EFFECTS OF ROAD-STREAM CROSSINGS ON STREAM GEOMORPHOLOGY AND THE MOVEMENT OF SMALL BODIED FISHES IN THE ETOWAH RIVER BASIN, U.S.A.

by

JAMES RANDOLPH NORMAN

(Under the Direction of Mary C. Freeman)

ABSTRACT

I evaluated the efficacy of the U.S. Fish and Wildlife Service's Reasonable and Prudent Measures (RPMs) in protecting federally listed fish species by minimizing effects of road-stream crossings on channel geomorphology and fish movement. Measurements at 11 crossings constructed under RPM terms and conditions showed that RPMs were relatively effective in preventing changes to channel geomorphology with respect to channel depth and substrate composition, but less effective in preventing channel width changes and blocked fish passage. In a separate study, mark-recapture studies at four road crossings showed varying degrees of movement by small-bodied fishes (8 to 21 species per site) through culverts that differed in elevation above the streambed. Although none of the culverts were complete barriers to passage, only a bottomless-box culvert appeared to permit unrestricted, upstream and downstream movements by benthic and water-column fishes, based on multi-state model estimates of movement probabilities.

INDEX WORDS: Road-stream crossings, culvert, geomorphology, fish passage, fish movement, mark-recapture, multi-state models

EFFECTS OF ROAD-STREAM CROSSINGS ON STREAM GEOMORPHOLOGY AND THE MOVEMENT OF SMALL BODIED FISHES IN THE ETOWAH RIVER BASIN, USA

by

JAMES RANDOLPH NORMAN

B.S., University of Georgia, 2004

A Thesis Submitted to the Graduate Faculty of The University of Georgia in Partial Fulfillment of the Requirements for the Degree

MASTER OF SCIENCE

ATHENS, GEORGIA

2006

© 2006

James Randolph Norman

All Rights Reserved

EFFECTS OF ROAD-STREAM CROSSINGS ON STREAM GEOMORPHOLOGY AND THE MOVEMENT OF SMALL BODIED FISHES IN THE ETOWAH RIVER BASIN, USA

by

JAMES RANDOLPH NORMAN

Major Professor: Mary C. Freeman

Committee: Judith L. Meyer
David S. Leigh

Electronic Version Approved:

Maureen Grasso
Dean of the Graduate School
The University of Georgia
December 2006

ACKNOWLEDGEMENTS

This work involved the efforts and support of many people, whom without I would not have been able to complete. First, I would like to thank my mother for always supporting me in my academic pursuits and helping instill my wonder for the natural world at an early age; I would not be where I am today without her love and unwavering support. I would like to thank Judy Meyer, Gene Helfman, Bruce Wallace and his past students Heidi Wilcox, Wyatt Cross, Erica Chiao and Sally Entrekin, all of whom inspired my fascination with the aquatic world while helping me develop the intellectual foundation to go out and learn more about it. Mary Freeman deserves tremendous credit and thanks for being a wonderful, engaging teacher and mentor – she has been an incredible advisor. Megan Hagler provided invaluable help throughout the entire research process, and I could not have completed this thesis without her assistance. I want to thank all of the Freeman lab students and techs for their assistance and friendship – you all made my work a lot of fun. And thank you to all my friends, especially Rebekah Chapman and my girlfriend Suji Valia, for helping me get through to the end, and making my time at the Institute such an enjoyable experience. Finally, thanks to my committee members, Judy Meyer and David Leigh, for their thoughtful comments and suggestions that helped improve this thesis.

TABLE OF CONTENTS

Page

ACKNOWLEDGEMENTS iv

LIST OF TABLES vi

LIST OF FIGURES viii

CHAPTER

1 INTRODUCTION 1

2 ASSESSMENT OF THE EFFICACY OF U.S. FISH AND WILDLIFE SERVICE REASONABLE AND PRUDENT MEASURES IN PREVENTING GEOMORPHIC ALTERATION AND BLOCKED FISH PASSAGE AT ROAD-STREAM CCROSSINGS 8

3 APPLICATION OF A MULTI-STATE MODEL TO ESTIMATE CULVERT EFFECTS ON THE MOVEMENT OF SMALL STREAM FISHES 47

4 CONCLUSIONS 104

APPENDICES 107

A SUPPLEMENTAL MATERIAL FOR CHAPTER 2 108

B SUPPLEMENTAL MATERIAL FOR CHAPTER 3 141

LIST OF TABLES

Page

Table 2.1: Road-stream crossing description, culvert diameter for pipe culverts and width for box culverts, watershed size (km^2) and 2001 percent urban land use for study sites 30

Table 2.2: Mean basal channel width, maximum culvert openness, relative culvert openness, and the depth and percent of culvert diameter or height embedded at each site 31

Table 2.3: Upstream and downstream culvert impact measurements for basal channel width, thalweg depth and phi, with results of paired t-tests for each variable.....32

Table 3.1: Comparison of physical and water quality characteristics of mark-recapture study sites 81

Table 3.2: Description of the road-stream crossing and culvert characteristics for study sites.....82

Table 3.3: Marking scheme for all fishes except *C. oligolepis* and *H. etowanum*.....83

Table 3.4: Marking scheme for *C. oligolepis* and *H. etowanum* 84

Table 3.5: Description of site-specific candidate models and model selection procedure 85

Table 3.6: Description of combined-site candidate models and model selection procedure.....86

Table 3.7: List of benthic and water column fishes species caught at mark-recapture study sites 87

Table 3.8: The overall recapture rate and the number of fish captured 1, 2, 3 and 4 times at each study site 88

Table 3.9: The total number and percentage of moving and non-moving fish, and the total number of fish moving upstream and downstream, at each site and the sites combined (excluding *C. oligolepis* and *H. etowanum*).....89

Table 3.10: Summary of fish movements over the course of the study for each site 90

Table 3.11: AICc values and number of estimated parameters for site-specific candidate models 91

Table 3.12: Parameter estimates for survival and the range of parameter estimates for reach-specific recapture probabilities for benthic and water column fishes from the best-supported model for each site 92

Table 3.13: Transition probabilities from RMCS model for benthic fishes and water column fishes at Canton Creek 93

Table 3.14: Transition probabilities for benthic fishes and water column fishes at Bluff Creek (FMCS model), Champion Creek (FMCS model) and Nimblewill Creek (FM model) 94

Table 3.15: Culvert effect size on transition probabilities for benthic and water column fishes ..95

Table 3.16: Change in stage height, *effective min PAWS*, and the number fish that had moved upstream or downstream through a culvert after each time interval between sampling occasions 96

Table 3.17: AICc values and number of estimated parameters for combined-site candidate models 97

LIST OF FIGURES

Page

Figure 2.1: Map of the Etowah River Basin 33

Figure 2.2: Map of RPM road-stream crossing study sites.....34

Figure 2.3: Examples of multiple-barrel pipe and multiple-barrel box culverts from study sites 35

Figure 2.4: Longitudinal plot of basal channel width (raw data and running averages of 4 transects) at Site 1.....36

Figure 2.5: Longitudinal plot of basal channel width (raw data and running averages of 4 transects) at Site 2.....37

Figure 2.6: Longitudinal plot of basal channel width (raw data and running averages of 4 transects) at Site 9.....38

Figure 2.7: Longitudinal plot of basal channel width (raw data and running averages of 4 transects) at Site 10.....39

Figure 2.8: Longitudinal plot of basal channel width (raw data and running averages of 4 transects) at Site 11.....40

Figure 2.9: Longitudinal plot of basal channel width (raw data and running averages of 4 transects) at Site 6.....41

Figure 2.10: Longitudinal plot of thalweg depth (raw data and running averages of 4 transects) at Site 2.....42

Figure 2.11: Longitudinal plot of thalweg depth (raw data and running averages of 4 transects) at Site 10 43

Figure 2.12: Longitudinal plot of phi values (raw data and running averages of 4 transects) at Site 9 44

Figure 2.13: Delta values for upstream basal channel width, downstream basal channel width, thalweg depth, and phi versus relative culvert openness 45

Figure 2.14: Delta values for upstream basal channel width, downstream basal channel width, thalweg depth, and phi versus percent urban land use in 2001 46

Figure 3.1: Map of the Etowah River Basin showing the candidate road-stream crossings considered in site selection for mark-recapture studies 98

Figure 3.2: Diagram and photographs of study design and sampling methods 99

Figure 3.3: Diagram showing the *upstream* transition probabilities estimated in the Full Movement (FM) grouped model 100

Figure 3.4: Diagram showing the *upstream* transition probabilities estimated in the Reduced Movement (FM) grouped model. 101

Figure 3.5: Diagram showing assumptions of model with reach paired survival 102

Figure 3.6: Diagram showing assumptions of model with constant survival 103

CHAPTER 1

INTRODUCTION

Roadways have a significant presence on landscapes across the globe, with more than 6.4 million km of roads occurring in the United States alone. Although less than half of 1% of the land area in the United States is covered by roadways, roads have considerable ecological impacts (Forman and Alexander 1998, NCR 2005), threatening or endangering 94 species (Czech et al. 2000) and directly affecting an estimated 20% of land area in the United States (Forman 2000). Roads affect abiotic and biotic conditions in both terrestrial and aquatic environments, and the effects of roadways can vary spatially and temporally depending on location on the landscape, road and traffic density, road surface (e.g., impermeable vs. permeable), and engineering structures (e.g., concrete barriers, right-of-way fences, guardrails) (Trombulak and Frissell 2000, Forman and Deblinger 2000, Forman et al. 2003, Angermeier et al. 2004, NRC 2005).

Roads can alter the hydrologic and geomorphic characteristics of watersheds. For example, during storm events impermeable roadways typically increase the volume of runoff and sediment supplied to river and stream channels as well as the rate at which these materials are transported to channels (Forman and Alexander 1998). As a result, the processes that control channel geomorphology and form habitat for aquatic fauna can be altered (Wang et al. 2001, Forman et al. 2003, Wheeler et al. 2005). In addition to sediment, runoff generated on roadways carries chemical pollutants that accumulate from spills on roadways, application of road salts, waste by-products of vehicles, litter, and adjacent land uses which degrade water quality and

harm both terrestrial and aquatic fauna (Sutherland and Tolosa 2000, Trombulak and Frissell 2000, NRC 2005, Wheeler et al. 2005). Roads can also impact terrestrial and aquatic fauna by preventing dispersal (i.e., habitat fragmentation, avoidance behavior), enhancing dispersal (i.e., habitat corridors, vehicle transport), or causing direct mortality (i.e., roadkill) (Swihart and Slade 1984, Jones 2000, Hels and Buchwald 2001, Gelbard and Belnap 2003, Forman et al. 2003, NRC 2005).

Rivers and streams are longitudinally connected ecosystems that are particularly susceptible to fragmentation and geomorphic alteration. Due to the substantial number of roadways across the landscape and the high frequency with which roads cross streams, there is increasing recognition that road-stream crossings can affect the connectedness and local geomorphology of stream ecosystems. The rigid boundaries that culverts form do not change with stream channels as stream geomorphology changes through time, which can lead to fragmentation of adjusting channels and a number of geomorphic effects upstream and downstream of the crossing (Johnson and Brown 2000, Chin and Gregory 2001, Gubernick et al. 2003, Wheeler et al. 2005). These geomorphic effects often include local destabilization of stream banks, sediment deposition and channel aggradation upstream of the culvert, and scour and erosion downstream of the culvert.

Mounting evidence shows that road-stream crossings, in particular pipe and box culverts as these have typically been constructed, can indeed fragment stream ecosystems thereby limiting passage and dispersal of fishes (Belford and Gould 1989, Warren and Pardew 1998; Toepfer et al. 1999; Schaefer et al. 2003; Gibson et al. 2005) and many other aquatic fauna (Voelz et al. 1998, Blakely et al. 2006, Kerby et al. 2005, Resh 2005). Culverts that serve as barriers to fish movement cause fragmentation of habitat and fish populations, preventing fish

from accessing habitat necessary for the long-term maintenance of populations while promoting the loss of genomic heterogeneity and increasing the likelihood of local extinctions (Fausch and Young 1995, Schlosser and Angermeier 1995, Warren and Pardew 1998, Fausch et al. 2002, Jackson 2003, Wofford et al. 2005).

In a study of road-stream crossings constructed under U.S. Fish and Wildlife Service Reasonable and Prudent Measures (RPMs) (Chapter 2), I evaluated the efficacy of RPMs in protecting federally listed fish species by minimizing changes to channel geomorphology and effects on fish movement at road-stream crossings sites. Specifically, I evaluated geomorphic and stream crossing characteristics at a set of road-stream crossings constructed according to RPM terms and conditions, and assessed potential barriers to fish passage and local geomorphic effects associated with the RPM road-stream crossings. I also evaluated the level of compliance of each road-stream crossing in accordance with RPM terms and conditions.

In an open population mark-recapture study (Chapter 3), I evaluated the effects of culverts on the movement of small stream fishes. To assess the effects of differing types of culverts on fish movement, I conducted open population mark-recapture studies using resident fish assemblages at two sites with road-stream crossings constructed according to RPM terms and conditions and two sites with road-stream crossings not constructed according to RPM terms and conditions. I used a series of multi-state models to estimate culvert effects on the movement of small stream fishes at these sites.

Taken together, these studies provide further insight into the effects of road-stream crossings on stream geomorphology and fish movement. In addition, the open population mark-recapture study demonstrates novel methods for assessing fish movement by using multi-state

models to estimate movement probabilities while accounting for spatial variation in capture probabilities and in loss of individuals from the study area.

