating the Froude number. Froude number is the dimensionless velocity/depth ratio $Fr = V_m/\sqrt{(gY)}$, where V_m is the mean water column velocity, Y the water depth, and g the acceleration due to gravity (9.81 m/sec). It is used in two ways. It can define tranquil or sub-critical flow (where $Fr < 1$) and rapid or super critical flow (where $Fr > 1$). Alternately it can be used to define habitat type in that pools are associated with Froude number < 0.18, riffles with Froude number > 0.41, and run habitats with intermediate values (≥0.18 and ≤ 0.41) (Jowett 1993, Tetzlaff et al. 2005a). Froude numbers were calculated for all wetted quadrats in each study reach. Each study reach was then summarized according to the proportion of pool, run, and riffle habitat and the proportion of sub-critical and super-critical flow.

7.3 Results

7.3.1 Integrated quality index

7.3.1.1 Integrated spawning habitat assessment

The proportion of physical habitat assessed as very-high, high, good, low, or not-suitable in quality (see Chapter 3 for assessment methods) was compared using a chi-squared test. This was done to compare the seven datasets to see if the proportion of quadrats classified according to the five quality criteria differed between sites for the total and wetted streambeds. The results of this analysis are presented in Tables 7.2 and 7.3. The chi-squared test for both the total ($X^2 = 263.3$, df = 18, p = 0.000) and wetted ($X^2 = 205.2$, df = 18, p = 0.000) streambed, based on the integrated criteria for spawning habitat, indicates that the relative proportion of the quadrates classified into the integrated habitat categories are not independent of reach (i.e. they are statistically different). The proportion of quadrats categorized based on the integrated criteria for each site is outlined in Table 7.4 and illustrated in Figure 7.1 (A-G). The greatest

Table 7.2: Results for the chi-square test that compared the proportions of quadrats classified by integrated quality values (very high, high, good, low, satisfactory, and not-useable) between the seven datasets for both the total and wetted streambed.

Life Stage	Streambed	X^2 value	DF	Cells < 5†	p-value	X^2 (Likelihood)*
Spawn	Total	263.3	18	n/a	0.000	no
	Wetted	205.2	18	n/a	0.000	no
Nursery	Total	276.7	18	n/a	0.000	no
	Wetted	122.2	18	n/a	0.000	no
Juvenile	Total	577.8	24	n/a	0.000	no
	Wetted	335.5	24	n/a	0.000	no
Adult	Total	240.0	18	8	0.000	yes
	Wetted	131.3	18	8	0.000	yes

† Number of cells with expected values less than 5; * Used likelihood ratio chi-square

Table 7.3 Results for the chi-square tests that compared the proportions of quadrats classified by integrated quality values (very high, high, good, low, satisfactory, and not-useable) between the Bin Burn datasets (before and after the spate) for both the total and wetted streambed.

Life Stage	Streambed	X^2 value	DF	Cells < 5†	p-value	X^2 (Likelihood)*
Spawn	Total	12.4	3	n/a	0.006	no
	Wetted	16.9	3	n/a	0.001	no
Nursery	Total	96.4	3	n/a	0.000	no
	Wetted	34.8	3	n/a	0.000	no
Juvenile	Total	122.1	4	n/a	0.000	no
	Wetted	53.3	4	n/a	0.000	no
Adult	Total	82.2	3	n/a	0.000	no
	Wetted	39.4	3	n/a	0.000	no

† Number of cells with expected values less than 5; * Used likelihood ratio chi-square

proportion of quadrats that was assessed as very-high quality for spawning trout in the total streambed was observed in March Burn-upstream (37.6%) but more typically very-high quality physical habitat ranged from 0.0% (Burnhouse Burn-upstream) to 1.3% (Burnhouse Burn-downstream). High quality spawning habitat in the total streambed was again highest at March Burn-upstream (18.6%) although the second highest was observed at Burnhouse Burn-downstream after the spate (16.3%). The least amount of

Table 7.4: Summary of brown trout (*Salmo trutta*) physical habitat assessment based on integrated criteria for both the total and wetted streambeds for each of the four life stages.

		Spawn Total (%)	Wetted (%)	Nursery Total (%)	Wetted (%)	Juvenile Total (%)	Wetted (%)	Adult Total (%)	Wetted (%)
March Burn Upstream	Very High	37.6	39.3	42.7	44.6	28.5	29.8	9.5	9.9
	High	18.6	19.4	52.4	54.7	39.4	41.1	44.9	46.9
	Good	5.3	5.5	0.0	0.0	27.2	28.4	1.5	1.6
	Low	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Not-useable	38.5	35.8	4.9	0.7	4.9	0.7	44.1	41.6
March Burn Downstream	Very High	0.3	0.5	14.4	23.7	1.9	3.1	4.4	7.2
	High	6.2	10.2	32.4	53.3	23.5	38.6	14.7	24.1
	Good	1.2	2.0	10.9	17.9	29.4	48.4	0.0	0.0
	Low	0.0	0.0	0.0	0.0	4.4	7.2	0.0	0.0
	Not-useable	92.3	87.3	42.3	5.1	40.8	2.7	80.9	68.7
Burnhouse Burn Upstream	Very High	0.0	0.0	12.6	32.7	0.8	2.1	0.0	0.0
	High	1.7	4.3	19.5	50.6	11.0	28.4	4.1	10.7
	Good	2.3	6.0	5.0	12.9	24.5	63.5	0.0	0.0
	Low	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Not-useable	96.0	89.7	62.9	3.8	63.7	6.0	95.9	89.3
Burnhouse Burn Downstream	Very High	1.3	3.0	20.8	49.2	0.4	1.0	0.0	0.0
	High	6.4	15.1	14.6	34.5	21.9	51.9	7.2	17.1
	Good	6.8	16.0	1.7	4.0	13.3	31.5	0.0	0.0
	Low	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Not-useable	85.5	65.9	62.9	12.3	64.4	15.6	92.8	82.9
Bin Burn Upstream	Very High	1.2	1.6	25.8	34.4	15.6	20.8	12.7	16.9
	High	16.3	21.7	21.4	28.5	22.6	30.1	18.5	24.7
	Good	0.6	0.8	14.2	18.9	17.5	23.4	1.2	1.6
	Low	0.0	0.0	0.0	0.0	17.1	22.8	0.0	0.0
	Not-useable	81.9	75.9	38.6	18.2	27.2	2.9	67.6	56.8

Table 7.4: Continued

		Spawn Total (%)	Wetted (%)	Nursery Total (%)	Wetted (%)	Juvenile Total (%)	Wetted (%)	Adult Total (%)	Wetted (%)
Bin Burn Downstream (Pre-spate)	Very High	9.2	13.6	22.5	33.3	5.9	8.7	2.3	3.4
	High	10.6	15.7	31.2	46.2	33.0	48.7	19.7	29.1
	Good	1.7	2.5	8.5	12.6	23.1	34.2	2.8	4.2
	Low	0.0	0.0	0.0	0.0	4.0	5.9	0.0	0.0
	Not-useable	78.5	68.2	37.8	7.9	34.0	2.5	75.2	63.3
Bin Burn Downstream (Post-spate)	Very High	4.0	9.7	18.4	44.1	1.3	3.1	1.7	4.2
	High	12.0	28.9	18.5	44.5	28.0	67.1	4.5	10.8
	Good	1.7	4.2	0.6	1.4	8.0	19.2	0.5	1.1
	Low	0.0	0.0	0.0	0.0	0.3	0.8	0.0	0.0
	Not-useable	82.2	57.2	62.5	10.0	64.2	9.7	93.3	83.9

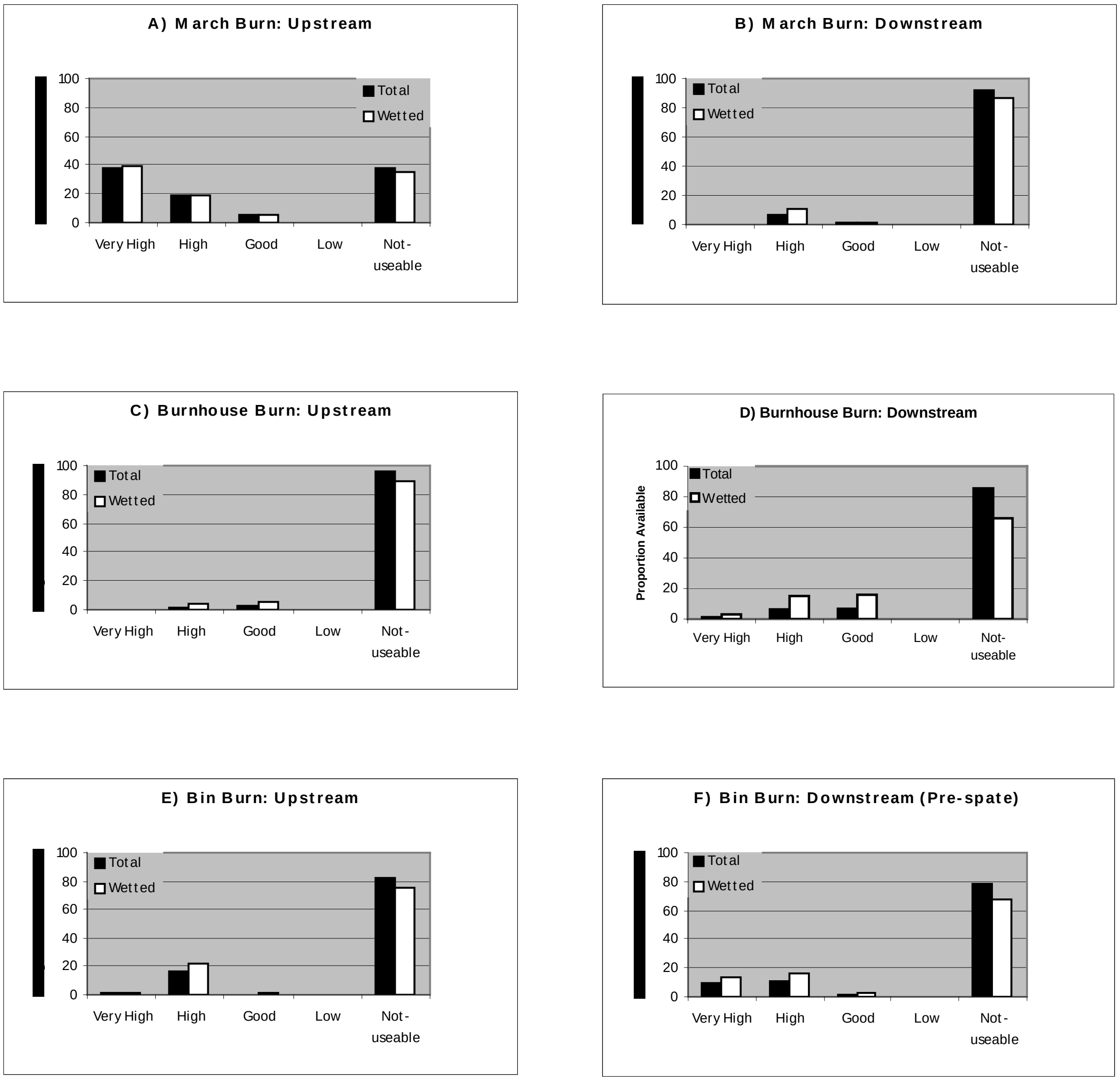
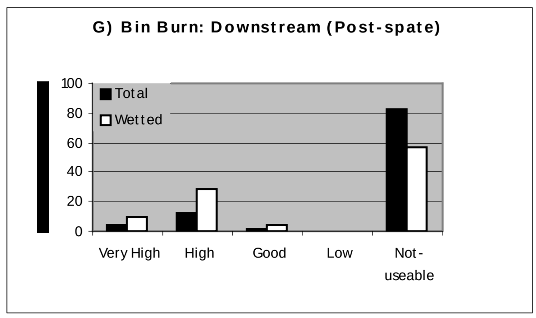


Figure 7.1 (A-G): Histograms of the proportion of streambed, both total and wetted, available for spawning brown trout at all study sites classified using the integrated quality criteria (i.e. very high to low and not-useable).



high quality physical habitat was seen at Burnhouse Burn-upstream (1.7%). Physical spawning habitat assessed to be good quality ranged from 0.6% (Bin Burn-upstream) to 6.8% (Burnhouse Burn-downstream). There was no physical habitat classified as low-quality. The proportion of the quadrats surveyed in the total streambed considered not-useable for spawning ranged from a high of 96% at Burnhouse Burn-upstream to 38.5% and March Burn-upstream.