Literature Cited

Angermeier, P.L., A.P. Wheeler, and A.E. Rosenberger. 2004. A conceptual framework for assessing impacts of roads on aquatic biota. Fisheries 29: 19-29.

Belford, D.A., and W.R. Gould. 1989. An evaluation of trout passage through six highway culverts in Montana. North American Journal of Fisheries Management 9: 437-445.

Blakely, T.J., J.S. Harding, A.R. McIntosh, and M.J. Winterbourn. 2006. Barriers to recovery of aquatic insect communities in urban stream. Freshwater Biology 51: 1634-1645.

Czech, B., P.R. Krausman, and P.K. Devers. 2000. Economic associations among causes of species endangerment in the United States. BioScience 50: 593-601.

Chin, A., and K.J. Gregory. 2001. Urbanization and adjustments of ephemeral stream channels. Annals of the Association of American Geographers 91: 595-608.

Fausch, K.D., and M.K. Young. 1995. Evolutionary significant units and movements of resident stream fishes: a cautionary tale. Pages 360-370 *in* J.L. Nielson , editor. Evolution and the Aquatic System: Defining Unique Units in Population Conservation. American Fisheries Society Symposium 17.

Fausch, K.D., C.E. Torgersen, C.V. Baker, and H.W. Li. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. BioScience 52: 483-498.

Forman, R.T.T., and L.E. Alexander. 1998. Roads and their major ecological effects. Annual Review of Ecology and Systematics 29: 207-231.

Forman, R.T.T. 2000. Estimate of the area affected ecologically by the road system in the United States. Conservation Biology 14: 31-35.

Forman, R.T.T., and R.D. Deblinger. 2000. The ecological road-zone effect of a Massachusetts (U.S.A.) suburban highway. Conservation Biology 14: 36-46.

Forman, R.T.T., D. Sperling, J.A. Bissonette, A.P. Clevenger, C.D. Cutshall, V.H. Dale, L. Fahrig, R. France, C.R. Goldman, K. Heanue, J.A. Jones, F.J. Swanson, T. Turrentine, and T.C. Winter. 2003. Road ecology: science and solutions. Island Press, Washington, D.C.

Gelbard, J.L., and J. Belnap. Roads as corridors for exotic plant invasion in a semiarid landscape. Conservation Biology 17: 420-432.

Gibson, R.J., R.L. Haedrich, C.M. Wernerheim. 2005. Loss of fish habitat as a consequence of inappropriately constructed stream crossings. Fisheries 30: 10-17.

Gubernick, B., K. Clarkin, and M.J. Furniss. 2003. Design and construction of aquatic organism passage at road-stream crossings: site assessment and geomorphic considerations in stream simulation culvert design. Pages 30-41 *in* C.L. Irwin, P. Garnett, and K.P. McDermott , editors. 2003 Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment. North Carolina State University, Raleigh, NC.

Hels, T., and E. Buchwald. 2001. The effects of road kills on amphibian populations. Biological Conservation 99: 331-340.

Jackson, S.D. 2003. Ecological considerations in the design of river and stream crossings. Pages 20-29 *in* C.L. Irwin, P. Garnett, and K.P. McDermott , editors. 2003 Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment. North Carolina State University, Raleigh, NC.

Johnson, P.A., and E.R. Brown. 2000. Stream assessment for multicell culvert use. Journal of Hydraulic Engineering 126: 381-386.

Jones, M.E. 2000. Road upgrade, road mortality and remedial measures: impacts on a population of eastern quolls and Tasmanian devils. Wildlife Research 27: 289-296.

Kerby, J.L., S.P.D. Riley, L.B. Kats, and P. Wilson. 2005. Barriers and flow as a limiting factor in the spread of an invasive crayfish (*Procambarus clarkii*) in southern California streams. Biological conservation 126: 402-409.

NRC (National Research Council). 2005. Assessing and managing the ecological impacts of pave roads. National Academy Press, Washington, D.C.

Resh, V.H. 2005. Stream crossings and the conservation of diadromous invertebrates in South Pacific island streams. Aquatic Conservation 15: 313-317.

Schafer, J.F., E. Marsh-Matthews, D.E. Spooner, K.B. Gido, and W.J. Matthews. 2003. Effects of barriers and thermal refugia on local movement of the threatened leopard darter, *Percina pantherina*. Environmental Biology of Fishes 66: 391-400.

Schlosser, I.J., and P.L. Angermeier. 1995. Spatial variations in the demographic processes of lotic fishes: conceptual models, empirical evidence, and implications for conservation. Pages 392-401 *in* J.L. Nielson , editor. Evolution and the Aquatic System: Defining Unique Units in Population Conservation. American Fisheries Society Symposium 17.

Sutherland, R.A., and C.A. Tolosa. 2000. Multi-element analysis of road-deposited sediment in and urban drainage basin, Honolulu, Hawaii. Environmental Pollution 110: 483-495.

Swihart, R.K., and N.A. Slade. 1984. Road crossing in *Sigmodon hispidus* and *Microtus ochrogaster*. Journal of Mammalogy 65: 357-360

Toepfer, C.S., W.L. Fisher, and J.A. Haubelt. 1999. Swimming performance of the threatened leopard darter in relation to road culverts. Transactions of the American Fisheries Society 128: 155-161.

Trombulak, S.C., and C.A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology 14: 18-30.

Voelz, N.J., J.V. McAuthur, and R.B. Rader. 1998. Upstream mobility of the Asiatic clam *Corbicula fluminea*: identifying potential dispersal agents. Journal of Freshwater Biology 13: 39-45.

Wang, L., J. Lyons, and P. Kanehl. 2001. Impacts of urbanization on stream habitat and fish across multiple spatial scales. Environmental Management 28: 255-266.

Warren, M.L., and M.G. Pardew. 1998. Stream crossings as barriers to small-stream fish movement. Transactions of the American Fisheries Society 127: 637-644.

Wheeler, A.P., P.L. Angermeier, and A.E. Rosenberger. 2005. Impacts of new highways and subsequent landscape urbanization on stream habitat and biota. Reviews in Fisheries Science 13: 141-164.

Wofford, J.E.B., R.E. Gresswell, and M.A. Banks. 2005. Influence of barriers to movement on within-watershed genetic variation of coastal cutthroat trout. Ecological Applications 15: 628-637.

CHAPTER 2

ASSESSMENT OF THE EFFICACY OF U.S. FISH AND WILDLIFE SERVICE REASONABLE AND PRUDENT MEASURES IN PREVENTING GEOMORPHIC ALTERATION AND BLOCKED FISH PASSAGE AT ROAD-STREAM CROSSINGS

Introduction

Because streams are dynamic environments that experience geomorphic change through time, constructing immovable concrete and metal culverts at road-stream crossings can have geomorphic effects on stream channels. Natural streams undergo changes in depth and bank locations over time as stream width, channel depth and channel slope adjust where the streambed and banks are erodable (Knighton 1998). The rigid boundaries imposed by culverts resist geomorphic change and this can lead to fragmentation of adjusting channels (Chin and Gregory 2001, Gubernick et al. 2003, WDFW 2003).

Culverts often decrease the flow dimensions and alter slope of the channel at the crossing, thereby altering the capability of the channel to transport water, sediment and woody debris (Trombulak and Frissell 2000, Wellman et al. 2000, Gubernick et al. 2003, Wheeler et al. 2005). When the transport capabilities of a stream channel are altered at a road-stream crossing, the consequences often include local destabilization of stream banks, sediment deposition and channel aggradation upstream of the culvert, and scour and erosion downstream of the culvert (Johnson and Brown 2000, Chin and Gregory 2001, Gubernick et al. 2003, WDFW 2003, Chin and Gregory 2005). Culverts frequently concentrate and accelerate storm flows, leading to excessive flow velocities at culvert outlets that can further enhance scour and erosion of the streambed below the culvert (Johnson and Brown 2000). Excessive rates of erosion below a culvert can cause it to become perched above the streambed, or even above the water surface, potentially blocking fish passage (Jackson 2003, WDFW 2003, Wheeler et al. 2005). Moreover, culverts tend to trap sediment and debris around the culvert inlet, especially when culverts are narrow, poorly aligned with the stream channel, or when the barrels of multiple-barrel culverts are separated by areas of road-fill or concrete. This trapping of sediment and debris can lead to

debris dams at culvert inlets, further altering the transport capabilities of the channel and causing additional impediments to fish passage, and possibly even failure of the roadway (Wellman et al. 2000, Gubernick et al. 2003, Wheeler et al. 2005).

The problem of blocked fish passage at culverts is of particular concern in areas where Threatened or Endangered species protected under the Endangered Species Act (ESA) persist. Culvert installation typically requires a permit under Section 404 of the Clean Water Act, typically issued by the U.S. Army Corps of Engineers. Federal actions, including permitting actions, that may affect a species protected under the ESA can only proceed after the responsible federal agency consults with the U.S. Fish and Wildlife Service (FWS). FWS must find that the action will not jeopardize the species' survival or recovery, although 'take' may occur as a result of the permitted action. FWS may place requirements on the permit for the proposed action that are intended to minimize harm to the species. These requirements are issued as part of a FWS Biological Opinion, and are referred to Reasonable and Prudent Measures, or RPMs.

Over the course of the last several years, FWS has developed a series of RPMs as part of their Biological Opinions for listed fish species of the Etowah River Basin, Georgia, USA. A number of these RPMs have been for road-stream crossings, concerning both new crossings and crossings where bridges are being replaced with culverts. FWS has outlined RPMs for road-stream crossings in Biological Opinions issued for the Etowah River Basin to minimize the take of *Etheostoma scotti* (Cherokee darter) and *Etheostoma etowahae* (Etowah darter). The terms and conditions by which the RPMs are achieved vary somewhat, depending on the specifics of the proposed project. Nonetheless, they often share identical requirements. In the Biological Opinions issued to date for road-stream crossings in the Etowah River Basin, the RPMs are generally the same:

1. Minimize bank erosion and sedimentation in the construction areas.
2. Install new structures and replace existing structures in a manner that minimizes change to channel morphology and fish movement.
3. Minimize potential impacts to listed fish associated with pesticide and chemical use near the stream.

In order to help the FWS evaluate the efficacy of RPMs for road-stream crossing in protecting listed fish species and their habitat in action areas, we evaluated a set of road-stream crossings constructed under RPM terms and conditions and assessed the impacts these culverts may be currently having on channel geomorphology and fish movement. Based on previous findings, (e.g. Johnson and Brown 2000, Chin and Gregory 2001, Gubernick et al. 2003) we hypothesized that stream channels with road-stream crossings causing geomorphic change would exhibit some degree of channel narrowing and aggradation (sediment deposition and decreased depth/bar formation) upstream of the crossing, and channel widening and degradation (scour and increased depth/pool formation) downstream of the crossing. We used field measurements of current geomorphic condition at eleven RPM crossings to evaluate these hypotheses as well as the potential for the crossings to impede fish passage. We also assessed degree to which culvert installations complied with the RPM terms and conditions.

Methods

Study Sites and Site Selection

Study sites were located on small tributary streams in Cherokee and Paulding counties within the Etowah River Basin, GA. The Etowah River Basin lies on the northern edge of the Atlanta metropolitan area and drains 4823 km^2 in parts of eleven counties in northwest Georgia

(Fig. 2.1). Study sites were selected from all the locations in the Etowah River Basin where FWS had issued RPMs for road-stream crossings. We chose our study sites based on the road-stream crossing type and watershed size, focusing on single- and multiple-barrel box and pipe culverts and sites with watershed areas less than 52 km^2. Sites where bottomless arch-span culverts and bridges were constructed were excluded from the study. The upper watershed size limit of 52 km^2 was chosen because in Georgia, road-stream crossings on streams with watershed sizes greater than 52 km^2 typically are built with bridges instead of culverts. We also made reconnaissance visits to candidate sites to be sure that the project work associated with the individual U.S. Fish and Wildlife Service Biological Opinions had been completed and that the sites were accessible.

A total of eleven sites out of fifteen evaluated met criteria for inclusion. We conducted geomorphic and road-stream crossing surveys at seven sites in Cherokee County and four sites in Paulding County (Fig. 2.2). The eleven sites were covered by four separate Biological Opinions issued by the U.S. Fish and Wildlife Service between 03 December 1998 and 23 July 2003. Seven of the eleven sites were sites where bridges had been replaced by culverts while the remaining four sites were new road-stream crossing installations, all of which were associated with residential development.

All road-stream crossings surveyed for this study were constructed using single- or multiple-barrel box or pipe culverts (Fig. 2.3). Watershed size and land use were calculated using digital U.S. Geological Survey (USGS) topographic maps and 2001 USGS National Land Cover Database zone 60 land cover in ArcView® 3.3 geographic information systems (GIS) software.

Geomorphic and Road-Stream Crossings Surveys

Geomorphic and road-stream crossing surveys entailed measurements of channel and culvert characteristics at each study site. All surveys were conducted at baseflow. Particle size (pebble count), thalweg depth and basal channel width (wetted channel width at baseflow) were measured upstream and downstream of the culvert along longitudinal transects that were 25 times stream width. Measurements were taken at ½ stream width intervals for a total of 100 sample points, 50 upstream and 50 downstream of the culvert. Particle size (along the median particle axis, mm) was measured at a random proportion of wetted width at each sample point. Particle size measurements from the pebble count data were converted to phi scale values (i.e., the negative logarithm (in base two) of the median axis length in mm) before analysis (Gordon et al. 2004). The phi scale organizes particle size into the following intervals: boulders, -12 to -8 (4096-256 mm); cobbles, -8 to -6 (256 – 64 mm); gravels, -6 to -1 (64 – 2 mm); sands, -1 to 4 (2 – 0.0625 mm); and silts, 4 to 8 (0.0625 – 0.0039 mm). Bedrock was given a phi value of -12.

Culvert height and width, width of culvert bottom (where culvert intersected with stream bed), water depth inside the culvert, and the presence and type of sediment inside the culvert were measured with a tape measure or meter stick. We also measured the height (m) by which culverts were perched above the stream bed (PASB) and perched above the water surface (PAWS) both at the culvert inlet and outlet. Culvert measurements were used to calculate culvert embeddedness (defined as the depth to which the culvert bottom was buried into the stream bed). We also measured the estimated length of the culvert impact zone at each site. We defined the culvert impact (CI) zone as the length of stream immediately upstream or downstream of the culvert that appeared to have geomorphic characteristics differing from those

of the stream reach further away from the culvert and which may have been due to culvert effects on local geomorphic processes around the crossing.

Data Analysis

We analyzed data from the upstream and downstream transects separately to test the following specific hypotheses about how the presence of road-stream crossings might affect upstream and downstream geomorphology: decreased basal width upstream, increased basal width downstream, decreased thalweg depth and particle size upstream, and increased thalweg depth and particle size downstream. To test these hypotheses, the five measurements closest to the culvert were chosen to represent the CI, while the last 30 measurements taken away from the culvert were chosen to represent the stream channel outside of the CI (non-CI measurements). Based on visual estimates of CI length at each study site (Table 2.2), the first five measurements (i.e., 2.5 times basal channel width) near the culvert served as a conservative estimate of the CI, when present, at all sites. The last 30 measurements were outside of the visually estimated CI at all sites except at sites 10 and 11 (Table 2.2), where culvert impact may have extended 1 or 2 measurements downstream or upstream, respectively, into the last 30 measurements. At other sites, visual estimates of the CI impact extended ≤11 transect measurements away from the culvert. Using the last 30 measurements to characterize the stream-channel outside of the culvert impact area was thus conservative at the majority of the sites.

Basal channel width, thalweg depth and phi data were standardized at all sites by expressing the individual measurements from each transect in standard deviation units calculated as $(u_i - \bar{u}) / SD$, where u_i = sample *i* from the transect, $\bar{u}$ = mean of the 50 upstream or downstream transect measurements, and SD = standard deviation of the 50 upstream or

downstream transect measurements. The mean of the five CI and the 30 non-CI measurements were calculated using the standard deviation units (SDU) for the upstream and downstream transects at each site. The mean CI and non-CI values for were compared using separate paired one-tailed t-tests ($\alpha = 0.05$) for the upstream and downstream transect data.

We calculated a delta values for each site by taking the difference between the means of the standardized CI and non-CI data for the upstream and downstream reaches at each site. We also calculated a measure of relative culvert openness for each site by taking a ratio of the sum of the culvert barrel diameters (circular culverts) or widths (box culverts) at a crossing to the mean basal channel width. Relative culvert openness was considered an approximation of the maximum stream channel width available to pass stream flow, and thereby a surrogate for the amount of structure in the channel. The delta values for each site were plotted against relative culvert openness and 2001 percent urban land use to evaluate relations between geomorphic effects and the amount of structure in the channel and land use. Finally, longitudinal profiles were plotted using a running average of every four measurements superimposed over the raw data to graphically assess geomorphic changes moving away from the crossing.

Results

Watershed sizes ranged from 0.49 km^2 to 18.15 km^2, and sites were located primarily in rural and suburban residential areas. Urban land use ranged from 4.5% to 41% across the eleven sites. Of the eleven culverts surveyed, we surveyed seven three-barrel pipe culverts, one four-barrel pipe culvert, one single-box culvert, and two three-barrel box culverts (Table 2.1).

Mean basal channel width (MBCW) ranged from 2.40 to 6.39 m, and maximum culvert openness (Max O) ranged from 3.23 to 10.42 m across sites. The lowest and highest Max O

corresponded with the narrowest and widest MBCW, respectively (Table 2.2). Relative culvert openness (RCO) ranged from 1.24 to 3.48, with nine out of eleven sites having RCOs ranging between 1.24 and 1.91. The lowest RCO also corresponded with the narrowest MBCW, but the highest RCO did not correspond to the widest MCBW (Table 2.2). Pipe culvert embeddedness ranged from 0% to 38% of the culvert diameter, except at Site 7 where the culvert was perched 0.08 m above the streambed at the culvert outlet. Of the three box culverts surveyed, the two multiple-barrel culverts (Sites 3 and 6) had bottomless middle barrels and the single-barrel culvert (Site 11) was perched above the streambed at the culvert outlet 0.20 m (Table 2.2). Estimated upstream culvert impact zone (CI) length ranged from 0 to 37.2 m, and estimated downstream CI ranged from 0 to 33.4 m (Table 2.2).

We found no significant differences between mean CI and non-CI values for basal channel width (BCW) upstream or downstream the crossings at our sites (Table 2.3). Although no systematic changes in BCW were found between CI and non-CI stream segments, BCW delta values showed large shifts (>1 SDU) in the predicted directions (i.e., narrower upstream CI and/or wider downstream CI) at Sites 1, 2, 9, 10 and 11 (Table 2.3, Figs. 2.4 – 2.8) and in the opposite directions at Sites 11 (i.e., wider upstream CI) and 6 (i.e., narrower downstream CI) (Table 2.3, Figs. 2.8 and 2.9). Otherwise, differences between CI and non-CI BCW were small (<1 SDU) and not consistent. CI and non-CI thalweg depths upstream of the crossings were significantly different ($p = 0.04$), indicating streams tended to be more shallow near crossings compared to unimpacted upstream areas (Table 2.3). Although differences in upstream CI and non-CI thalweg depths were significant indicating shallower CI, differences were generally <1 SDU (Table 2.3). Downstream differences in thalweg depth were not significant, but we observed shifts >1 SDU in the predicted direction (i.e., deeper downstream CI) at Sites 2, 3 and

10 (Table 2.3, Figs. 2.10 and 2.11). We did not observe significant differences between CI and non-CI phi upstream of the crossings at our sites, and any apparent upstream differences were <1 SDU (Table 2.3). Differences between CI and non-CI phi values downstream of the crossing were significant ($p = 0.02$) and indicated systematic fining downstream of crossings (Table 2.3), but differences were small (<1 SDU) at all sites except Site 9 (Table 2.3, Fig. 2.12). There were no apparent linear trends between delta values for upstream or downstream BCW, thalweg depth or phi and RCO or percent urban land use in 2001 (Figs. 2.13 and 2.14), although the sites with the lowest RCO also exhibited the greatest downstream widening near the culvert (Figure 2.13b).

Discussion

Changes to channel geomorphology

Changes in channel geomorphology at RPM road-stream crossings varied among sites, with a consistent culvert effect evident only as shallower upstream thalweg depth and finer downstream bed sediment size. Additionally, geomorphic change did not generally appear to be propagated very far upstream or downstream away from the crossings. However, channel alteration around the culvert was pronounced at 7 of the 11 sites, including box and pipe culverts, and embedded and perched culverts, as discussed below.

Changes in basal channel width within the culvert impact zone (CI) were relatively greatest at Sites 1, 2, 6, 9, 10 and 11. We observed narrower upstream CI and wider downstream CI at Site 9, narrower upstream CI at Sites 1 and 2, and wider downstream CI at Sites 10 and 11. Upstream CI narrowing at Sites 1, 2 and 9 was due to sediment and debris accumulation along bank edges just upstream of the crossings. This sediment and debris accumulation appeared to result from a combination of sediment trapping, likely caused by culverts being undersized to

pass larger storm flows and their accompanying sediment and debris loads, and bank erosion and sluffing resulting from bank instability. Evidence that these culverts may have been undersized to pass larger storm flows and sediment and debris loads was given by the formation of channel bars directly upstream of the crossings and the erosion of road fill and riprap above and around the crossings. Observed downstream CI widening at Sites 9, 10 and 11 appeared to be the result of erosional processes in addition to artificial widening of stream channels as a result of the construction of crossings. Contrary to our predictions, the channel appeared widened upstream of the culvert at Site 11 and narrower downstream of the culvert at Site 6. However, these changes also reflected effects of culvert installation and downstream erosion. Upstream CI widening at Site 11 appeared to be the result of artificial widening caused by the construction process, while downstream CI narrowing at Site 6 was caused by bank sluffing and erosion of riprap and road fill immediately downstream of the crossing.

None of the apparent effects on basal channel width corresponded to culvert type or whether or not culverts were embedded. The sites showing largest relative changes in channel width included pipe and box culverts, and spanned the observed range of embeddedness (i.e., from perched by 20 cm at Site 10 to 38% embedded at Site 1).

Although there was evidence for a systematic shift in the predicted direction for CI thalweg depths upstream of the crossings at our sites (i.e., shallower CI), the differences between CI and non-CI values were relatively small (<1 SDU) and within the natural range of variation seen in the extended reaches beyond the CI zone (Appendix A: Figs. 12 – 22). Differences CI and non-CI thalweg depths downstream of the crossings were not significant across sites, however we did observe downstream deepening by >1 SDU below the culverts at Sites 2, 3 and 10. At Site 2, local hydrologic and geomorphic processes may also have been affected by a

small dam and reservoir less than 1 km downstream of the crossing (resulting in channel deposition downstream) and by a ≈ 90° meander bend with high, severely undercut banks about 40 m downstream of the culvert, which may have enhanced erosion near the culvert. Changes in downstream thalweg depth at Site 3 may have been related to natural variability in channel form because there were 5 other, equivalently deep pools within the study reach (Appendix A, Fig.14). Deeper CI depth at Site 10 appeared to result from the erosion of the streambed immediately downstream of the culvert, which appeared to have been inadequately restored after construction of the crossing was completed. The three sites with deepening downstream of the culvert again spanned culvert type (pipe and box) and degree of embeddedness (bottomless to perched).

There was no evidence of systematic fining of streambed sediments upstream of the crossings at our sites. Any apparent differences between upstream CI and non-CI phi values were small (<1 SDU), and did not appear to exceed the natural range of variation in phi values observed in the non-CI reaches (Appendix A: Figs. 23 – 33). Differences between CI and non-CI phi values downstream of the crossing were significant and indicated systematic fining downstream of the crossings at our sites. However, differences between CI and non-CI phi values were <1 SDU and also were within the natural range in variation of phi values outside of the CI zone (Appendix A), except at Site 9 (Fig. 12). Downstream CI fining was opposite to what we predicted, and indicates occurrence of depositional areas just below the crossings at many of our sites, including Site 9. Again, any apparent effects on phi did not seem to be related to culvert type or whether or not culverts where embedded. The sites with the largest relative changes in downstream CI phi (Sites 1, 7, 9 and 10) included perched and embedded pipe culverts and a perched box culvert.

Changes to fish passage

The crossings at Sites 1, 2, 3, 5 and 6 were all bottomless or embedded culverts (both box and pipe) with substrate throughout and water depths adequate for fish passage. In addition, we observed fishes in the culverts at these five sites during our surveys, so it is likely that these crossings are not acting as barriers to fish movement. We did not observe fishes in the culverts at Sites 4, 7, 8, 9, 10 or 11. The crossings at Sites 4 and 9 (pipe culverts) are probably not barriers to fish passage, as they are embedded, have water depths that appear adequate to pass fish, and have gravel and sand through the barrels conveying most of the flow in the channel. However, the crossings at Sites 7, 8, 10 and 11 (box and pipe culverts) all have the potential to be barriers to fish passage at baseflow.

The crossing at Site 7 was perched above the stream bed 0.08 m, no substrate was present inside the culvert, water depth inside the culvert was excessively shallow (<3 cm) at base flow, and a debris dam (0.7 m tall) was blocking the culvert inlet, all of which could potentially block upstream fish passage. The presence of the debris dam, the shallow water depth inside the culvert, and the lack of substrate inside the culvert could potentially prevent downstream fish passage as well. Even though the crossing at Site 8 was embedded 14%, it may still have been serving as a barrier to both upstream and downstream fish passage as there was a 0.4 m debris dam blocking the culvert inlet in summer 2005, and that was still present and noticeably larger in summer 2006. The crossing at Site 10 was the most poorly constructed crossing, with water depth in the culvert ≤5 mm, no substrate present inside the culvert, and the outlet perched 0.20 m above the streambed and 1 cm above the water surface. The crossing at Site 10 almost certainly blocks upstream and downstream fish passage at base flow, although fish may be able to pass through the culvert at higher flows. At Site 11, although the crossing was embedded 19%, the

streambed upstream and downstream of the crossing had been filled in with cobble-sized riprap and the downstream channel had been artificially widened so extensively that the stream was barely flowing and water depths were quite shallow (DS thalweg depth = 3 cm, US thalweg depth = 2 cm). We suspected water quality in the downstream segment might have been poor because thick mats of algae and orange bacteria covered the riprap and streambed. Algal growth may have been enhanced by treated sewer water being pumped into the channel upstream and by the stagnant flow through the widened segment. We observed dead amphibians in the same segment, which also was suggestive of degraded water quality. With little to no current, shallow water depths and poor water quality downstream of the crossing and equally shallow water depths upstream of the crossing, the channel around the crossing, not necessarily the crossing itself, is likely to block upstream and downstream fish passage at Site 11.