In the wetted portion of the streambed the highest proportion of very-high quality spawning habitat was observed at March Burn-upstream (39.3%) and the least amount at Burnhouse Burn-upstream where there were no quadrats with this classification. High quality spawning habitat ranged from 28.9% to 4.3% at Bin Burn-downstream (Post-spate) to Burnhouse Burn-upstream, respectively. Good quality physical habitat in the wetted portions of these streams ranges from 0.8% at Bin Burn-upstream to 16.0% to Burnhouse Burn-downstream. Again, there were no quadrats classified as low habitat and the not-useable portion of the wetted streambeds ranged from 35.8% to 89.7% at March Burn-upstream to 89.7% at Burnhouse Burn-upstream. Obviously the dry portions of the streambed will unavailable for use by any age class of brown trout, however, habitat that falls outside of the tolerance limits for each life stage is also not accessible (not-useable). The actual surface area available for use by trout is the proportion of the surface area that has been classified as suitable (very-high to low but not include the 'not-useable quadrats) multiplied by the total area. These calculations were completed and are presented in Table 7.5 for all sites and all age classes. The smallest surface area with physical habitat that has been classifies as useable (to varying degrees) by spawning trout was found at Burnhouse Burn-upstream where only 1.2 m^2

Table 7.5: The total area that is useable by brown trout for each study reach and each age class. Total accessible area calculated by multiplying total area by proportion of the streambed that was rate as useable (very-high, high, good, or low).

Age Class	Site	Total area of streambed (m^2)	Proportion of total area useable (%)	Total area available (m^2)
Spawning	March Burn-us†	12.6	61.5	7.7
	March Burn-ds‡	80.5	7.8	6.3
	Burnhouse Burn-us	30.1	4.0	1.2
	Burnhouse Burn-ds	58.8	14.4	8.5
	Bin Burn-us	43.0	18.0	7.8
	Bin Burn-ds (pre spate)	87.7	21.6	18.9
	Bin Burn-ds (post spate)	216.3	17.8	38.5
Nursery	March Burn-us	12.6	95.1	11.9
	March Burn-ds	80.5	57.8	46.6
	Burnhouse Burn-us	30.1	37.1	11.2
	Burnhouse Burn-ds	58.8	37.1	21.8
	Bin Burn-us	43.0	61.3	26.4
	Bin Burn-ds (pre spate)	87.7	62.3	54.7
	Bin Burn-ds (post spate)	216.3	37.5	81.1
Juvenile	March Burn-us	12.6	95.1	11.9
	March Burn-ds	80.5	59.3	47.7
	Burnhouse Burn-us	30.1	36.3	10.9
	Burnhouse Burn-ds	58.8	35.7	21.0
	Bin Burn-us	43.0	72.8	31.3
	Bin Burn-ds (pre spate)	87.7	66.0	57.9
	Bin Burn-ds (post spate)	216.3	37.6	81.3
Adult	March Burn-us	12.6	56.0	7.0
	March Burn-ds	80.5	19.0	15.3
	Burnhouse Burn-us	30.1	4.1	1.3
	Burnhouse Burn-ds	58.8	7.2	4.3
	Bin Burn-us	43.0	32.3	13.9
	Bin Burn-ds (pre spate)	87.7	24.8	21.8
	Bin Burn-ds (post spate)	216.3	6.7	14.5

† upstream; ‡ downstream

of the wetted streambed was classified as having the appropriate combination of depth, velocity, and streambed substrate. There was 8.5 m^2 of spawning habitat at Burnhouse Burns's downstream site. It might be expected that the upstream sites would have less available habitat then the downstream sites, at least in small streams of this scale. However, in March Burn a slightly larger surface area was available at the upstream site

(7.7 m^2 upstream and 6.3 m^2 downstream). Bin Burn had the most habitat classified as useable for spawning trout. At the upstream site 7.8 m^2 was available and 18.9 m^2 and 38.5 m^2 were available at the downstream site, pre and post spate respectively.

What is most notable in the assessment of the area available in the wetted portion of the streams that has been classifies as useable for spawning brown trout is the large increase in available habitat in Bin Burn-downstream after the spate and reconstruction. The chi-squared test that compares the relative proportion of the quadrats classified according to the quality criteria indicates that these proportion are different, before and after the spate, in both the total (X^2 = 12.4, df = 3, p = 0.006) and wetted (X^2 = 16.9, df = 3, p = 0.001) portions of this reach. Thus, both the total area available for spawning increased after the spate and the relative proportions of the habitat classifications have changed.

7.3.1.2 Integrated nursery habitat assessment (fish length ≤ 7 cm)

The chi-squared analysis of the relative proportion of the quadrats assessed according to the integrated criteria for young brown trout (≤ 7 cm) revealed that these proportions differ, statistically, for the total and wetted surfaces between the seven data sets examined (X^2 = 276.7, df = 18, p = 0.000 and X^2 = 122.2, df = 18, p = 0.000, for the total and wetted streambed, respectively - Table 7.2 and Table 7.3). The relative proportion of the integrated physical habitat criteria observed at the study reaches is outlined in Table 7.4 and illustrated in Figure 7.2 (A-G). Unlike the analysis for spawning habitat, a large proportion of the integrated physical habitat assessed in the total streambed has been classified as very-high quality nursery habitat for young trout. The greatest proportion of very-high quality habitat was found at March Burn-upstream (42.7%) and the

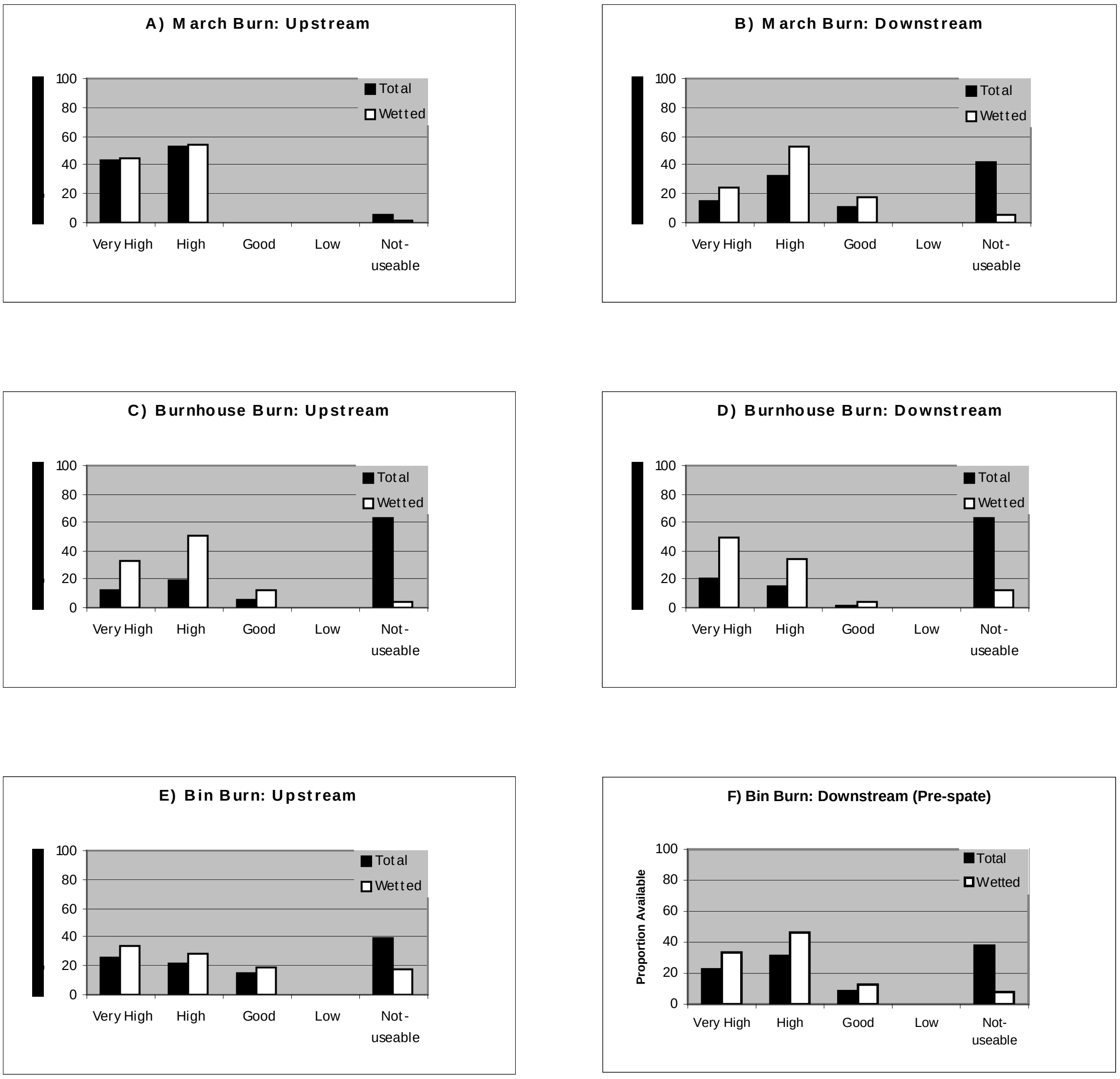
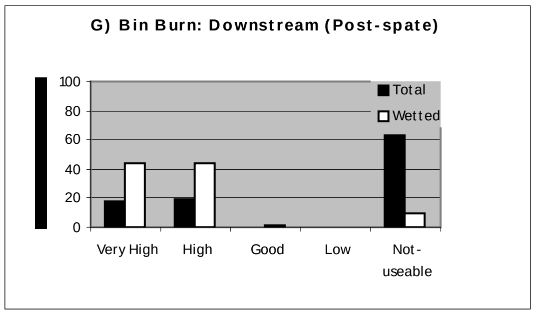


Figure 7.2 (A-G): Histograms of the proportion of streambed, both total and wetted, available to young trout (≤ 7 cm) at all study sites classified using the integrated quality criteria (i.e. very high to low and not-useable).



smallest proportion at Burnhouse Burn-downstream (12.6%). Much of the remaining useable habitat was considered high quality: ranging from 14.6 (Burnhouse Burn-downstream) to 52.4% (March Burn-upstream). There was no habitat classified as good at March Burn-upstream and very little at the remaining sites [0.6% to 14.2 at Bin Burn-downstream (post-spate) and Bin Burn-upstream, respectively]. The was no habitat classified as low quality and the not-useable habitat ranged from 4.9% at March Burn-upstream to 62.9% at both Burnhouse Burn sites.

In the wetted portion of the streambed an even greater portion of the quadrats were considered high or very-high quality. Very-high quality quadrats ranged from 23.7% to 49.2% at March Burn-downstream and Burnhouse Burn-downstream, respectively. High quality quadrats ranged in abundance from 28.5% (Bin Burn-upstream) to 54.7% (March Burn-upstream). None of the useable physical habitat was classified as good at either of the March Burn sites or at Bin Burn-downstream (Pre-spate). The remaining sites had intermediate quality habitat classifications ranging from 1.4% (Bin Burn-downstream (Post-spate) to 18.9% (Bin Burn-upstream). There were no sites with physical habitat classified as a low-quality. The proportion of the wetted streambeds considered not-useable by young trout ranged from 0.7% (March Burn-upstream) to 18.2% in Bin Burn-upstream.

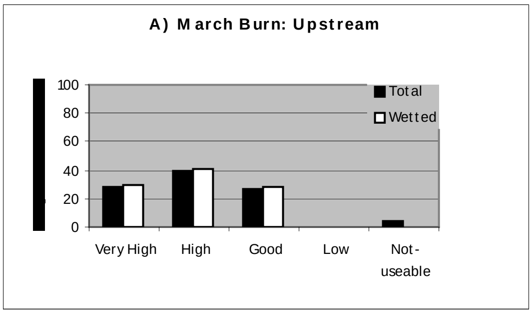
The area available in the streambeds with integrated physical habitat classifications considered useable (very-high, high, good, or low quality) was greater using the nursery classifications when compared with the spawning assessment. The smallest area available to young was found in Burnhouse Burn with 11.2 m^2 and 21.8 m^2 at the upstream and downstream sites, respectively. March Burn-upstream (11.9 m^2) had a similar useable area as Burnhouse Burn-upstream; however, there was more than double the amount of habitat classified as useable at the downstream March Burn site (46.2 m^2). Bin Burn has the largest surface area classified as nursery habitat of all the burns studied. The upstream site had 26.4 m^2 classified as useable and the downstream site had 54.7 m^2 and 81.1 m^2 before and after the spate, respectively.