Applicant compliance with RPM terms and conditions

The level of applicant compliance with RPM terms and conditions varied from site to site. Sites 1 – 7 were covered under the same Biological Opinion (03 December 1998), and Sites 1, 3, 5 and 6 appeared to be in full compliance with the RPM terms and conditions outlined in the Biological Opinion. At the time we conducted our surveys, the crossings at Sites 2, 4 and 7 were not embedded the minimum 0.46 to 0.61 m (18 to 24 in) required by the RPM terms and conditions stated in the Biological Opinion. At Sites 2 and 4, it is possible that the culverts were embedded at or beyond the required depth, but that embedded material has since been eroded out of the culvert. Consequently, we cannot say for certain whether or not the applicants were in full compliance with the RPM terms and conditions. However, Site 7 was quite clearly never

embedded below the streambed, and therefore the applicants were not in full compliance with the RPM terms and conditions.

Sites 8 and 9 were covered under a different Biological Opinion (01 September 1999). The terms and conditions for implementing the RPMs in the Biological Opinion stated that the center pipe of each triple-barrel crossing would be set about 0.305 m (12 in) below the streambed. At the time surveys were conducted, the crossings at Sites 8 and 9 were not embedded the 0.305 m (12 in) required by the RPM terms and conditions. Crossings at Sites 8 and 9 may have been embedded 0.305 m or more when originally constructed, with backfilled material subsequently eroding out of the culvert. Accordingly, we cannot say for certain whether the applicants fully complied with the RPM terms and conditions at Sites 8 and 9.

Site 10 was also covered under a separate Biological Opinion (02 October 2001). The terms and conditions for implementing the RPMs in the Biological Opinion did not state that the crossing needed to be embedded, but it was stated that the crossing should be designed so that the "normal width and depth of the stream will be maintained to minimize changes in stream velocity after culvert installation," which in turn is meant to minimize changes to fish passage. The crossing installed at Site 10 did not maintain normal stream depth (mean stream depth = 0.12 m, water depth in culvert = ≤0.05 m) or stream width (mean stream width = 2.40 m, culvert width = 3.23 m), and as a result, the applicant was not in full compliance with the RPM terms and conditions.

For Site 11, the RPM terms and conditions in the Biological Opinion (07 August 2001) stated that stream geomorphology must not be altered outside of the "direct construction area" as a result of construction activities. The applicants clearly violated this term and condition, as was evidenced by the excessive channel widening downstream of the crossing, and channel in-filling

with riprap upstream and downstream of the crossing, both of which created potential barriers to fish passage. Therefore, the applicants were not in full compliance with RPM terms and conditions as outlined in the Biological Opinion.

Effectiveness of RPMs and suggestions for improvement – Stream simulation design

In general, the road-stream crossing RPMs appeared to be relatively effective in preventing changes to channel geomorphology with respect to channel depth and substrate composition, but less effective in preventing changes to channel width and blocked fish passage. The RPMs mostly resulted in embedded culverts that, at present, should not pose barriers to fish passage. Most of the changes to channel geomorphology at RPM crossings resulted from the following factors, or some combination thereof: (1) unstable banks around the crossing, (2) excessive channel widening around the crossing during construction that was inadequately restored, and (3) culverts being undersized to pass high flows and high sediment and debris loads. Concerns about blocked fish passage at RPM crossings were due to (1) shallow depths inside the culvert or around the crossing, (2) lack of substrate inside the culvert, (3) debris dams at culvert inlets, and (4) culverts being perched above the streambed or water surface.

In order for road-stream crossing RPMs to be more effective in preventing changes to channel morphology and fish passage, RPM terms and conditions concerning long-term channel stabilization should be elaborated upon and given even greater importance than they are currently. All RPM terms and conditions should include provisions explicitly requiring that the geomorphic structure of the stream channel be fully restored once crossing installation is complete, giving special consideration to restoring channel width, channel bed forms, and bank edges, slopes and height. An even more effective measure would be for RPM terms and

conditions to disallow the use of multiple-barrel pipe culverts in crossing designs. When multiple culverts are used to span a stream there is high risk that debris will accumulate at one or more of the multiple culvert inlets, effectively creating a barrier to fish movement, blocking stream flow and preventing sediment transport (Bob Barnard, WDFW 2005, personal communication), as we observed at Sites 7 and 8. In place of multiple-barrel pipe culverts, RPMs should require applicants to use bottomless arch span culverts, or large single-barrel pipe or multiple-barrel box culverts (preferably with bottomless centers) designed using stream simulation culvert design techniques (Gubernick and Bates 2003, WDFW 2003, Norman et al. 2006, RSCP 2006).

Stream simulation design assumes that if fishes can migrate through a natural stream reach, then they will be able to migrate through a man-made reach that simulates the natural stream channel. Stream simulation culverts are designed to be wider than bankfull width (generally 1.2 times bankfull width plus an additional two feet) and are filled with a sediment mix that emulates the natural channel (WDFW 2003). Bankfull width is defined as the maximum discharge which can be contained within the channel without overtopping the banks (Leopold et al. 1964), and is generally accepted to represent the flow that occurs on average every 1 to 2.3 years (Williams 1978). Thus, bankfull width is typically wider than the basal channel width. Stream simulation culverts are also embedded 30% to 50% of the culvert rise or diameter (where possible). Embedding pipe and box culverts 30% to 50% creates a deep, monolithic bed structure inside the culvert and allows for significant bed adjustments without encountering the culvert bottom, and potentially causing the culvert to become perched (WDFW 2003).

Because stream simulation culverts are sized wider than the wetted channel, it is necessary to reconstruct the channel form and bank margins inside the culvert after installation, or to preserve the natural channel during construction, in order to provide the appropriate water depths and velocities over a variety of flows (RSCP 2006). Without reconstructing or preserving channel form and bank margins inside culverts, the width of the culvert will likely create depths that are too shallow for fish passage during low flows. We observed such a situation at Site 11, where the width of the actual channel immediately downstream (widened during culvert installation) of the crossing created shallow depths that were likely to impede fish passage. Channel width inside a culvert (stream simulation or not), as well as immediately upstream and downstream of the crossing, should match the basal channel width of the stream, and bank margins should extend through the culvert, approximating the bank margins of the adjacent stream channel and providing instream bank-edge habitat (RSCP 2006).

Conclusions

The efficacy of the FWS road-stream RPMs in preventing geomorphic changes could be improved by placing greater emphasis on bank and channel form restoration following construction of crossings, by prohibiting the use of multiple-barrel pipe culverts in road-stream crossing design, and by requiring applicants to design road-stream crossings following the stream simulation approach (see Grubernick and Bates 2003, WDFW 2003, Norman et al. 2006, and RSCP 2006 for more detailed discussions of the stream simulation approach to road-stream crossing design). Greater enforcement of applicant compliance with RPM terms and conditions could also help improve the efficacy of FWS RPMs in preventing channel alterations and blocked fish passage.

In watersheds where extensive changes in runoff and stream hydrology are expected as result from construction of future land developments, it may best to install bridges rather than culverts at road-stream crossings. Urban land use in the watersheds where the RPM road-stream crossings are located has been steadily increasing over the last two decades; increases in percent urban land use between 1992 and 2001 has ranged from 0.78% to 36.45%. As land use continues to change, so will the volume of water, sediment and woody debris that is delivered to the channels by their watersheds, and the channels will adjust to transport the amount of water and material being supplied to them. These adjustments could lead to future changes to channel morphology and blocked fish passage at crossings, including those studied here as well as at future crossings built with culverts. Use of bridges in watersheds experiencing rapid urban development will allow for greater stream channel adjustments as watershed hydrology changes, minimizing the effects of road-stream crossings on channel geomorphology and fish passage and the need to replace culverts as they become inadequate to pass changing stream flows.

Acknowledgements

Mary Freeman and Megan Hagler were incredibly helpful with study design and data analysis and interpretation, and have my sincere gratitude. Fieldwork was conducted by members of the Freeman lab, with special thanks to Megan Hagler, Greg Anderson, Renee Machyousky and Rebecca Coleman. We wish to thank Judy Meyer and David Leigh for suggestions that improved this work. Funding was provided by the U.S. Fish and Wildlife Service to Byron J. Freeman, through Research Work Order 79, USGS GA Cooperative Fish and Wildlife Research Unit.

Literature Cited

Chin, A., and K.J. Gregory. 2001. Urbanization and adjustments of ephemeral stream channels. Annals of the Association of American Geographers 91: 595-608.

Chin, A., and K.J. Gregory. 2005. Managing urban river channel adjustments. Geomorphology 69: 28-45.

Gordon, N.D., T.A. McMahon, B.L. Finlayson, C.J. Gipple, and R.J. Nathan. 2004. Stream hydrology: an introduction for ecologists, 2nd ed. John Wiley and Sons, West Sussex, England.

Gubernick, B., and K.K. Bates. 2003. Designing culverts for aquatic organism passage: stream simulation culvert design. Pages 42-52 *in* C.L. Irwin, P. Garnett, and K.P. McDermott , editors. 2003 Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment. North Carolina State University, Raleigh, NC.

Gubernick, B., K. Clarkin, and M.J. Furniss. 2003. Design and construction of aquatic organism passage at road-stream crossings: site assessment and geomorphic considerations in stream simulation culvert design. Pages 30-41 *in* C.L. Irwin, P. Garnett, and K.P. McDermott , editors. 2003 Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment. North Carolina State University, Raleigh, NC.

Jackson, S.D. 2003. Ecological considerations in the design of river and stream crossings. Pages 20-29 *in* C.L. Irwin, P. Garnett, and K.P. McDermott , editors. 2003 Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment. North Carolina State University, Raleigh, NC.

Johnson, P.A., and E.R. Brown. 2000. Stream assessment for multicell culvert use. Journal of Hydraulic Engineering 126: 381-386.

Knighton, D. 1998. Fluvial forms and processes: a new perspective. Oxford University Press, New York, NY.

Leopold, L.B., M.G. Wolman, and J.P. Miller. 1964. Fluvial processes in geomorphology. W.H. Freeman and Co., New York, NY.

Norman, J.R., E. Franzen, H. Millington, B. Ensign, S. Wenger, M. Freeman, and M. Hagler. 2006. Stream crossing and culvert design policy. http://www.etowahhcp.org/research/documents/road_crossings_tech_rpt_2006_08_18.pdf.

RSCP (River and Stream Continuity Partnership) (Univ. of Massachusetts Amherst, MA Riverways Program, The Nature Conservancy). 2006. Massachusetts River and Stream Crossing Standards. http://www.streamcontinuity.org/pdf_files/MA_Crossing_Stds_3-1-06.pdf.

Trombulak, S.C., and C.A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology 14: 18-30.

WDFW (Washington Department of Fish and Wildlife). 2003. Design of Road Culverts for Fish Passage. Washington Department of Fish and Wildlife. http://wdfw.wa.gov/hab/engineer/cm/.

Wellman, J.C., D.L. Combs, and S.B. Cook. 2000. Long-term impacts of bridge and culvert construction or replacement on fish communities and sediment characteristics of streams. Journal of Freshwater Ecology 15: 317-328.

Wheeler, A.P., P.L. Angermeier, and A.E. Rosenberger. 2005. Impacts of new highways and subsequent landscape urbanization on stream habitat and biota. Reviews in Fisheries Science 13: 141-164.

Williams, G.P. 1978. Bankfull discharge of rivers. Water Resources Research 14: 1141-1154.

Table 2.1. Road-stream crossing description, culvert diameter (D) for pipe culverts and width (W) for box culverts, watershed size (km^2) and 2001 percent urban land use for study sites.

Site	Road-Stream Crossing Description	Culvert D or W (m)	Watershed Size (km^2)	2001 % Urban
Canton Creek (Site 1)	3 barrel pipe culvert, corrugated metal	3.18	4.09	9.6
Canton Creek (Site 2)	3 barrel pipe culvert, corrugated metal	2.09	6.29	9.9
Scott Mill Creek (Site 3)	3 barrel box culvert, concrete, headwalls; middle barrel bottomless	3.05	8.54	23.2
Unnamed trib to the Etowah River (Site 4)	4 barrel pipe culvert, corrugated metal	2.16	1.85	18.6
Unnamed trib to Sharp Mountain Creek (Site 5)	3 barrel pipe culvert, corrugated metal	2.19	3.49	35.9
Bluff Creek (Site 6)	3 barrel box culvert, concrete, headwalls; middle barrel bottomless	3.47	18.15	4.5
Unnamed trib to Smithwick Creek (Site 7)	3 barrel pipe culvert, corrugated metal	1.43	1.62	7.4
McEver's Branch (Site 8)	3 barrel pipe culvert, corrugated metal, headwalls	1.46	1.42	41.1
McEver's Branch (Site 9)	3 barrel pipe culvert, corrugated metal, headwalls	1.46	1.55	41.1
Unnamed trib to Little Pumpkinvine Creek (Site 10)	1 barrel box culvert, concrete, headwalls	3.23	0.49	14.2
Unnamed trib to Westbrook Creek (Site 11)	3 barrel pipe culvert, corrugated metal, headwalls	1.77	3.55	10.1

Table 2.2. Mean basal channel width (MBCW), maximum culvert openness (Max O), relative culvert openness (RCO), and the depth and percent of culvert diameter (pipe culverts) or height (box culverts) embedded at each site. The length of the geomorphic culvert impact (CI) area is based on visual field assessment. (PASB = the height (m) by which culverts were perched above the stream bed).

Site	MBCW (m)	Max O (m)	RCO	Depth (m) and % embedded	Estimated CI length (m) US/DS
1	4.51	9.54	2.14	1.19 / 38%	0 / 14
2	4.15	6.27	1.55	0.38 / 18%	16.2 / 15.3
3	4.93	9.15	1.59	Bottomless	0 / 0
4	3.13	8.64	3.48	0.33 / 25%	7.3 / 11.1
5	3.13	6.57	1.91	0.64 / 29%	12.4 / 0
6	6.39	10.42	1.80	Bottomless	23.2 / 12.9
7	2.63	4.29	1.59	0 (Culvert outlet PASB = 0.08 m)	12.8 / 0
8	2.67	4.38	1.78	0.21 / 14%	8.5 / 14.6
9	2.40	4.38	1.57	0.12 / 8%	7.9 / 13.7
10	2.40	3.23	1.24	0 (Culvert outlet PASB = 0.20 m)	13.2 / 25.4
11	3.28	5.31	1.44	0.34 / 19%	37.2 / 33.4

Table 2.3. Upstream (US) and downstream (DS) culvert impact measurements (Δ) for basal channel width (BCW), thalweg depth and phi, with results of paired t-tests for each variable. Bolded and underlined values indicate observed effects corresponding to the following hypothesized effects within the culvert impact zone (CI): channel narrowing US, channel widening DS, decreased depth and phi US, increased depth and phi DS.

Site	**Δ US BCW**	**Δ DS BCW**	**Δ US Depth**	**Δ DS Depth**	**Δ US phi**	**Δ DS phi**
1	**-1.81**	**0.24**	**-0.18**	**0.56**	-0.52	0.87
2	**-1.33**	**0.64**	1.15	**1.44**	**0.31**	**-0.18**
3	0.37	-2.34	**-0.64**	**1.35**	-0.64	**-0.36**
4	0.45	**0.09**	**-0.91**	-0.51	-0.44	0.28
5	**-0.37**	**0.11**	**-0.50**	-0.84	-0.48	0.11
6	**-0.67**	-1.86	**-0.08**	0.31	-0.06	0.08
7	0.44	-0.83	**-0.69**	-0.08	-0.05	0.78
8	0.86	**0.91**	0.17	**0.24**	**0.54**	0.32
9	**-1.14**	**1.80**	**-0.76**	-0.50	**0.07**	1.56
10	0.33	**1.17**	**-0.88**	**2.26**	-0.03	1.00
11	1.91	**2.53**	**-0.58**	-0.56	**0.13**	0.50
Paired t-test (t, p)	-0.271, 0.40	0.508, 0.31	-1.946, 0.04	1.108, 0.15	-0.951, 0.18	2.668, 0.02

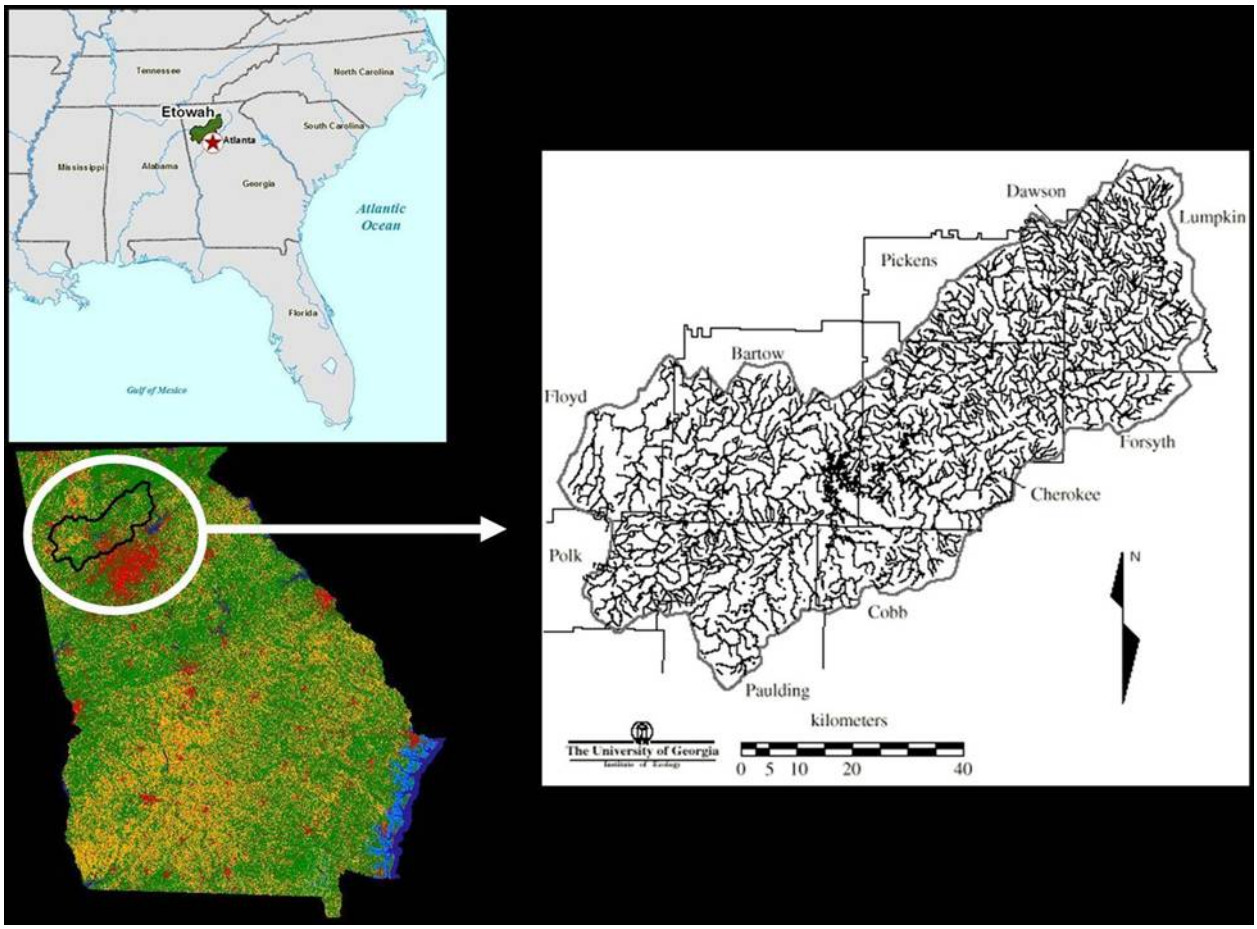

Fig. 2.1. Map of the Etowah River Basin. The location of the Etowah River Basin in the southeast (upper left), in the state of Georgia (lower left – 2001 USGS National Land Cover Database zone 60 land cover shown; urban land cover shown in red, forest land cover shown in green), and the counties that are included in the basin (right) are shown.

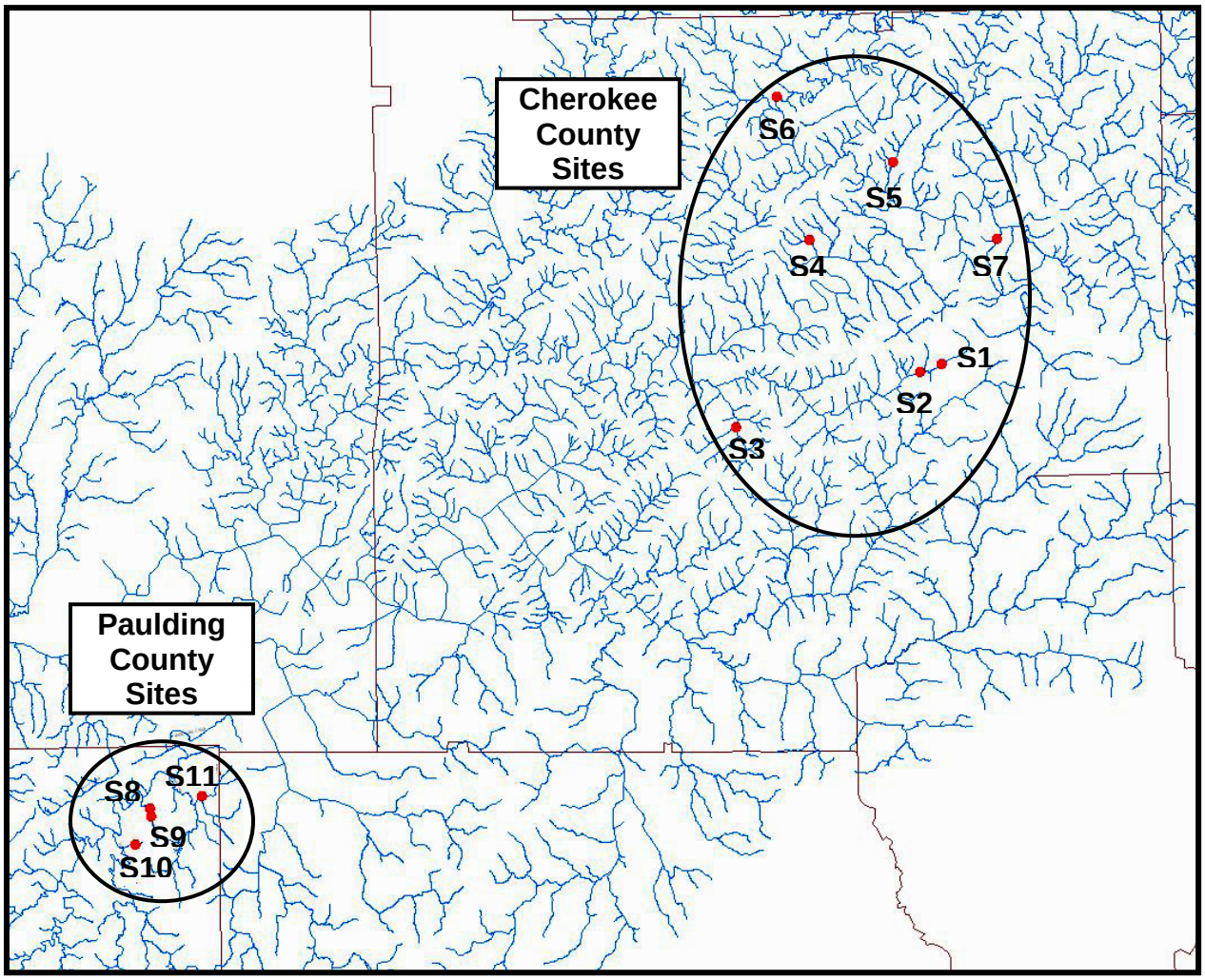

Fig. 2.2. Map of RPM road-stream crossing study sites.

Fig. 2.3. Examples of multiple-barrel pipe and multiple-barrel box culverts from study sites. A.) Canton Creek (Site 1) B.) Unnamed trib to Westbrook Creek (Site 11) C.) Scotts Mill Creek (Site 3) D.) Bluff Creek (Site 6).

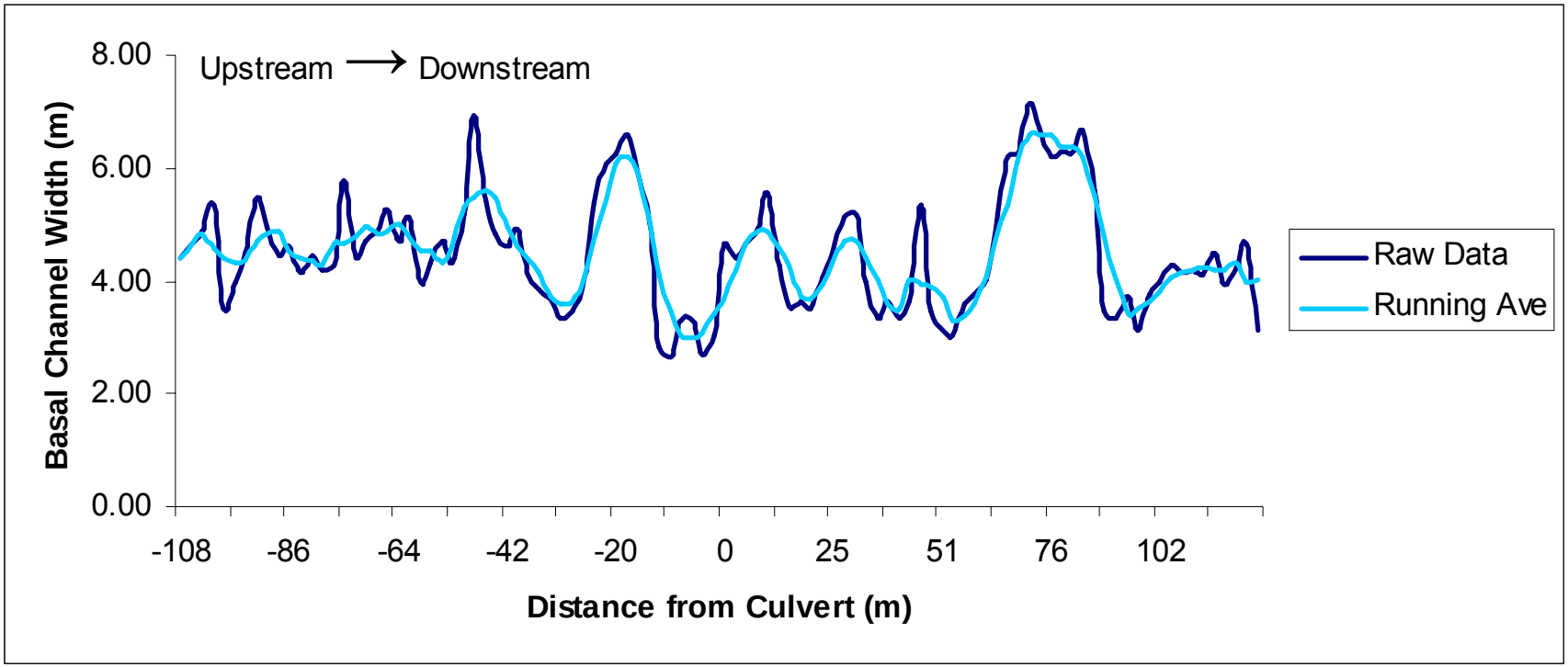

Fig. 2.4. Longitudinal plot of basal channel width (raw data and running averages of 4 transects) at Site 1, showing channel narrowing upstream of the culvert (located at 0 on the x-axis) in comparison to other upstream measurements.