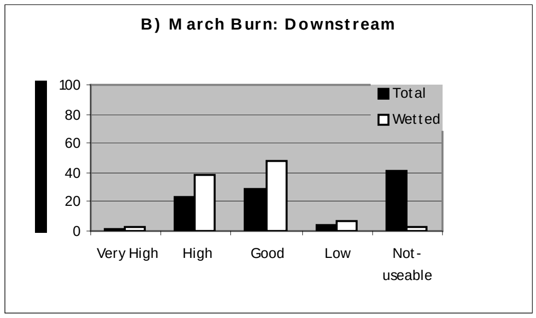
Like the assessment for spawning habitat, a larger amount of the streambed was classified as useable after the spate and reconstruction at the downstream Bin Burn site. The chi-squared test of the relative proportions of the quadrats classified using the integrated methodology at Bin Burn before and after the spate indicates that there are differences between the distributions of these classification criteria in both the total ($X^2 = 96.4$, df = 3, p = 0.000) and the wetted ($X^2 = 34.8$, df = 3, p = 0.000) portions of the streambed. As previously mentioned the discharge at the time of sampling was similar before and after the spate (Table 4.8); however, the total area accessible has increased (Table 7.5). Despite the statistical difference in the relative proportions of the integrated habitat classifications the bulk of quadrates that were considered useable by young trout were classified into the very-high or high quality categories for both datasets in both the wetted and total streambeds.

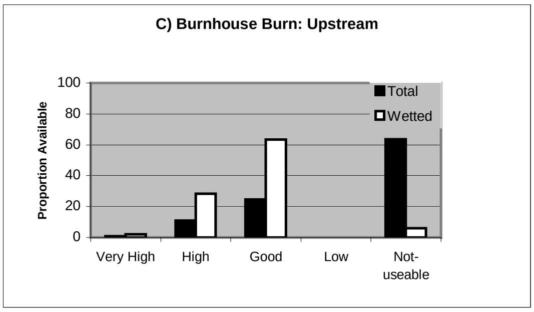
7.3.1.3 Integrated juvenile habitat assessment (fish length > 7 to 20 cm)

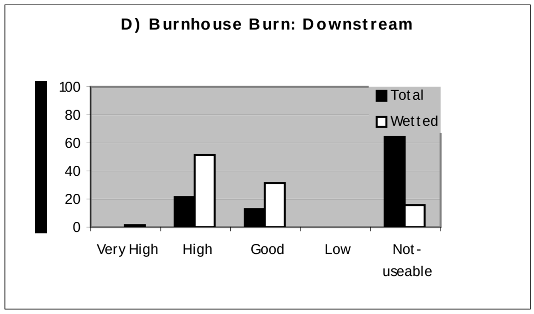
The chi-square test reveals that there is a statistical relationship between the data sets and the integrated habitat quality assessment based on the criteria developed for juvenile trout (fish length > 7 cm to 20 cm). The results of the chi-square test for the total and wetted surface portions of the stream are $X^2 = 577.8$, df = 24, p = 0.000 and $X^2 = 335.5$, df = 24, p = 0.000, respectively. Like nursery habitat there is a large proportion of the useable physical habitat in the total and wetted portions of the stream that are well suited for juvenile trout (Table 7.3 and Figure 7.3). Unlike the assessment for nursery habitat there seems to be a shift in the dominance from very-high and high rated habitat to high and good rated habitat. In the total streambed, very-high rated physical habitat ranged from 0.4% at Burnhouse Burn-downstream to 28.5% at March Burn-upstream. High rated habitat ranged from 11.0% to 39.3% at Burnhouse Burn-upstream and March Burn, respectively. The intermediate classification was much more commonly encountered than in spawning and nursery assessments. Good rated habitat ranged from 8.0% [Bin Burn-downstream (post-spate)] to 29.4% (March Burn-downstream). Low quality physical habitat ranged from 0% at March Burn-upstream, and the Burnhouse Burn sites to 17.1% at the upstream Bin Burn site. The portion of the total streambed that was considered not-useable ranged from 4.9% at March Burn-upstream to 64.4 % at Burnhouse Burn-downstream.

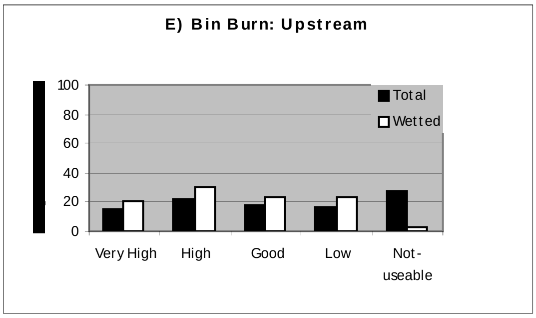
The basic patterns observed in the total streambed were repeated in the wetted portion of the streambed. Physical habitat rated as high-quality ranged from 1.0% at Burnhouse Burn-downstream to 29.8% at March Burn-upstream. The most commonly encountered classification was high quality in assessment based on the criteria for juvenile Trout. High quality ranged from 28.4% (Burnhouse Burn-upstream) to 67.1% (Bin Burn-downstream (post-spate). Good quality habitat ranged from 19.2% observed at Bin Burn-downstream (post-spate) to 63.5% at March Burn-downstream. Low quality habitat was not observed at March Burn-upstream and the Burnhouse Burn sites. Low-quality physical habitat ranged from 0.8% [Bin Burn-downstream (post-spate)] to 22.8% (Bin Burn-upstream) at the remaining sites. There was very little physical habitat that was considered not-useable using the criteria for juvenile trout in the wetted











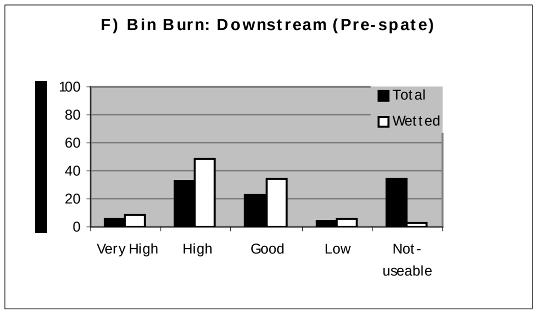
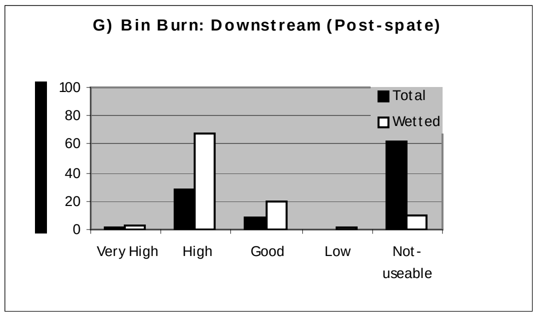


Figure 7.3 (A-G): Histograms of the proportion of streambed, both total and wetted, available to juvenile trout (> 7 cm to 20 cm) at all study sites classified using the integrated quality criteria (i.e. very high to low and not-useable).



portions of this stream. Habitat not accessible to juvenile trout ranged from 0.7% (March Burn-upstream) to 15.6% (Burnhouse Burn-downstream).

The total area of streambed available to juvenile trout, based on the integrated physical habitat criteria was the greatest at the post-spate downstream site on Bin Burn (81.3 m^2- Table 7.5). As observed in pervious assessment the surface area available to this age class increased after the spate and reconstruction. The surface area available before the spate was 57.9 m^2. The smallest area available for juvenile trout was calculated for Burnhouse Burn-upstream (10.9 m^2), and the next smallest was March Burn-upstream (11.9m^2). The remaining sites had intermediate values. The trends observed for the streambed area available based on the criteria for young trout (i.e. nursery habitat) were similar to that seen in the juvenile habitat with the downstream sites at all stream having large available surface areas than the upstream sites.

The spate and reconstruction of the study reach at Bin Burn (downstream site) resulted in different proportions of streambed quality based on the integrated criteria developed for juvenile trout. The chi-squared test for both the total and wetted portion of the streambed indicated a relationship between quality criteria and site (X^2 = 122.1, df = 4, p = 0.000 and X^2 = 53.3, df = 4, p = 0.000; total and wetted, respectively). The surface area thought to be available did increase after the spate (Table 7.5); however, most of the useable habitat was still classified as high or good quality in both the total and wetted streambeds. Although the amount of physical habitat increased, and statistically the quality differed before and after the spate, the site was still assessed to contain largely high to good quality habitat for juvenile trout.

7.3.1.4 Integrated adult habitat assessment (fish length > 20 cm)

The chi-square test reveals that there is a statistical relationship between the data sets and the integrated habitat quality assessment based on the criteria developed for adult trout (fish length > 20 cm). The results of the chi-square test for the total and wetted portions of the streambed are $X^2 = 240.0$, df = 18, p = 0.000 and $X^2 = 131.3$, df = 18, p = 0.000, respectively (Table 7.2 and Table 7.3). A higher proportion of the habitat was considered not-useable based on the integrated adult criteria than for any of the other resident life stage criteria (Table 7.4 and Figure 7.4). The proportion of the physical habitat considered very high within the total streambed ranged from 0% at the Burnhouse Burn sites to 12.7% at Bin Burn-upstream. High-quality habitat ranged from 4.1% (Burnhouse Burn-upstream) to 44.9% (March Burn-upstream). A small proportion of the physical habitat surveyed was classified into the intermediate (good) category. There was no good-quality habitat observed at either of the Burnhouse Burn sites or the March Burn-downstream site and the remaining sites ranged from 0.5% [Bin Burn-downstream (post-spate)] to 2.8% [Bin Burn-downstream (pre-spate)]. None of the sites had what was considered low-quality physical habitat and the proportion of the total streambed assessed to be not-useable by adult brown trout ranged from 44.1% (March Burn-upstream) to 95.9% (Burnhouse Burn-upstream).

Similar trends were observed in the wetted portion of the stream. Physical habitat thought to be of very high quality for adult trout was not found at the either of the Burnhouse Burn sites and ranged from 3.4% [Bin Burn-downstream (pre-spate)] to 16.9% at Bin Burn-upstream. High-quality physical habitat was found least often at Burnhouse Burn-upstream (10.7% of the quadrats) and most often at March Burn-upstream (46.9%). Habitat rated as good-quality was not found at the Burnhouse Burn-

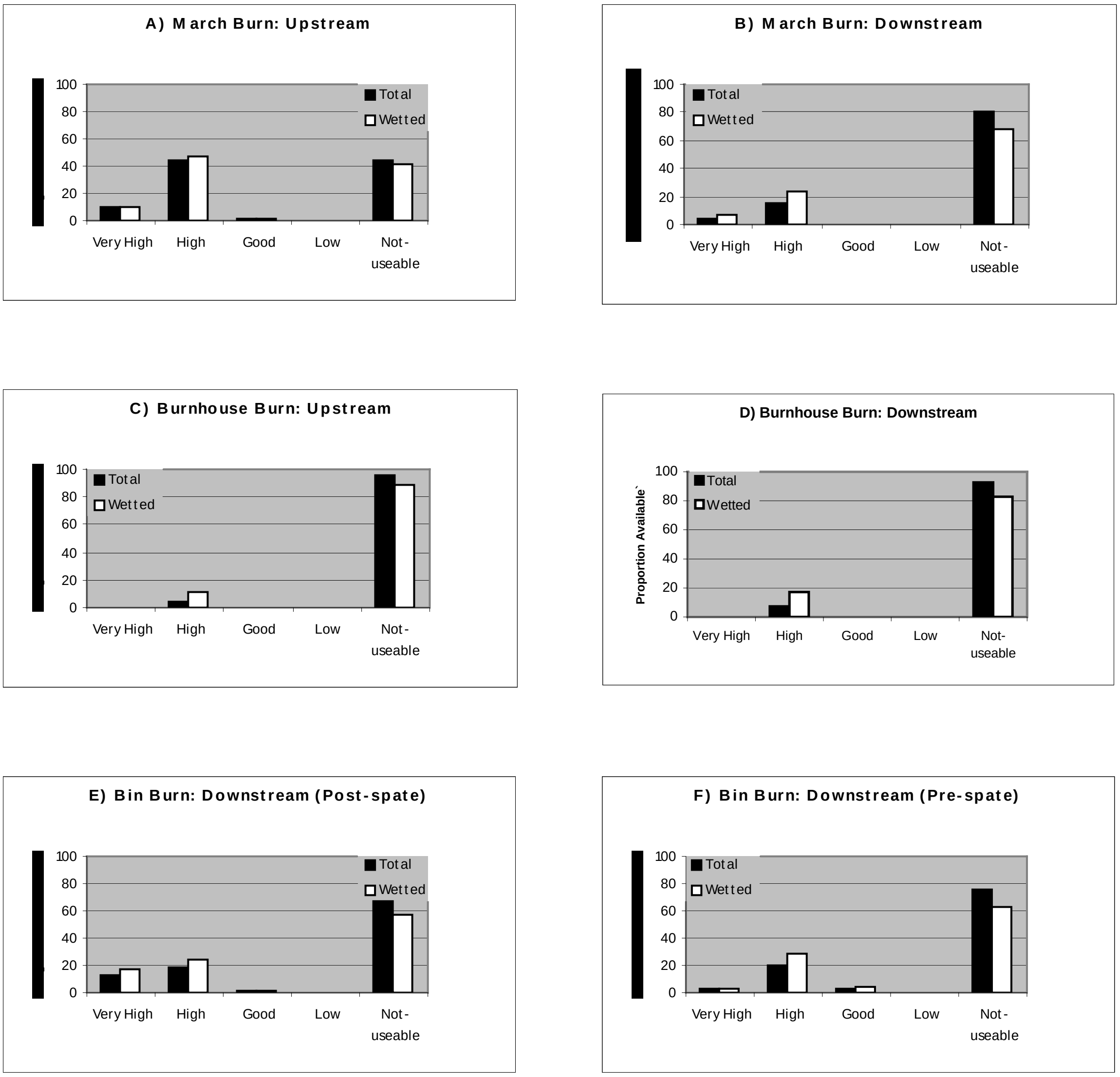
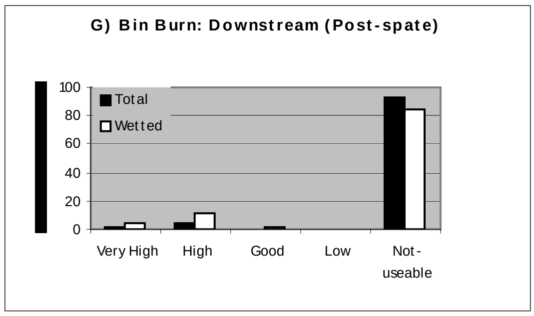


Figure 7.4 (A-G): Histograms of the proportion of streambed, both total and wetted, available to adult trout (> 20 cm) at all study sites classified according to using the integrated quality criteria (i.e. very high to low and not-useable).



sites or at March Burn-downstream. The remaining sites ranged from 1.1% [Bin Burn-downstream (post-spate)] to 4.2% at the same site before the spate. There was no low-quality habitat observed at any of the sites and habitat considered not-useable by adult trout in the wetted portions of the stream ranged from 41.6% (March Burn-upstream) to 89.3% at Burnhouse Burn-upstream.