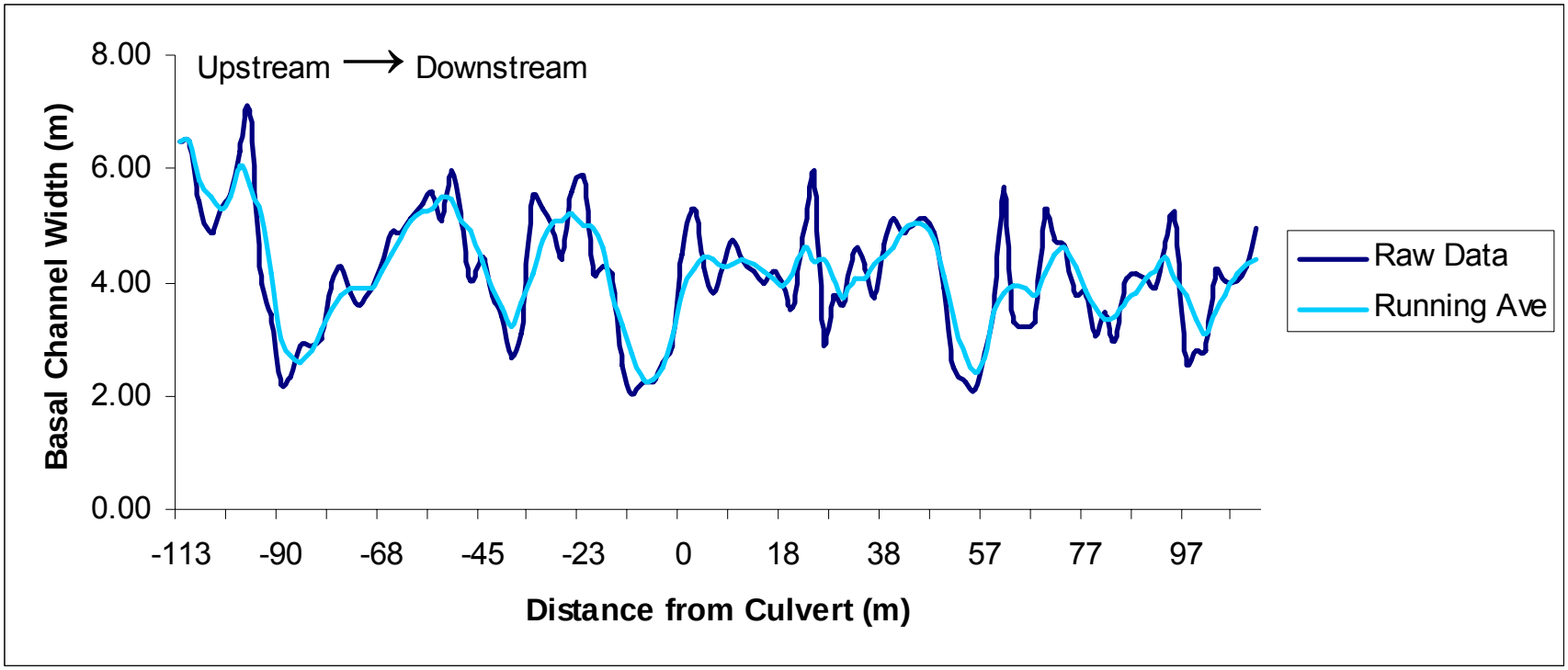

Fig. 2.5. Longitudinal plot of basal channel width (raw data and running averages of 4 transects) at Site 2, showing channel narrowing upstream of the culvert (located at 0 on the x-axis) in comparison to other upstream measurements.

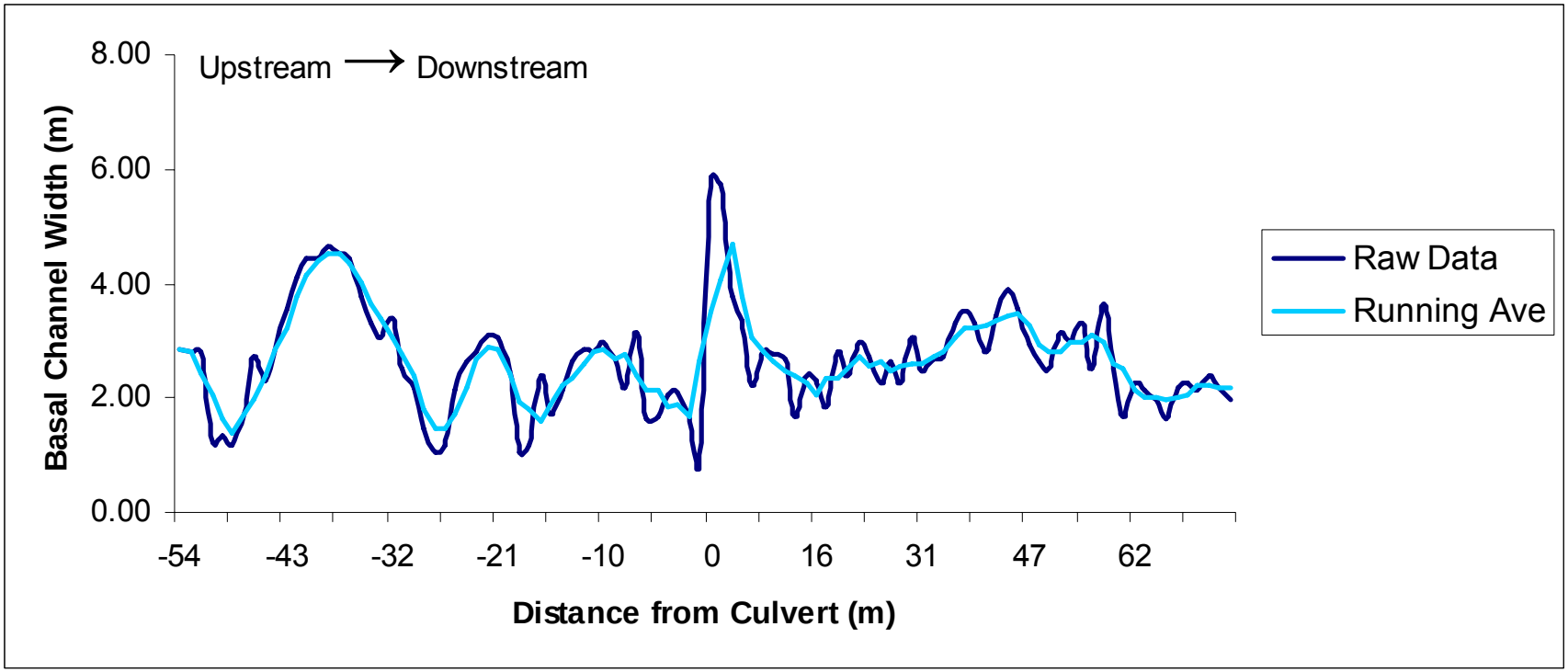

Fig. 2.6. Longitudinal plot of basal channel width (raw data and running averages of 4 transects) at Site 9, showing channel narrowing upstream of the culvert (located at 0 on the x-axis) and channel widening downstream of the culvert in comparison to other upstream and downstream measurements.

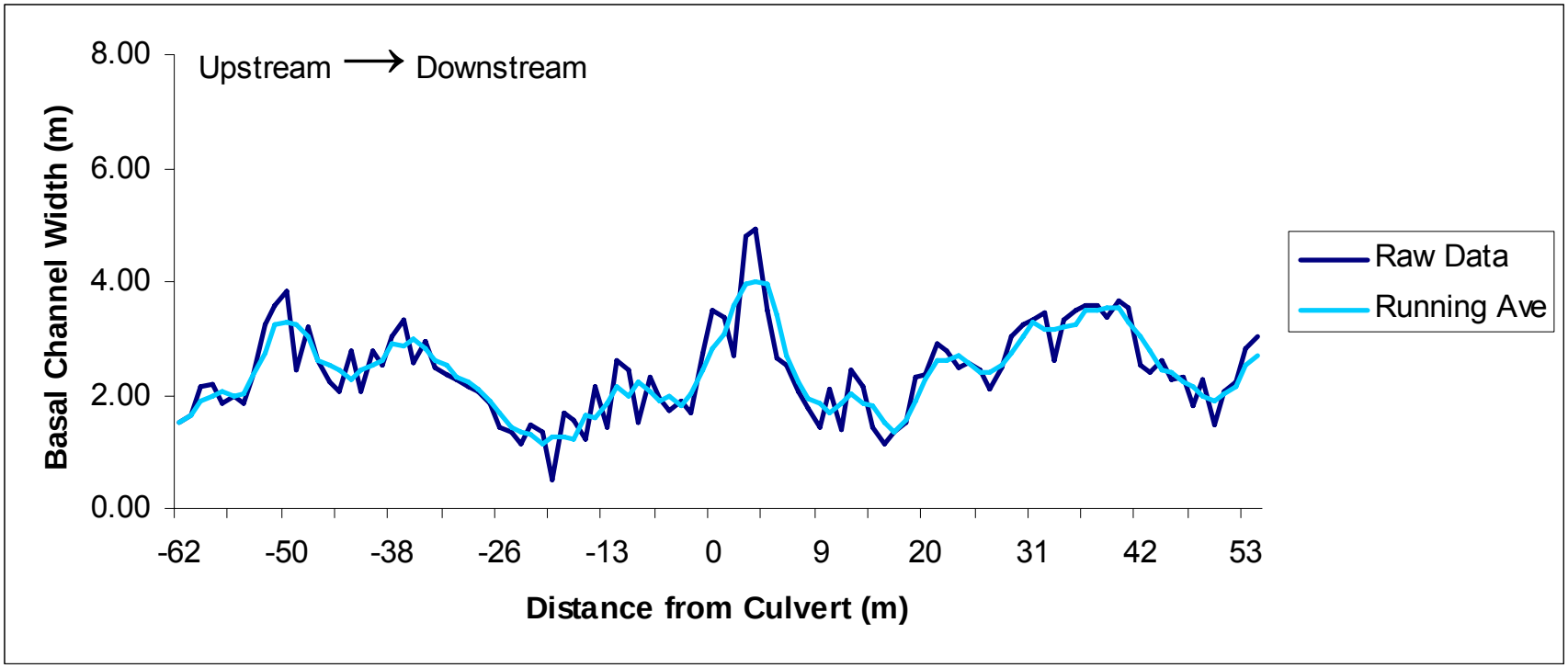

Fig. 2.7. Longitudinal plot of basal channel width (raw data and running averages of 4 transects) at Site 10, showing channel widening downstream of the culvert (located at 0 on the x-axis) in comparison to other downstream measurements.

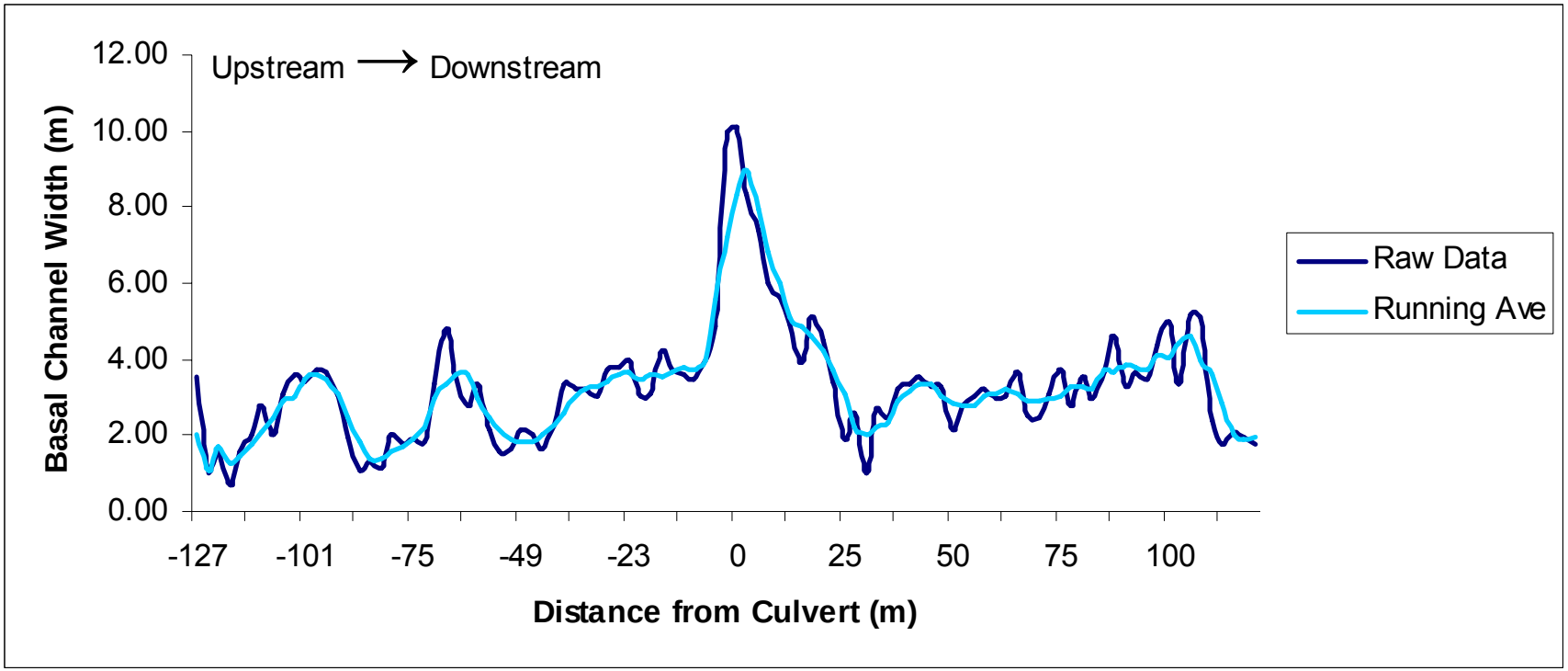

Fig. 2.8. Longitudinal plot of basal channel width (raw data and running averages of 4 transects) at Site 11, showing channel widening upstream of the culvert (located at 0 on the x-axis) and channel widening downstream of the culvert in comparison to other downstream measurements.

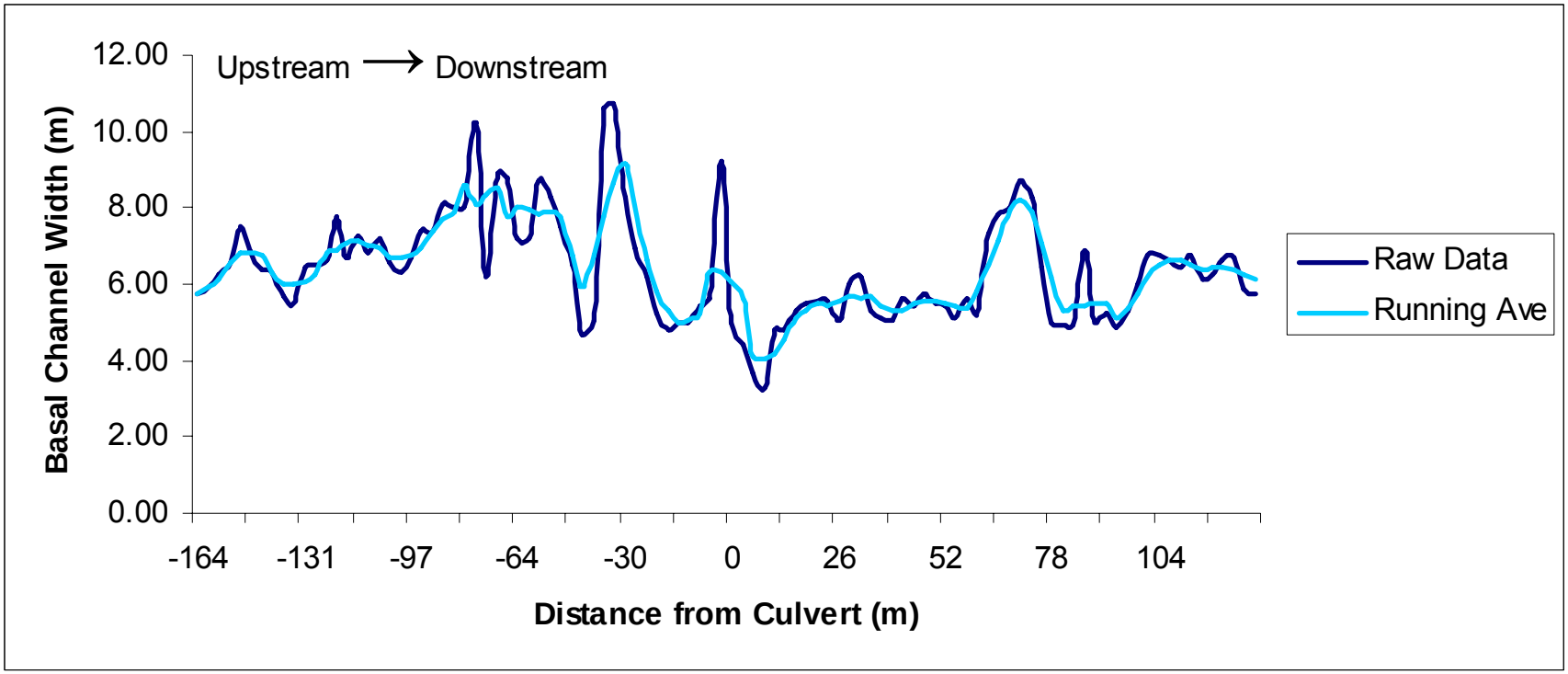

Fig. 2.9. Longitudinal plot of basal channel width (raw data and running averages of 4 transects) at Site 6, showing channel narrowing downstream of the culvert (located at 0 on the x-axis) in comparison to other downstream measurements.

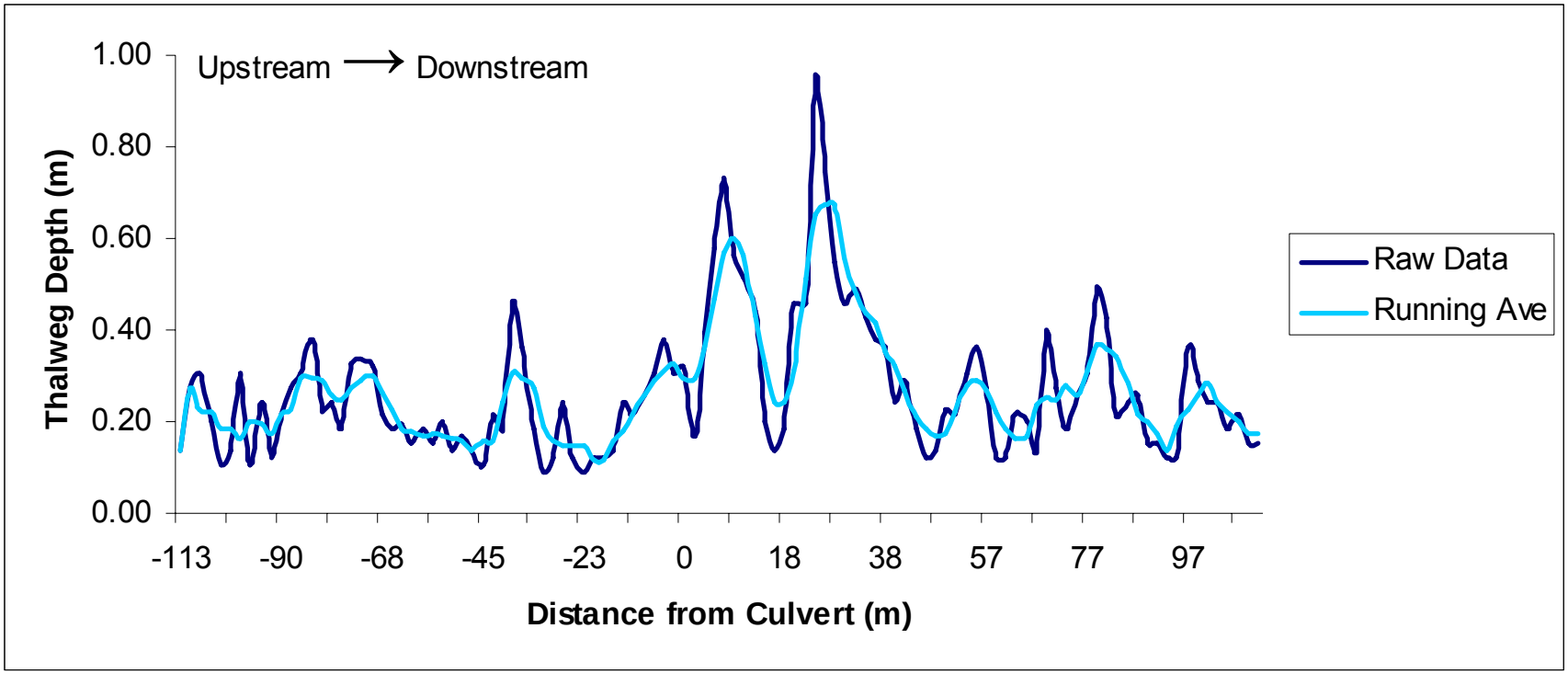

Fig. 2.10. Longitudinal plot of thalweg depth (raw data and running averages of 4 transects) at Site 2, showing channel deepening downstream of the culvert (located at 0 on the x-axis) in comparison to other downstream measurements.

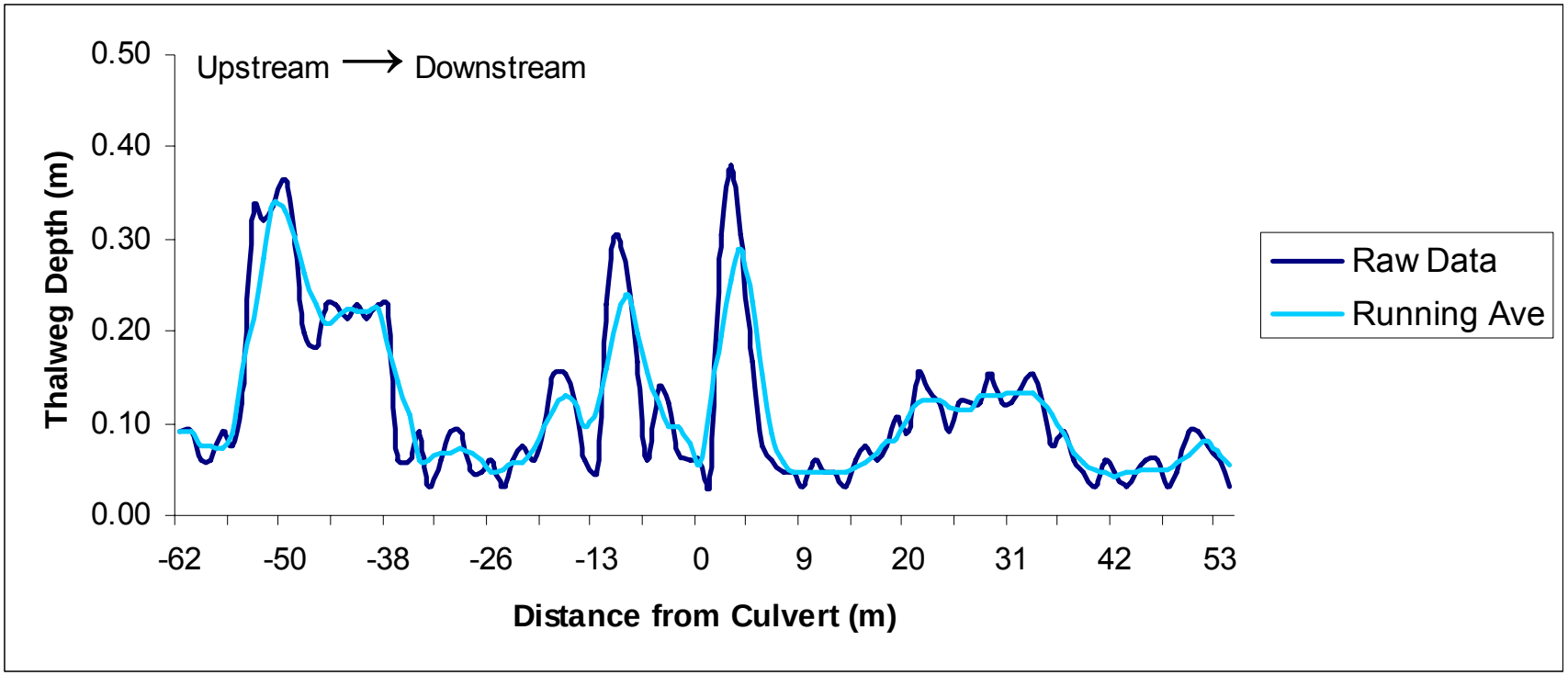

Fig. 2.11. Longitudinal plot of thalweg depth (raw data and running averages of 4 transects) at Site 10, showing channel deepening downstream of the culvert (located at 0 on the x-axis) in comparison to other downstream measurements.

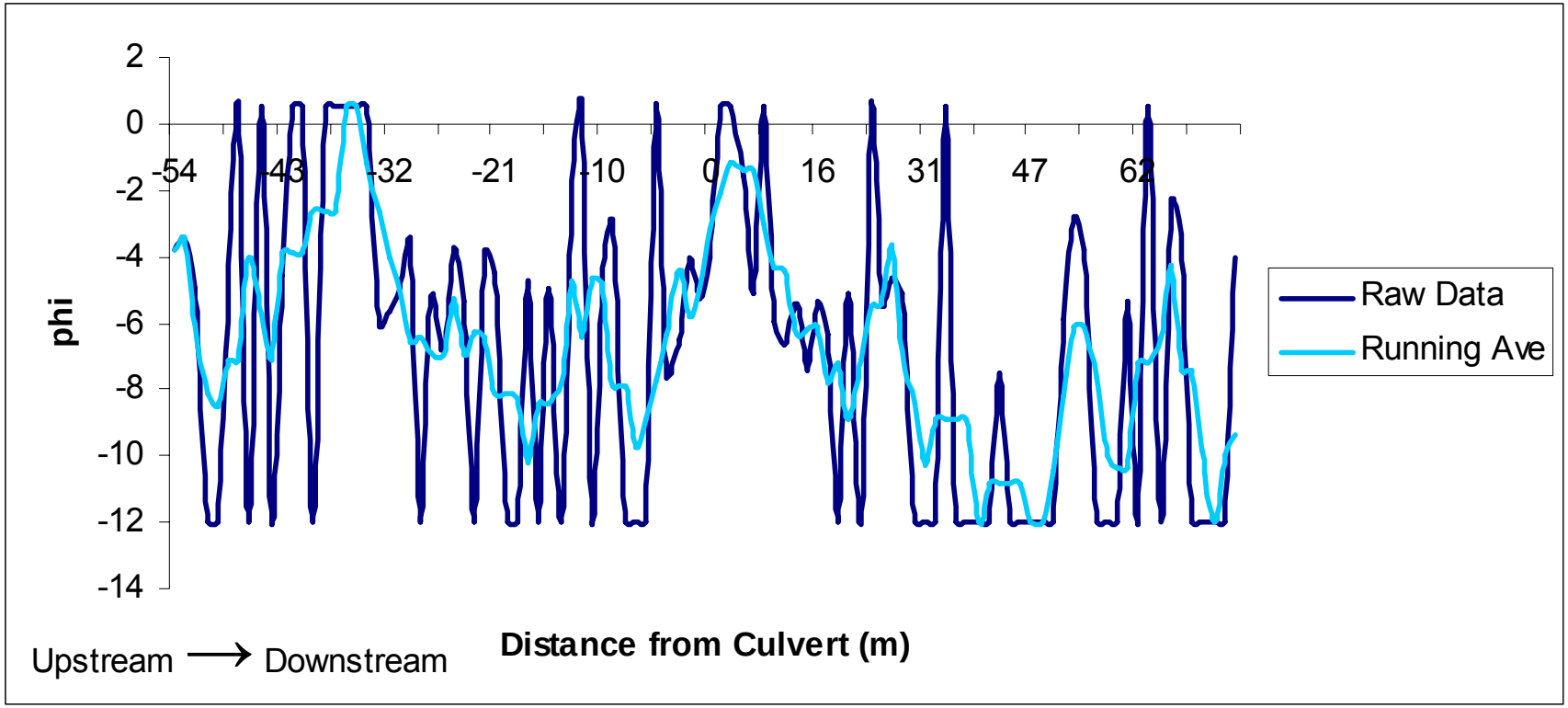

Fig. 2.12. Longitudinal plot of phi values (raw data and running averages of 4 transects) at Site 9, showing finer bed sediment size downstream of the culvert (located at 0 on the x-axis) in comparison to other downstream measurements.