The area available for use by adult trout in the streams studied in the Carron Valley, based on the integrated habitat criteria, is highlighted in Table 7.4. The smallest total area available for any life stage of resident trout was observed for adult trout in Burnhouse Burn-upstream (1.3 m^2) and very little habitat was available at the downstream site (4.3 m^2). Unlike the other life stages the Bin Burn downstream post spate assessment contained a smaller total area (14.5 m^2) compared to the pre-spate assessment (21.8m^2); however, the upstream site still had a smaller available surface area that either of the downstream sites (13.9 m^2). Typically, March Burn had intermediate values for useable area: 7.0 m^2 and 15.3m^2, for the up- and downstream sites, respectively.

As observed in the previous life stages, the chi-squared test that examines the relationship between the pre-and post distribution of the integrated habitat criteria based on the requirement of adult trout in both the wetted and total streambed indicate that the proportion of these quality criteria are dependent on site (X^2 = 82.2, df = 3, p = 0.000 and X^2 = 39.4, df = 3, p = 0.000 for the total and wetted streambed respectively – Table 7.2 and Table 7.3). Interestingly, the total area classified as useable by adult trout decreased in the post spate assessment although there was very little habitat available for adult trout when compared to the assessments based on the criteria for other resident life stages. The bulk of the quadrats surveyed were considered not-useable for adult trout and the remaining quadrats were primarily classed as high quality in both the pre- and post-spate assessments.

7.3.2 Habitat assessment maps

The integrated summary of available physical habitat provided a much more comprehensive assessment of microhabitat accessible to brown trout in the study reaches than does the assessments based on single physical variables. A graphical representation of the downstream sites on March and Burnhouse Burns illustrates the physical habitat available for spawning (Figure 7.5), nursery (Figure 7.6), juvenile (Figure 7.7) and adult trout (Figure 7.8) based on the integrated criteria. Unlike the previous maps only the suitable portions of the streambed are shaded. The dry and unsuitable portions of the streambed are indistinguishable while the streamsides are coloured black.

These habitat-maps are a very useful tool in that they clearly demonstrate the patterns of physical habitat available to the four life stages of brown trout as assessed using the integrated criteria presented in the previous tables and figures. A sequential examination of these figures (Figure 7.5 through 7.8) highlights the findings of the assessment: that there is a small proportion of the streambeds that can be used for spawning, that the most habitat is available to young trout and that the streams become increasingly poorly suited for residence as fish grow in size. It had been noted in the previous sections that the best habitat for spawning was located just downstream of the pool (Figure 7.5), a finding that was confirmed by the integrated assessment. Further, the best habitat for larger trout has been determined to be within the pool itself. In

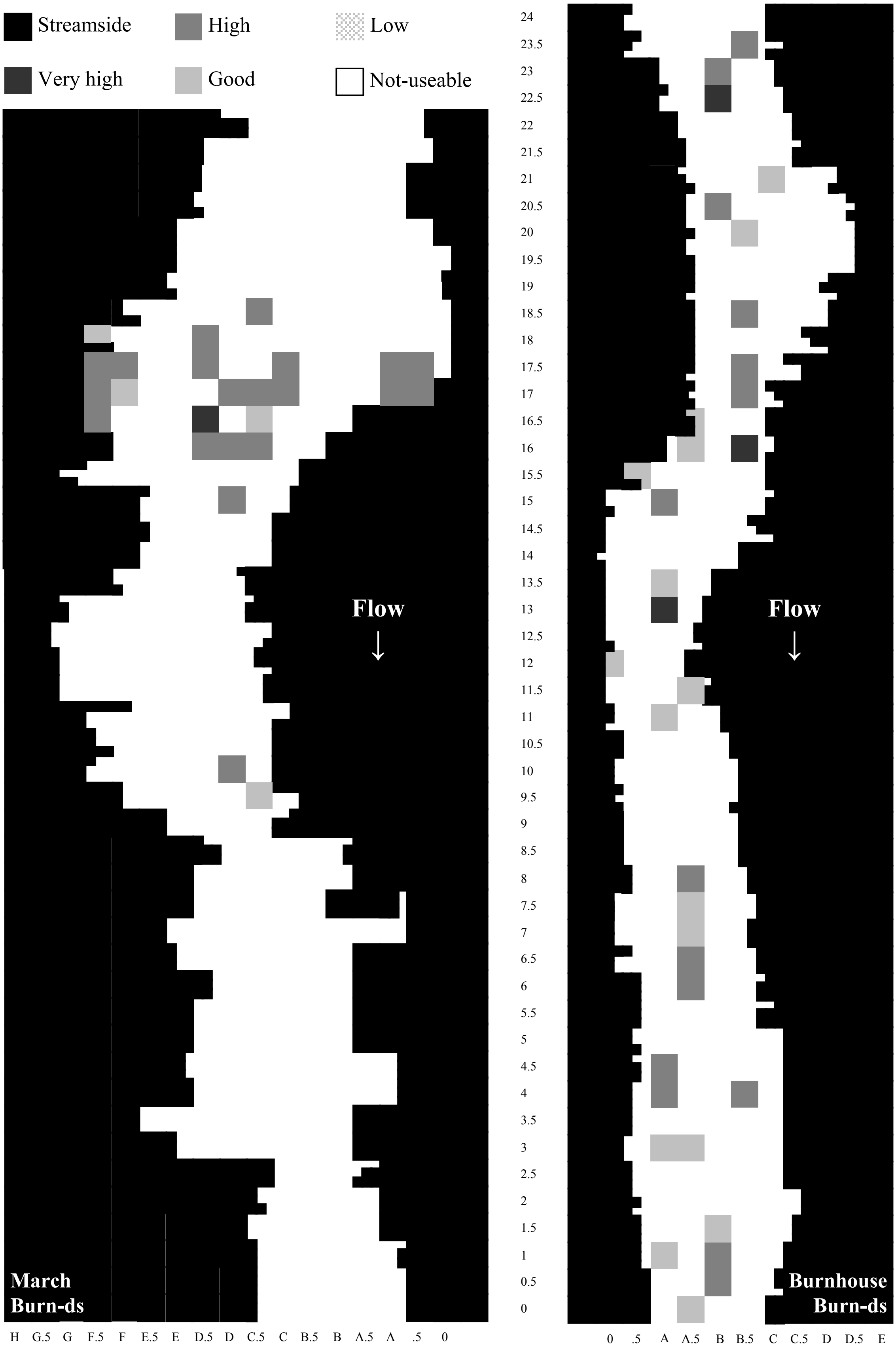


Figure 7.5: Habitat available for spawning brown trout (*Salmo trutta*) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec). This habitat assessment incorporates depth, velocity and substrate survey data.

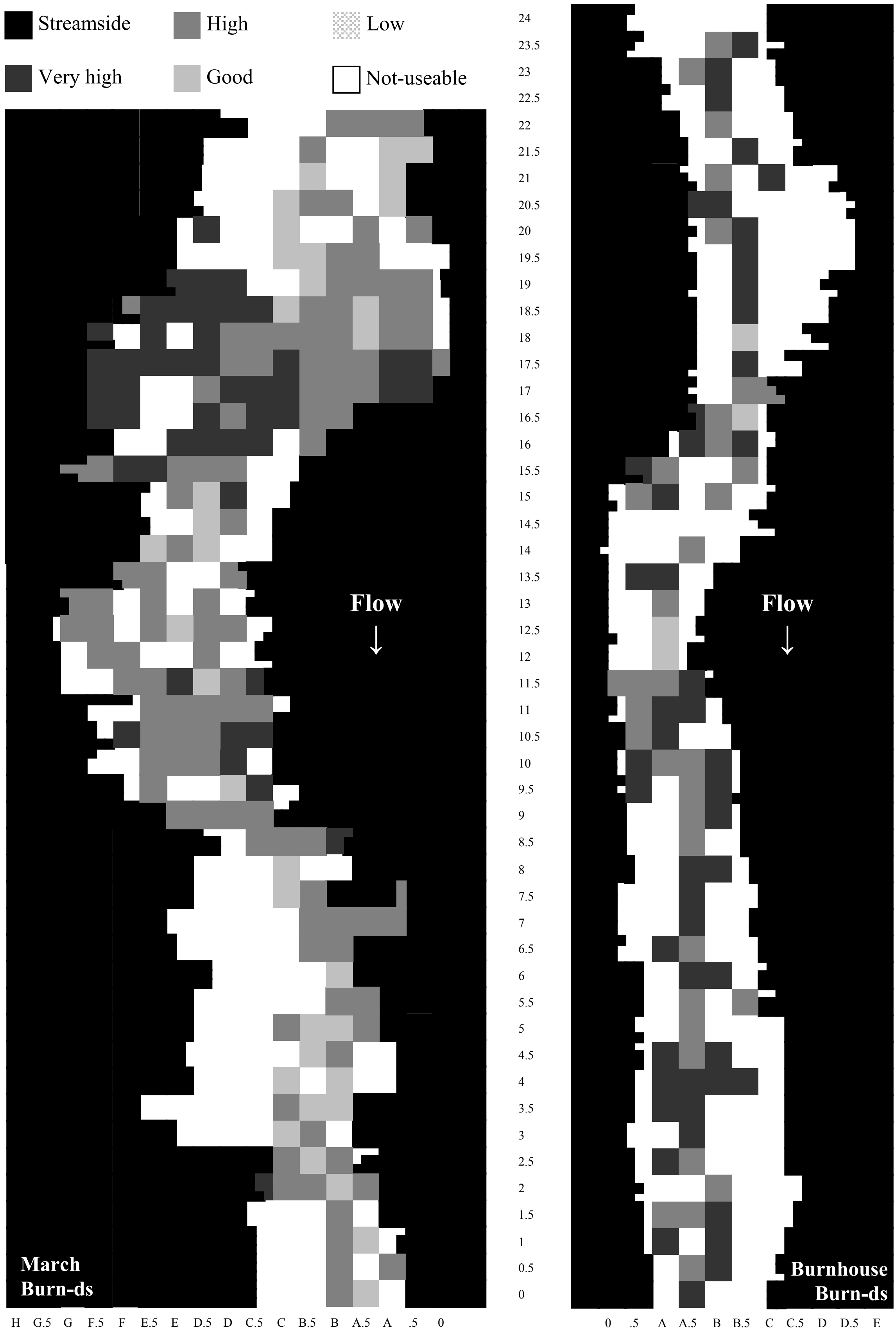


Figure 7.6: Habitat available for brown trout fry (*Salmo trutta*; length ≤ 7cm) in March (Q = 0.341 m^3/sec) and Burnhouse Burn (Q = 0.0058 m^3/sec). This habitat assessment incorporates depth, velocity and substrate survey data.

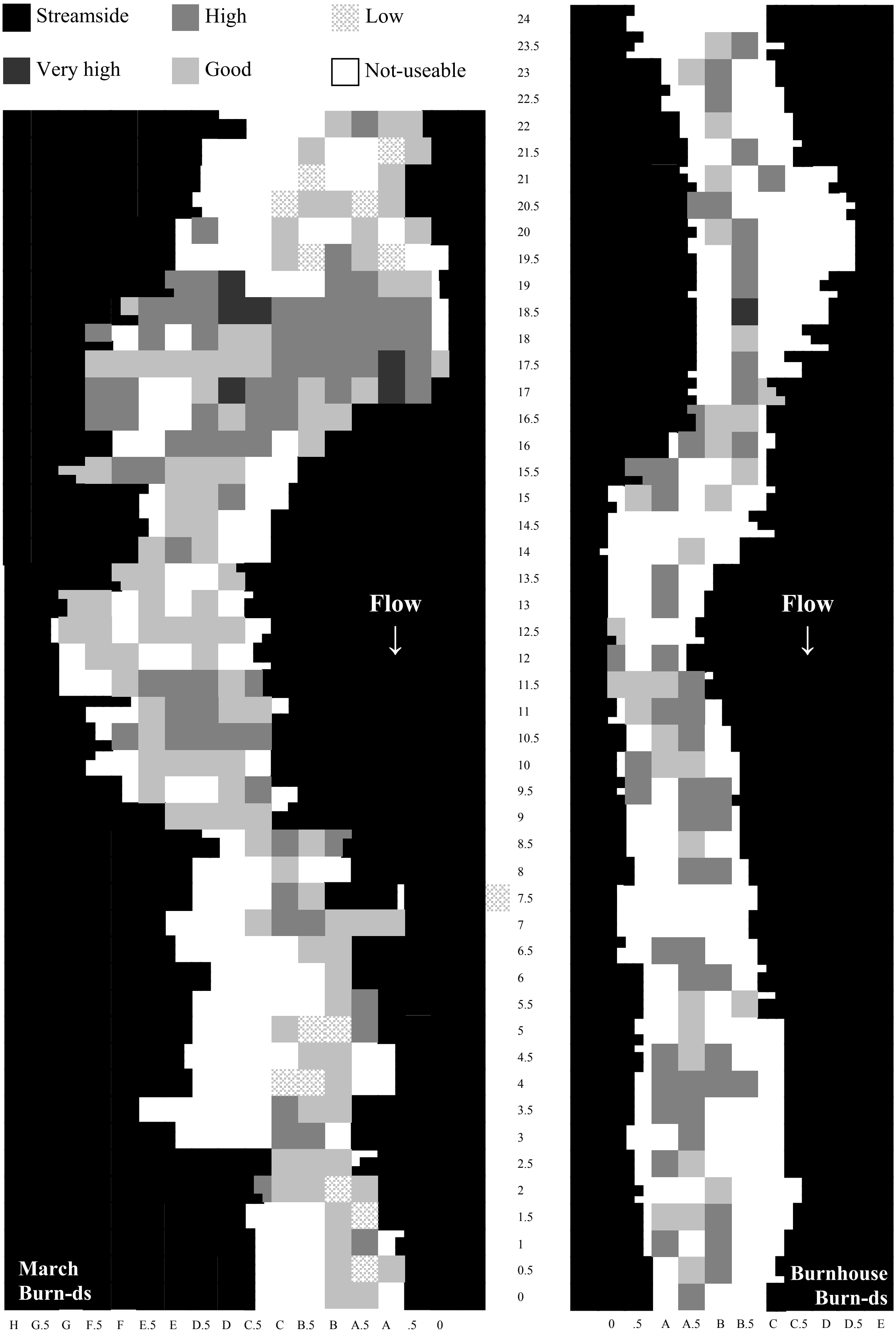


Figure 7.7: Habitat available for juvenile brown trout (*Salmo trutta*; length >7 to 20 cm) in March ($Q = 0.341\ m^3/sec$) and Burnhouse Burn ($Q = 0.0058\ m^3/sec$). This habitat assessment incorporates depth, velocity and substrate survey data.

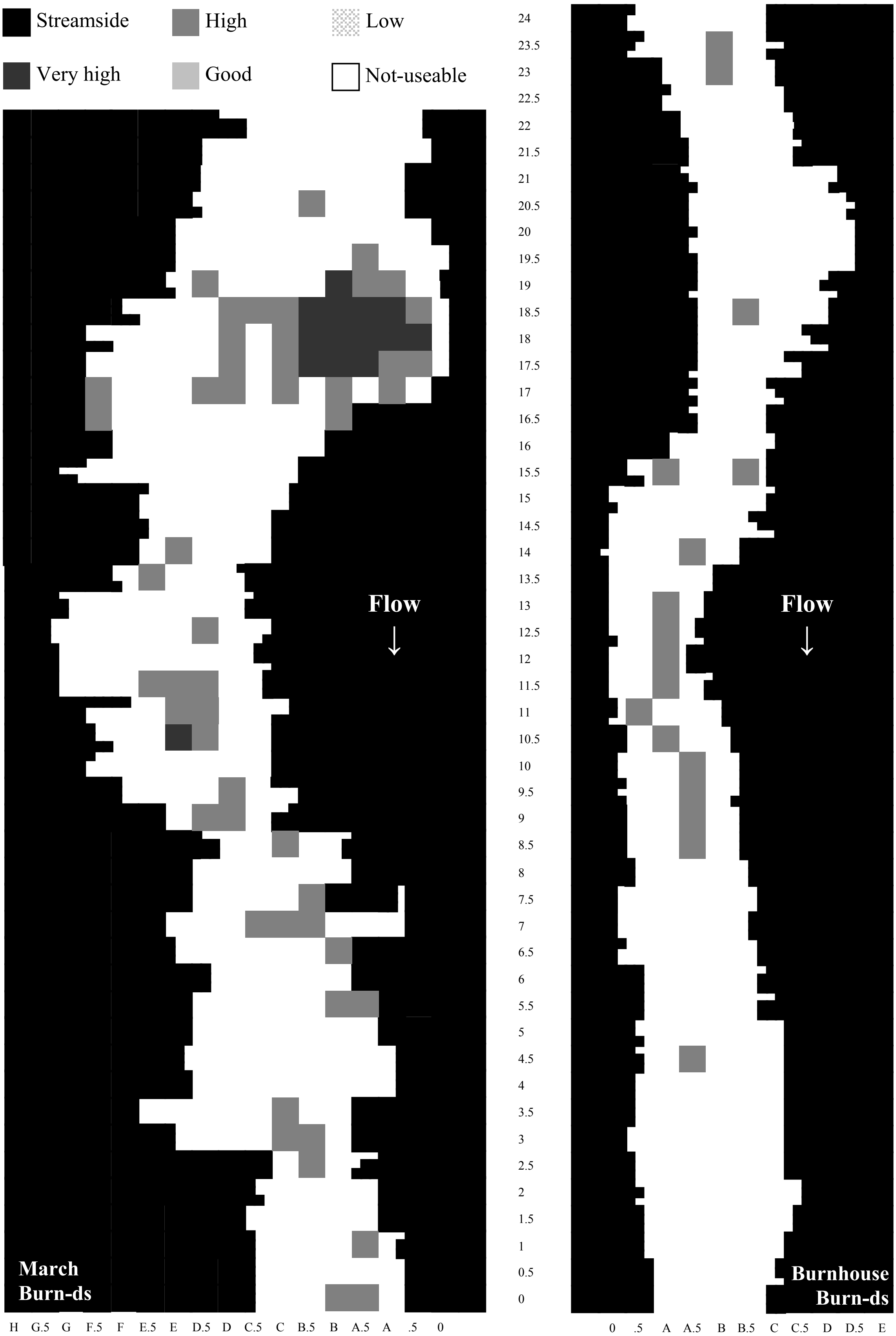


Figure 7.8: Habitat available for adult resident brown trout (*Salmo trutta*; length >20 cm) in March ($Q = 0.341$ m^3/sec) and Burnhouse Burn ($Q = 0.0058$ m^3/sec). This habitat assessment incorporates depth, velocity and substrate survey data.

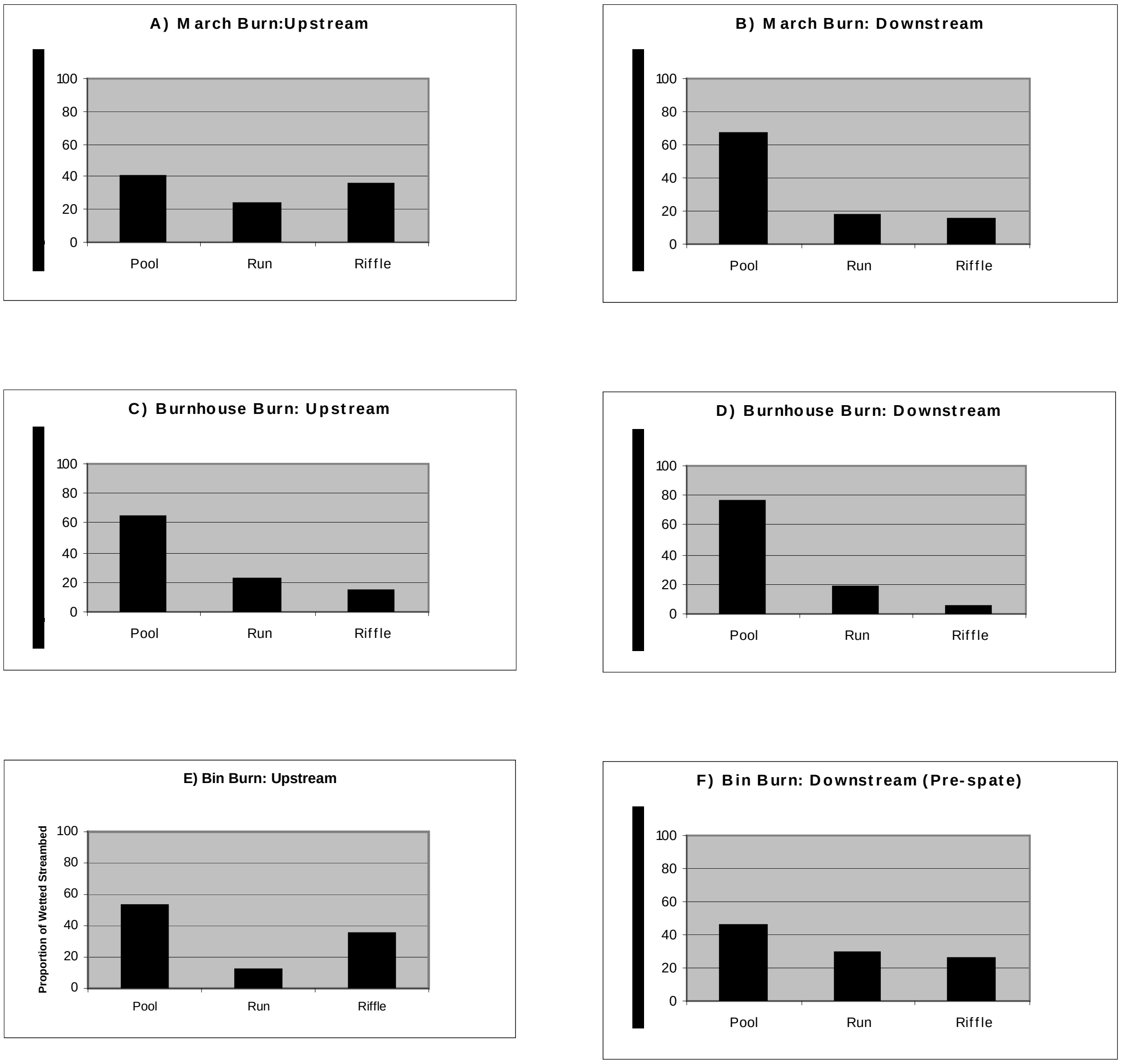
Burnhouse Burn, the physical habitat suited for the four life stages seems to be more evenly distributed throughout the reach.

7.3.3 Froude number

Table 7.6 and Figure 7.9 (A-G) outline the proportions of wetted streambed that were classified as pool, run, and riffle according to the criteria established for Froude number. In all instances the majority of the wetted surface was classified as pool habitat. The

Table 7.6: Proportion of pool, run, and riffle habitat in the wetted portions of each study reach based on Froude Number.

Site	Habitat type	Proportion (wetted) %
March Burn Upstream	Pool Run Riffle	40.3 24.2 35.5
March Burn Downstream	Pool Run Riffle	67.0 18.2 14.8
Burnhouse Burn Upstream	Pool Run Riffle	64.0 22.0 14.0
Burnhouse Burn Downstream	Pool Run Riffle	76.1 18.2 5.7
Bin Burn Upstream	Pool Run Riffle	53.0 11.9 35.1
Bin Burn Downstream (Pre-spate)	Pool Run Riffle	45.6 29.1 25.3
Bin Burn Downstream (Post-spate)	Pool Run Riffle	41.1 28.1 30.8



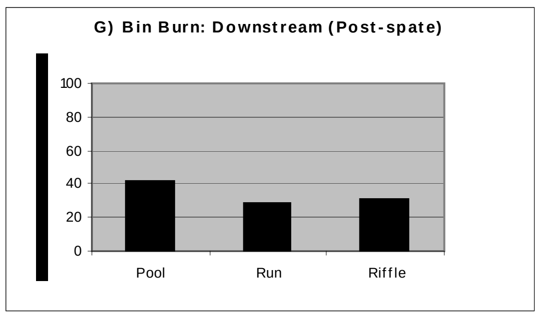


Figure 7.9 (A-G): Histograms of the proportion of pool, run, and riffle habitat in each study reach based on criteria developed for Froude number.

proportion of pool habitat ranged from 76.1% (Burnhouse Burn-downstream) to 40.3 % at March Burn-upstream. In five of the seven datasets riffle habitat was the least abundant. The proportion of riffle habitat ranged from 35.5% at March Burn-upstream to 5.7% at Burnhouse Burn-downstream. The proportion of the streams with intermediate Froude number (runs) ranged from 11.9% (Bin Burn-upstream) to 29.1% (Bin Burn-downstream (Pre-spate). The relative proportions of these three habitat classifications were examined using chi-squared analysis to test whether the relative proportion of these three habitat types differed between the data sets. The Chi-squared test indicated that the proportion of pools, runs, and riffles, are not independent of site (X^2 = 86.2, df = 12, p = 0.000). The relative proportions of habitat type observed pre- and post-spate at Bin Burn-downstream was also compared and the results indicate the relative proportions did not change after the disturbance (X^2 = 2.3, df = 2, p = 0.320).

Froude number can also be used to indicate sub critical and super-critical flow. The majority of quadrats measured returned Froude numbers less than 1.0 indicating that most of the wetted surface could be characterized as containing tranquil flow at the time of sampling. There are some exceptions, most notably the upstream Bin Burn site where just over 19% of the quadrats examined had Froude numbers greater than 1. The remaining sites ranged from no quadrats with Froude numbers greater than 1 (March Burn-upstream and both Burnhouse Burn sites). Bin Burn-downstream had 2.1% of the wetted surface with super-critical flow before the spate and 0.3% after. These results are outlined in Table 7.7.

7.3.4 Secondary analysis: correlation (Cramer's phi)

A re-examination of the three individual habitat utilization graphs (depth, velocity, and

Table 7.7 Proportion of wetted quadrats with super-critical flow (Froude number > 1).

Site	Froude Number >1 (%)
March Burn Upstream	0.0
March Bun Downstream	6.9
Burnhouse Burn Upstream	0.0
Burnhouse Burn Downstream	0.0
Bin Burn Upstream	19.4
Bin Burn Downstream (Pre-spate)	2.1
Bin Burn Downstream (Post-spate)	0.3

streambed substrate) that are used in the construction of the integrated habitat assessment maps reveals that the final distribution of useable habitat seems to be more strongly related to the distribution of one factor (e.g. depth) than the others (water velocity or streambed substrate). A good example of this is the availability of spawning habitat. The final distribution of useable spawning habitat in March Burn (Figure 7.5) more closely resembles the availability of streambed substrate (Figure 6.10) than either water depth (Figure 4.10) or velocity (Figure 5.12). This trend is less clear in Burnhouse Burn but it would appear that water depth or velocity may be more important to the final distribution in this stream.

In order to more objectively determine the habitat parameter that contributes most strongly to the integrated distribution and abundance of available physical habitat, correlation coefficients were calculated that compare the three habitat usability ratings (suitable, useable, not useable) with the five quality-designators (very high, high, good, low, not-useable) in the final distribution. This was done by calculating Cramer's phi coefficient, which is a variation on Pearson's correlation coefficient (r) and is designed to look for relationships between two qualitative variables (Witte 1993). Phi coefficients were calculated by comparing the final integrated habitat quality scores with those calculated for each of the three individual physical habitat assessments (water depth, velocity and streambed substrate) at all four life stages and in all six study reaches (seven datasets). The results are shown in Table 7.8. Values that approach 0 signify a weak relationship between variables and values that approach either positive or negative 1, indicate a strong positive or negative relationship. Calculations were done using the statistical software SYSTAT© version 10 (SPSS Inc. 2000).

A great deal of information is presented in Table 7.8; however, some trends can be discerned. To begin, the relationships between the individual and integrated habitat assessments based on the criteria developed for spawning brown trout is the least apparent. The strongest relationships were found in March Burn (both sites), Bin Burn-upstream, and Bin Burn-downstream (post-spate), all of which had correlation coefficients were greater than 0.7. However, the strongest relationships varied between the integrated assessment and water velocity [(March Burn-upstream & Bin Burn-downstream (post-spate)] or substrate (Bin Burn-upstream and March Burn-downstream). For all remaining comparison, at the time of sampling, the phi-coefficients were below 0.7 indicated moderate to weak relationships between the physical habitat available and the needs of spawning trout defined in this study. Further, the strongest associations between the individual and the integrated assessments vary from site to site indicating that the most important physical parameter.

Table 7.8: Cramers's phi coefficients resulting from the comparisons of suitability scores (suitable, useable, unsuitable) calculated for the individual assessments with the integrated quality score (very-high, high, good, low, not useable) in the total streambed. Comparisons were made for all three physical parameter (water depth and velocity, streambed substrate) for each life stage for all seven datasets.

		March Burn Upstream	March Burn Downstream	Burnhouse Burn - Upstream	Burnhouse Burn - Downstream	Bin Burn - Upstream	Bin Burn – Downstream (Pre-spate)	Bin Burn – Downstream (Post-spate)
Spawn	Depth	0.566	0.305	0.281	0.593	0.304	0.488	0.678
	Water Velocity	0.722	0.240	0.402	0.491	0.527	0.522	0.714
	Substrate	0.601	0.762	0.218	0.397	0.741	0.618	0.352
Nursery	Depth	0.918	0.700	1.000	0.968	0.587	0.916	0.995
	Water Velocity	1.000	0.957	0.984	0.864	0.850	0.845	0.825
	Substrate	-	0.832	0.574	0.529	0.721	0.637	0.295
Juvenile	Depth	0.924	0.817	1.000	0.748	0.856	0.800	0.813
	Water Velocity	0.964	0.970	0.955	0.936	0.935	0.819	0.835
	Substrate	-	0.571	0.569	0.506	0.666	0.665	0.253
Adult	Depth	0.984	0.955	1.000	1.000	0.957	0.904	1.000
	Water Velocity	0.423	0.305	0.262	0.356	0.456	0.504	0.371
	Substrate	-	-	-	-	-	-	-

The criteria developed for young trout used to evaluate nursery habitat demonstrates a much more consistent picture. With the exception of March Burn-downstream and Bin Burn-upstream, all the data sets showed strong relationships between both depth and water velocity, and the integrated assessment. In these streams, correlation coefficients ranged from 0.825 [Bin Burn-downstream (post-spate)] to1.0 (March Burn-upstream) for the comparison of water velocity and the integrated criteria; and between 0.916 [Bin Burn-downstream (pre-spate)] and 1.0 (Burnhouse Burn-upstream) for depth and the integrated assessment. In March Burn-downstream the coefficient of correlation for substrate, and the integrated analysis comparison was 0.832, although the strongest relationship was between the integrated analysis and water velocity (0.957). The strongest relationships in Bin Burn-upstream were also between water velocity, substrate, and the integrated analysis with phi coefficients of 0.721 and 0.850 respectively. The remaining relationships between substrate were much weaker ranging from 0.295 [Bin Burn-downstream (post-spate)] to 0.637 [Bin Burn-downstream (pre-spate)]. Thus, at the time of sampling, the streams examined in the Carron Valley were most dependent on depth and velocity for nursery habitat. March Burn-downstream and Bin Burn-upstream were an exception as there was a strong relationship between water velocity, substrate, and the integrated assessment.

A simpler picture can be drawn for the assessment based on criteria developed for juvenile trout. The largest phi coefficients were again found for the comparisons between depth and velocity, and the integrated habitat criteria. There were no exceptions in this instance as lowest phi coefficients were found between substrate and the integrated analysis in all seven datasets. The correlation coefficients for the depth-integrated comparison ranged from 0.748 (Burnhouse Burn-downstream) to 1.0

(Burnhouse Burn-upstream). The weakest relationship between water velocity and the integrated analysis was observed at Bin Burn-downstream (pre-spate) (0.819) and the strongest at March Burn-downstream (0.970). The phi-coefficients for the comparisons between substrate and the integrated analysis were all below 0.666 in the analysis of habitat for juvenile trout. No relationship could be established at March Burn-upstream in this instance as all the quadrats had the same value (constant).

The analysis of the relationship between the individual and integrated habitat assessments based on adult trout criteria was the most straightforward. In all instances the highest phi coefficients were found between stream depth and the integrated analysis. These were very strong relationships with values ranging from 0.904 to 1.0. The relationship between substrate surveyed and the adult trout criteria could not be calculated, as the substrate values were a constant (i.e. entire streambed was considered suitable for use by adult trout). Weak relationships were observed between water velocity and the integrated criteria ranging from 0.262 (Burnhouse Burn-upstream) to 0.504 (Bin Burn-downstream (pre-spate).

The comparisons between the habitat available based on individual and integrated assessments were conducted on the entire streambed. At the time of sampling the wetted surface covered only a portion of the total streambed (Table 4.8). It should not be surprising to learn that, at the time of sampling, these streams should demonstrate a strong relationship with physical parameters associated with the media that fish reside. This has significance from the perspective of a manager but trout using the streams are limited to the area of the stream that they can access. This analysis was re-run after eliminating the dry quadrats so comparisons could be made based on the assessments of the habitat actually available to trout. The results are outlined in Table 7.9.

Table 7.9: Cramers's phi coefficients resulting from the comparisons of suitability scores (suitable, useable, unsuitable) calculated for the individual assessments with the integrated quality score (very-high, high, good, low, not useable) in the wetted streambed. Comparisons were made for all three physical parameter (water depth and velocity, streambed substrate) for each life stage for all seven datasets.

		March Burn Upstream	March Burn Downstream	Burnhouse Burn - Upstream	Burnhouse Burn - Downstream	Bin Burn - Upstream	Bin Burn – Downstream (Pre-spate)	Bin Burn – Downstream (Post-spate)
Spawn	Depth	0.512	0.260	0.216	0.508	0.237	0.369	0.599
	Water Velocity	0.707	0.181	0.368	0.359	0.504	0.465	0.665
	Substrate	0.650	0.851	0.509	0.685	0.723	0.677	-
Nursery	Depth	-	0.470	1.000	0.803	0.315	0.421	0.984
	Water Velocity	1.000	0.903	1.000	0.743	0.825	0.794	0.531
	Substrate	-	0.988	0.977	0.888	0.863	0.713	0.508
Juvenile	Depth	0.934	0.726	1.000	0.632	0.771	0.703	0.814
	Water Velocity	0.964	0.858	-	0.709	0.776	0.590	0.749
	Substrate	-	0.666	0.970	0.852	0.777	0.746	0.441
Adult	Depth	0.983	0.958	1.000	1.000	0.990	0.904	0.063
	Water Velocity	0.370	0.145	-	1.000	0.362	0.410	0.092
	Substrate	-	-	-	-	-	-	-

With the exception of the spawning life stage, the relationship between the individual and integrated site assessments has become less consistent. Within the wetted portion of the stream the strongest relationship based on the criteria of spawning trout was between substrate and the integrated assessment in all but two of the datasets. The phi coefficients observed for the substrate-integration comparisons ranged from 0.509 (Burnhouse Burn-upstream) to 0.851 (March Burn-downstream). In both March Burn-upstream and Bin Burn-downstream (post-spate) the strongest relationship was between the integrated and water velocity assessments with ph coefficients of 0.707 and 0.665 respectively. This result is not surprising as spawning trout have very specific requirements for streambed substrate and need an adequate flow of water for both redd construction and egg survival (see Sections 5.2.1.1 and 6.2.1.1).

For young trout the availability of quality physical habitat seemed to be most strongly dependent on any of the three habitat parameters examined, depending on site. Sites where water depth had the strongest relationship with the integrated assessment included Burnhouse Burn-upstream (1.000) and Bin Burn-downstream (post-spate) (0.984). The integrated assessment was most strongly related to water velocity at the upstream sites of both March and Burnhouse Burns (both 1.000) and Bin Burn-downstream (pre-spate) (0.794). Substrate was the most important single physical habitat parameter for determining physical habitat quality within the remaining data sets. Similarly, the assessments based on criteria developed for juvenile trout resulted in variation between sites in the most influential single physical habitat parameter. The strongest relationship with the integrated assessment was with depth at Burnhouse Burn-upstream (1.000) and Bin Burn-downstream (post-spate) (0.814). Water velocity was most important at both March Burn sites (0.964 and 0.858 at the up- and downstream sites respectively). At the remaining sites the strongest relationship was between substrate and the integrated assessment and the phi coefficients that expressed these relationships ranged from 0.746 [Bin Burn-downstream (pre-spate)] to 0.852 (Burnhouse Burn-downstream). The substrate-integrated assessment comparison was the strongest for the juvenile criteria but only just. The phi-coefficients calculated at Bin Burn were very similar and ranged from 0.771 to 0.777 indicating the all three physical parameters examined seemed to influence the finals assessment equally.

The assessment of the relationship between the individual and integrated habitat assessments in the wetted portion of the stream, based on the criteria developed for adult trout, generated relationships which were very similar to those observed in the total streambed. Substrate was a constant; thus, no coefficient could be returned indicating that it did not influence the habitat quality within these reaches. In all data sets except one, where phi coefficients could be calculated, stream depth had the strongest relationship with the integrated assessment. The phi coefficnets for these datasets ranged from 0.904 [Bin Burn-downstream (pre-spate)] to 1.000 at the Burnhouse Burn sites. These strong relationships indicate that the habitat available for resident adult trout was most dependent on depth in the bulk of these streams. The phi-coefficients calculated at the post-spate downstream site in Bin Burn were close to 0 (0.063 and 0.092, depth and velocity, respectively) which suggests that neither of these physical habitat parameters had a strong relationship with the integrated assessment. Both, depth and velocity were strongly correlated with the integrated physical habitat estimated at the downstream site in Burnhouse Burn meaning both were important for resident trout and that there was important variability apparent in the assessment.

7.4 Final Site Characterization

7.4.1 March Burn-upstream

The upstream March Burn site was the smallest stream surveyed in terms of both total surface area and discharge. It was assessed to be best suited as a nursery for young fish or as habitat for juvenile residents. Just over 95% of the reach could be used by fry and juvenile trout and adult residents could use over half the quadrats surveyed. However, no fish were observed at this site. The habitat assessment was conducted at a low point in the hydrograph (see Figure A.1) and the correlation analysis suggested that water velocity then depth had the strongest relationship with the combined habitat assessment. It would seem this reach would improve, in terms of available habitat for adult trout at least, at higher points in the hydrograph. However, there were periods were the hydrograph dipped below the discharge recorded at the time of sampling. This reach had been canalized during past logging activities, which is implied by the relatively similar total and wetted surface areas. The analysis of Froude number suggests that a good portion of the reach could be considered pool habitat and there was no velocity-depth combination that was considered supercritical. Although the flow at the time of sampling was certainly tranquil, the canalized nature of the reach resulted in little in the way of shelter from either very high or very low flows. The lack of deep pool refugia during the low flow periods may limit the utility of this site as a permanent residence for any age class of fish, especially larger and older specimens. Further, the site was upstream of a number of small waterfalls and the gauging weir, obstructions that might be limiting access to recruitment either through dispersal of fry from other sites or directly by limiting access by spawning adults.

7.4.2 March Burn-downstream

There were fish observed at the downstream March Burn site. However, unlike the other downstream sites there were no spawning adults observed in this reach during the early December (2002) survey. There were adult fish observed downstream of the old gauging station indicating that this barrier could prevent the upward migration of fish. There were smaller residents observed in the study reach, which would suggest that at some point spawners were able to bypass this obstruction. This site had one of the larger discharges and streambed areas of the reaches studies. This site also had a large pool; a feature not replicated at the other study reaches. The assessment suggests that the site was best suited for young and juvenile fish but the presence of the pool could provide habitat for resident adults. At the time of the assessment 19% of the reach (15.3m^2) was considered useable for adult fish, which was the second largest area of the reaches studies. Although spawners were blocked and this reproductive activity was not observed in this reach the assessment did suggest that spawning habitat was available. The calculation of Froude number indicates that pools dominated this reach but there were some quadrats that exhibited super-critical flow ($\approx$ 7%). The survey was conducted near the low point of the hydrograph (Figure A.2); however, there were periods with lesser discharge. The pool habitat available could provide refugia in this instance. The quadrats of super-critical flow are indicative of riffle habitat; the diverse substrate may provide refugia for young trout during high period of flow. This reach does provide some of the most and best habitat observed in this study and could benefit from a removal of the downstream obstruction.

7.4.3 Burnhouse Burn-upstream

No fish were observed in the upstream reach studied on Burnhouse Burn. This site was one of the smaller stream segments examined in terms of both discharge and useable surface area available for all age classes. The assessment conducted indicated that this reach was best suited as nursery and juvenile habitat. Only a small portion of the streambed surveyed could be used by adult resident trout (1.3 m^2) and a similar portion (1.2 m^2) was suitable for spawning. This portion, about 4%, was typical of that observed in other studies (Hewitt and Newcomb 2000, Rubin et al. 2004); however, the total area available was still quite small. The survey at this site was conducted upstream of a logging road. The stream passed under this road through a culvert that was clogged with debris posing a significant barrier for the migration of fish. Further, natural waterfalls also occurred downstream of the study site, which may have provided a barrier under some discharge regimes. The strongest correlations with the integrated assessment were with depth in the resident life stages indicating that there was insufficient water to support many fish, if any. Again the survey was conducted at a low point in the hydrograph (Figure A.3) and it seems during these periods fish populations would have trouble maintaining residence, even with the absence of physical barriers to migration. The Froude number assessment indicated that much of the streambed could be considered pool habitat but this designation would have more to do with low water velocity and less to do with depth. There were no quadrats at the time of sampling with super-critical flow.

7.4.4 Burnhouse Burn-downstream

Both resident and spawning trout were observed at the downstream site at Burnhouse Burn. This site was of intermediate size relative to the streams examined but is the smallest of the downstream sites. The streambed was 58.8m^2, the discharge was 0.0058 m^3/sec (Table A.1 and Figure A.4), and the site was assessed to be most suited as habitat for young and juvenile trout. At the time of sampling approximately 8.5 m^2 were considered useable for spawning and spawners were observed at this site in early December (2002). There was an access road that crossed Burnhouse downstream of this site that was undercut by a pair of culverts. Clearly these culverts were navigable by trout returning to spawn in this stream. The Froude numbers indicate that the bulk of the habitat available was considered pools (>76%), the remaining being considered either run or riffle habitat. There was no super-critical flow observed at the time of sampling. In the wetted portion of the stream spawners were largely limited by the availability of adequate substrate, as were intermediate age classes, and adult residents by depth. Much of the substrate at this site was bedrock providing limited utility for resident age classes. Although the Froude numbers indicate that much of the physical habitat available to fish was in the form of pools many of these would be relatively shallow and of utility to only the smaller age classes. The site was surveyed at a point relatively low in the hydrograph and small resident fish were observed throughout the year. Some habitat was available for larger fish at the time of sampling (≈ 7 % of the total streambed) and more would become available during peaks in the hydrograph but continual resident of older age classes would be expected to be restricted by a lack of water depth during low flow periods.

7.4.5 Bin Burn-upstream

Bin Burn was the largest of the upstream sites and was comparable in size to the downstream Burnhouse Burn site (total streambed 58.8 m^2, discharge 0.0294 m^3/sec). Based on the assessment conducted, this site had adequate physical habitat suited for spawning and was considered best suited for nursery and juvenile trout. However, no fish were found at this site. There does seem to be adequate physical habitat available. Burnhouse Burn-downstream has a similar amount of streambed considered suitable for resident age classes (Table 7.4), supports a resident population of young trout, and spawners have been observed utilizing this site. About half of the area surveyed was considered pool habitat based on the Froude numbers calculated for the wetted quadrats; the remaining riffle, with a smaller portion assessed to be run habitat. The highest level of supercritical flow was observed at this site (≈ 20%), much of which was found in a waterfall at the upstream portion of this reach. The survey was conducted at an intermediate time in the hydrograph (Figure A.5) so both higher and lower discharges would be expected. The individual physical parameter more strongly correlated with integrated assessment would be expected to change at different points in the hydrograph but there is nothing to suggest that this site could not support a resident population of young and juvenile fish and be used as a spawning area. The reason fish are not observed is probably a result of a large natural waterfall (several meters in height) and a culvert under a logging road downstream of this reach. The culvert may restrict some fish from reaching this site, particularly if it becomes clogged, however these features can be navigated as observed at the downstream Burnhouse Burn site. The large waterfall would pose a much more difficult obstacle for migrating fish and is most likely responsible for the absence of fish.

7.4.6 Bin Burn-downstream (Pre-spate)

Bin Burn was affected by a large spate that washed logs and other debris that became lodged at a road crossing just downstream of the study site. This material was removed and the streambed reconstructed. This spate and reconstruction provided an opportunity to examine changes in the physical habitat before and after this event. Fish were observed at the downstream site; however, survey were taken after the disturbance. Before the spate, the assessment suggests that this reach was best suited for young and juvenile trout. There was adequate physical habitat available for returning spawners. The total surface area considered suitable was the largest surveyed for all age classes with the exception of the post spate assessment. Based on Froude numbers, roughly half the wetted streambed was considered pool habitat, the remaining approximately equal portions of run and riffle. Only 2.1% of the wetted quadrats surveyed were considered to have super-critical flows. The fish survey was conducted after the spate; however, young trout were observed while working in the reach prior to the disturbance. There is nothing to suggest that young and juvenile trout did not used this site as a residence, and that spawning did occur.

7.4.7 Bin Burn-downstream (Post-spate)

Trout were observed at this site after the spate. The first survey after the spate reported diminished numbers compared to Burnhouse Burn (Table 4.10 A-C), particularly considering the area of useable streambed was larger. However, the severity of the disturbance, which was less than two-month prior to the survey, suggests that fish density would be affected. Over time, the numbers of fish did increase and were roughly similar to those seen at the downstream Bin Burn site. The total streambed increased substantially, becoming more than twice as large. The wetted portion of the reach grew by about 65% even though the discharge at the time of sampling before and after the spate was similar (Figure A.6). The reconstruction created a much wider streambed resulting in shallower overall depths. With the exception of the adult assessment the total area considered useable by all age classes increased after the spate.

The correlations conducted in between the assessment of individual and integrated habitat parameters resulted in a shift from strong correlations with substrate (spawning and juvenile assessments) before the disturbance to the strong correlations with depth and water velocity after the event. The Froude number calculated for the wetted quadrats resulted in the lowest portion of the wetted streambed considered pool of all datasets, with the exception of March Burn-upstream. Only 0.3% of the wetted quadrats displayed flow that would be considered super-critical. The spate and reconstruction seemed to have resulted in a wider shallower streambed but did improve the availability of physical habitat based on the assessment used in this study. The survey was conducted at a relatively low point in the hydrograph so much more physical habitat could become available at different points in the year. However as depth had the strongest relationship with the integrated assessment, low flow periods could make larger fish vulnerable, especially considering that the amount of pool habitat seems to have been reduced.

7.5 Discussion

The tolerance profiles that were created in this study are standardized assessment criteria or 'reference conditions' that when compared with stream survey data can produce an appraisal of habitat availability in any fluvial freshwater system that supports populations of brown trout (*Salmo trutta*). This was demonstrated by assessing the habitat availability at six sites in the Carron Valley, Scotland, in the late summer of 2002. The habitat data collected in a survey of these streams combined with the tolerance profiles have been applied to the reaches examined in this study and have generated summaries for the availability of water depth and velocity, and streambed substrate for the four life stages of brown trout. These analyses have been combined to produce an integrated habitat assessment, which is a realistic approach as salmonids choose their microhabitat based on multiple factors (Baldes and Vincent 1969). This approach allows an investigator to determine the amount and relative portion of useable physical habitat and to determine the quality of that habitat. Finally, by examining the habitat variable that most strongly correlates with the final integrated habitat distribution the physical habitat parameter that is most important to the distribution of trout can be determined.

Although, on a cursory inspection, these streams are similar in size and are located in the Carron Valley the habitat available differs. They differ in the total area that can be used, the proportion of the streambed that is wetted or inaccessible, as well as the quality of the available habitat. This is not unexpected as it is unlikely that any two streams would appear structurally identical (Cunjak 1996). Despite these structural differences all stream segments streams seem best suited as nursery and spawning areas. To a lesser extent juvenile trout can use these burns. This functional assessment was supported at the Burnhouse and Bin Burn downstream sites as the bulk of the fish present were fry and small juvenile and spawning runs were observed in early December 2002. Less support came from March Burn-downstream and the upstream sites because of the old gauging station, culverts, and natural waterfalls that may have prevented the up- and downstream movement of trout. However, it does illustrate one of the strengths of this assessment method in that evaluation can be conducted in the absence of the target species.

The chi-square tests used to determine differences in the suitability criteria most often found differences between the study reaches; however, as noted above, the trends in stream suitability were very similar. This could be explained by the large amount of data that was used in this analysis, which would increase the likelihood of finding statistical differences. These tests should be used in the accompaniment of graphical data so that trends are not misrepresented.

Another attribute that can be determined using this method is the single physical habitat parameter that may contribute strongest to the final assessment of habitat availability. The literature review that was conducted in earlier chapters has shown that any of the three habitat parameters examined have been thought to be the most important factor in the distribution of trout distribution. For example depth (Bohlin 1977),velocity (Bachman 1984), streambed substrate (Gatz et al. 1987) or combinations of these three (Karlstrom 1977) have been reported as limiting trout abundance. This study has demonstrated that within an individual stream the availability of useable habitat can be more strongly associated with one of the physical habitat parameters and this varies at different life stages. As well, differences have been observed in the relationship between single and combined assessments when making stream-to-stream comparison of these burns in both life-stage and perspective (wetted areas or entire streambed).

The physical habitat variable that is correlated strongest with the integrated distribution of trout microhabitat may also be the variable most sensitive to disturbance. Put another way, disturbance or change within the stream or the surrounding watershed that change the availability of ecosystem resources may have strong impacts on the biological communities that depend on these resources. This was clearly demonstrated by Schindler and colleagues (1971) in their trials with inorganic nutrients in the experimental lakes studies. By identifying the habitat variable most strongly associated with the integrated assessment we can determine the types of environmental change that may have the strongest ability to influence ecosystem structure and function. For example, at the time of sampling the streambeds at the downstream sites had large portions of dry substrate. The correlation analysis indicated depth and velocity (discharge) were most strongly correlated with the availability of habitat for resident trout. Changes in the hydraulic conditions within these streams are more likely to influence the distribution of trout then are changes in substrate composition.

Changes in the hydraulic regime, especially in these systems, are not unusual. The hydraulic summary in the Appendix illustrates the degree to which the discharge varies in these systems. The analysis of these burns was conducted at one time during the year and multiple measures throughout the year, at least once per season (winter, spring, summer, autumn) will more accurately represent the habitat available to trout in this system. This is particularly true when asking questions about specific life stages of trout. For example an analysis of spawning habitat should be conducted in autumn and nursery habitat should be examined in spring and summer when the target life stage of the species of interest is using the stream resources. In the late summer, it seems these streams are best suited for nursery and juvenile trout. These sites were also suited as spawning habitat. However, as the survey was conducted when fish were not spawning (too early in the season); thus, the utility of these streams for spawning cannot be assessed with any certainty unless a survey is conducted when fish are observed spawning (are at similar discharges). The empirical methods used in this survey were cumbersome and time consuming making multiple measures impractical. The incorporation of the suitability criteria into hydraulic models would aid assessment of physical habitat in these streams.

The three physical habitat parameters (depth, velocity and substrate) were integrated using a simple index methodology. Depth and velocity information was also integrated using Froude number. Froude number can define sub- and super-critical flows for salmonids (Tetzlaff et al. 2005b). This calculation does not incorporate substrate or other physical habitat variables that may influence microhabitat selection but it does represent a functional response, which is useful in dynamic systems such as river ecosystems. Habitat assessment would benefit for the use of measure such as Froude number particularly if they could be developed for the specific life stages of the target species or incorporate a fish length into the calculation.

A limitation with the development of this methodology at this stage is validation. The fish survey data have demonstrated that the age classes seen in the study reach (at least in Burnhouse Burn) are those that would be expected based on the habitat analysis conducted from the tolerance profiles. However, the habitat availability was conducted on a microhabitat scale (quadrat) while the fish survey was conducted on a mesohabitat scale (study reach). Stronger evidence to support the accuracy of the assessment methodology would come from comparisons of microhabitat usage of resident trout usage (Heggenes 2002) with the type of microhabitat assessments already conducted.

Finally minor improvements or adjustments can be made to improve this methodology. As mentioned in previous chapters a broader range of microhabitat habitat usage studies should be carried out. The bulk of the investigations that have been used to construct the tolerance profiles come from northern Europe, North American and New Zealand. More studies from southern and eastern Europe, eastern Asia and some of the isolated populations such as central Africa could provide more confidence in the tolerance profiles (see comments in Chapter 4). Secondly, microhabitat stream assessment may be useful for assessing areas of limited size however broad scale watershed or landscape scale surveys would be too time consuming and expensive for many agencies responsible for ecosystem stewardship. Alternatives, such as integrateion with mathematical models, should be explored in order to streamline this technique. Other possibilities includes linking quadrate size to actual microhabitat size used by resident fish with the hope of using larger and fewer quadrat sizes with increases in size (age class) of fish. As well, the development of a mesohabitat scale (reach or stream segments) assessment might reduce the workload associated with the current protocol. An extension of the methods outline in this study that uses mesohabitat scale measurements is outlined in the Appendix.

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8.0 General Discussion

8.1 General Overview

The original objectives were to build tolerance profiles for the physical habitat parameters water depth, velocity and substrate based on observations of brown trout in their natural environment. Secondly, to compare measures of these three parameters in the study stream with the tolerance profile to access the availability of each parameter in the study reaches. Thirdly, to integrate the three study parameters in a combined assessment of the physical habitat available in the study reaches for the four life stages of *S. trutta*. The strengths and shortcomings in the development and application of the tolerance profiles are discussed in the individual chapters covering each parameter. In general, however, I was able to achieve these goals. The tolerance profiles were constructed for each of the three physical habitat parameters from a survey of the literature, these criteria were then compared with the survey data and assessment of the habitat available was conducted. This data was presented as table, histograms and, in some instances, habitat maps. Finally, the three habitat parameters were integrated into a combined assessment using an index and the calculation of Froude numbers to assess the amount and quality of the habitat available to brown trout. Additionally, I was able to compare the distribution of useable habitat of the individual habitat parameters with the combined distribution, using correlation analysis, to determine which of the three physical habitat parameters is most responsible for the availability of physical habitat based on the integrated assessment.

8.2 Strengths of the Assessment Protocol

There are a number of strengths of this method. To begin, once the tolerance profiles are constructed they can be used universally. What that means is that the criteria (tolerance profile) used to assess habitat availability can be used in any river in which brown trout are currently resident or have resided in the past. The suitability criteria developed can be used in hydraulic models such as IFIM (Bovee 1982, Milhous et al. 1984) in instances where transferability of site specific habitat suitability index is not assured. Secondly, the assessment criteria remain compartmentalized, which allows managers to identify specific problem areas. Further, the correlation analysis has allowed for the identification of the limiting habitat feature, which can have implications in management decisions. As well, the sampling methodologies (i.e. measures of depth, current velocity, and substrate) are simple, straightforward and employ basic and commonly used in equipment. Thus, they can easily be adopted and understood by field workers and can easily be standardized so that comparable data can be accumulated.

The choice of brown trout, a ubiquitous species in Europe, allows for pan-European assessment criteria that can be standardized and comparable anywhere that trout reside. Although trout cannot be expected in all surface water in Europe the methodology can be easily adapted to other ubiquitous and well known species such as Atlantic salmon (*Salmo trutta*), charr (*Salvelinus alpinus*), or eels (*Anguilla anguilla*). As well, by incorporating the stages of the life history of brown trout into the method many of the surface water types can be adopted into the model as trout use many types of surface waters throughout their life. The tolerance profiles are in essence reference conditions that can be utilized in any brown trout waters in Europe (again, transferable) and the grades incorporated into the tolerance profile (usable, suitable, not useable) can be adopted to classify waters using quality qualifiers such high, good, low, or poor.

8.3 Limitations of the Model

This monitoring protocol has limitations as well. Some of these have been mentioned in previous chapters. Briefly, the studies used to construct the tolerance profiles often came from studies in northwestern Europe and parts of North America. The confidence in the suitability ranges defined in these studies could be achieved by further research in the understudied portions of the species range. The grid system used to measure the three physical habitat parameters in the study streams was time consuming and is practical only in areas where microhabitat scale environmental assessment is warranted. As well, the boundaries in the tolerance profiles that distinguish between suitable, useable, and unsuitable habitat were constructed using largely subjective techniques. This is in part a result of the nature of the literature available for their construction as the data employed were developed and reported for the purposed of the original study. Thus, the data was not reported in a manner that was complete or in a fashion that would allow for statistical comparisons. Further research using standardized sampling and reporting techniques could improve the confidence in boundaries defined and would allow these boundaries to be defined in an objective or standardized manner.

The incorporation of as broad a survey of microhabitat choice in the creation of the suitability is useful in that it helps insure the criteria can be broadly applied. However, an individual organisms choice of microhabitat is influenced by numerous factors such as inter-cohort competition (Bohlin 1977), inter- and intraspecific competition (Heggenes et al. 1999) and parameters such as temperature (CDV: Section 5.4). The interplay of these influences will result in narrower bands of habitat actually utilized by individual organism. Thus, boundaries between suitable, useable and not-useable will be site-specific and change as conditions change (such as drops in temperature). Unfortunately, the static nature of the suitability criteria does not allow them to reflect changes in habitat suitability as environmental condition change.

There are a number of other limitations that can be grouped into two broad categories. These are: development or validation. Issues related to development are concerned with the early stage of development the methodology and the limitations that this imposes. Chiefly, the habitat requirements of only one target species could be developed for this project. Clearly, other fish species and other non-fish species reside in these systems. A more holistic approach to ecosystem management (as outlined by the WFD) suggests that the habitat requirements of multiple species (e.g. invertebrates and macrophytes) also be considered. Thus, if this approach is to be utilized generalized habitat suitability criteria will need to be developed for a broader range of species. As well, the model in its current state has been developed strictly for application in rivers. Assessment criteria for other surface water such as lakes, reservoirs, estuaries and coastal areas need to be developed in order to meet the full requirements of the holistic oriented environmental protection legislation such as the Water Framework Directive.

An assumption that is implicit in the model is that rivers under investigation have either had populations of trout or current support population of trout. If the habitat is based on the supposition that trout had lived in a given water body we need to make sure that this was actually the case. There are methods that can be used to determine species that may have been present in waters in the past through catch records, photographs, and oral accounts of elders. If this information is not available we may be assessing a watercourse based on criteria that never occurred. There is no easy solution. It may be possible to create general habitat criteria based on the most common requirements of resident species found in a particular type of surface water within a given region and apply these criteria to streams that lack information regarding former residents. Again, the model will need to be developed to include more species if this is to be accomplished.

Another area of limitation for this model had to do with validation. As touched on in the last chapter, I was able to confirm that the assessment was accurate in that the fish expected to be present based on the criteria used in the method were actually found in the study streams. However, this comparison was made between habitat analysis that was done on a microhabitat scale (quadrats) and a fish survey that was conducted on a mesohabitat scale (study reach). A more convincing argument could be made if comparisons between habitats that fish are expected to use and observation of the presence or absence of fish in those habitats could be done on a quadrat-by-quadrat basis. Further, habitat was available at sites where few or no fish were found. It was assumed that these fish were not present because of natural and man-made obstructions. This assumption could be easily tested, at least in the case of the March Burn-downstream site, by the removal of the old gauging station. This site was assessed to be available for nursery, juvenile and spawning fish. Spawners were not observed. The return of spawning fish after the removal of the obstruction would help validate the assessment method. Finally, as previously discussed in Chapter 7 (Section 7.5), the assessment method needs to be conducted at a variety of discharges over the course of the year in order to more clearly determine habitat suitability for the life stages in the river at the critical periods of use. Thus, the incorporation of the suitability criteria into a hydraulic model would be desirable.

8.4 Future Studies

The first area that needs to be developed is to complete the measure of the physical habitat. As mentioned earlier, the distribution of salmonids is most often determined by four physical habitat parameters including water depth, current velocity, streambed substrate and cover (Heggenes 1988). Aspects of cover are integrated into measures of depth (overhead cover) and streambed substrate (horizontal cover). However, much of the forms of cover provided by undercut banks, surface turbulence, overhanging vegetation (Heggenes et al. 1999) is not included in the current measures. Physical habitat assessment criteria will only be complete when the tolerance profiles for *S. trutta*'s use of cover are constructed and incorporated into the habitat assessment model. Secondly, the criteria should be integrated into a hydraulic model so that the physical habitat can be assessed without the time and expense associated with the empirical method used. This would provide more insights into how habitat availability changes with changes in discharge and could aid in validation of the suitability criteria. Further, assessments at multiple points in the hydrograph should be coupled with fish survey. These fish surveys should be conducted in which microhabitat usage based on instream observations is compared with the expectation of occupancy based on the habitat criteria developed from the tolerance profiles should be conducted to confirm that the model is accurately assessing habitat availability. Again, these validation studies should be conducted on a microhabitat (quadrat) scale. As well, the fieldwork conducted in this study was limited to a nursery stream. Examining riverine habitats that are likely to support larger juveniles and adult trout could further test and strengthen the model.

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