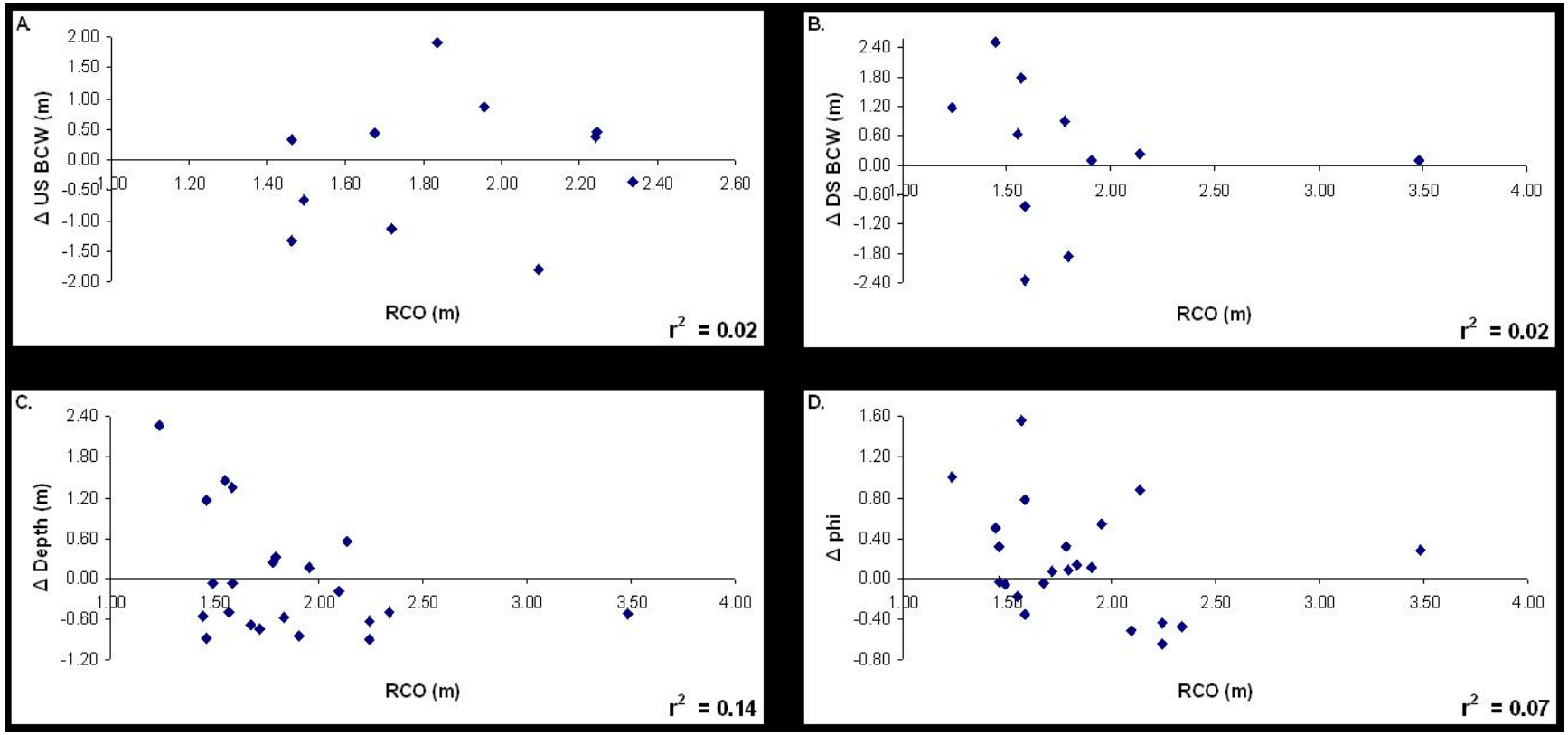

Fig. 2.13. Delta (Δ) values for A.) upstream basal channel width (US BCW), B.) downstream basal channel width (DS BCW), C.) thalweg depth (Depth), and D.) phi versus relative culvert openness (RCO).

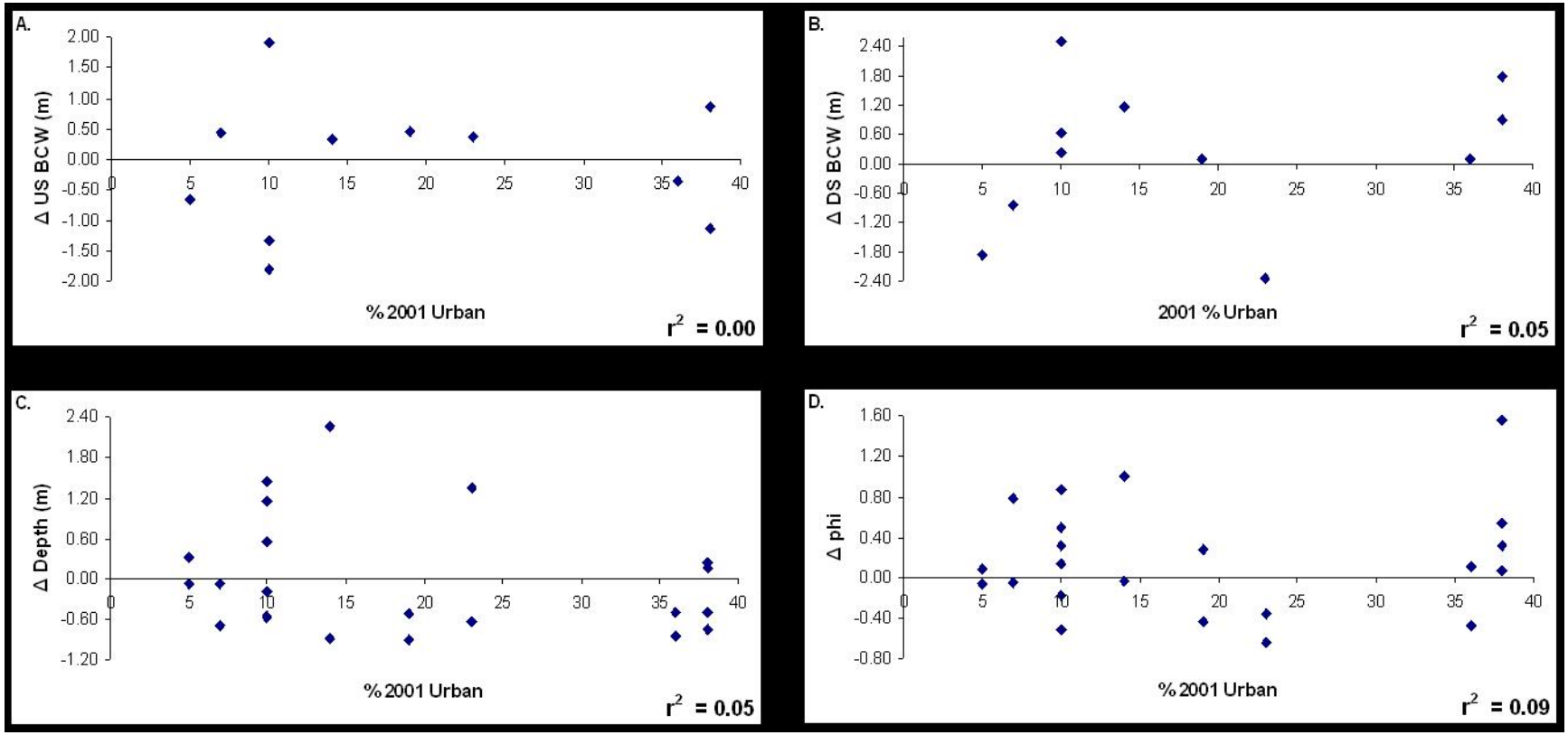

Fig. 2.14. Delta (Δ) values for A.) upstream basal channel width (US BCW), B.) downstream basal channel width (DS BCW), C.) thalweg depth (Depth), and D.) phi versus percent urban land use in 2001 (% 2001 Urban)

CHAPTER 3

APPLICATION OF A MULTI-STATE MODEL TO ESTIMATE CULVERT EFFECTS ON MOVEMENT OF SMALL STREAM FISHES

Introduction

Rivers and streams are longitudinally connected ecosystems that are particularly susceptible to fragmentation (Allan 1995, Pringle 1997, Ward 1998). With more than 6.4 million km of roadways across the United States (NRC 2005) and the high frequency with which roads cross streams, there is increasing recognition that road-stream crossings can affect the connectedness of stream ecosystems. Mounting evidence shows that road-stream crossings, in particular traditional pipe and box culverts, can indeed fragment stream ecosystems thereby limiting passage and the dispersal ability of aquatic organisms such as mussels (Voelz et al. 1998), aquatic insects (Blakely et al. 2006), shrimps (Resh 2005), crayfishes (Light 2003, Kerby et al. 2005), and, in particular, fishes (Belford and Gould 1989, Warren and Pardew 1998; Toepfer et al. 1999; Schaefer et al. 2003; Gibson et al. 2005).

Stream fishes have a variety of home range sizes and exhibit varying degrees of movement throughout the year (Freeman 1995, Gowan and Fausch 1996, Goforth and Foltz 1998, Larson et al. 2002, Albanese et al. 2003, Albanese et al. 2004, Schmetterling and McEvoy 2004, Schrank and Rahel 2004). The ability of fish to move freely throughout stream networks has important consequences for the completion of life history cycles of many species as well as the long term maintenance of populations and ecosystem function (Schlosser and Angermeier 1995, Johnston 2000, Fausch et al. 2002, Jackson 2003). Culverts that serve as barriers to fish movement can cause fragmentation of fish habitat and populations, promoting the loss of genomic heterogeneity and increasing the likelihood of local extinctions (Slatkin 1987, Warren and Pardew 1998, Jackson 2003, Wofford et al. 2005). Blocked passage may prevent fish from recolonizing areas where extirpation has occurred (Meffe and Sheldon 1990, Utzinger et al. 1998), alter nutrient cycling and predator-prey dynamics (Power et al. 1985, Jackson 2003), and

prevent access to habitat needed for spawning (Fausch and Young 1995, Trombulak and Frissell 2000, Wheeler et al. 2005), foraging (Clapp et al. 1990), thermal refugia (Schaefer et al. 2003), and for avoidance of areas of poor water and habitat quality (Bergstedt and Bergersen 1997).

There are a number of characteristics of culverts that may reduce or prevent fish passage. These factors include (Jackson 2003):

- Insufficient depth of water in hard bottom culverts;
- Discontinuity in streambed substrate which may inhibit movement of benthic fishes confined to the streambed;
- Excessive velocities (exceeding fish swimming capacities) inside culverts;
- Excessive turbulence at culvert inlets and outlets, and/or inside the culvert, created by flow constriction;
- Lack of bank-edge areas inside culverts, which can help reduce flow velocities;
- Debris blockage at culvert inlets or physical barriers such as weirs or baffles inside culverts; and,
- Inlet or outlet drops where the culvert is perched above the streambed or water surface.

While it is widely acknowledged that these culvert characteristics may pose problems for fish passage, the effects of culverts on restricting fish movement have not been extensively studied in the field, especially where the passage of non-game and small-bodied stream fishes are concerned. Warren and Pardew (1998) assessed movement for 21 fish species from seven families through pipe culvert, slab, open-box and ford crossings as well as through natural reaches in forested Arkansas streams. They reported that observed fish movement through crossings was bidirectional but an order of magnitude lower through pipe culverts than other

crossings or natural reaches, while open-box and ford crossings showed little difference from natural reaches in overall fish movement. Using live-resight data from snorkel and SCUBA surveys, Schafer et al. (2003) found that upstream movement of the federally Threatened leopard darter, *Percina pantherina*, appeared to be inhibited by a low-water pipe culvert crossing in a southeastern Oklahoma river. More recently, Coffman (2005) used mark-recapture data to help validate and modify predictive models for upstream passage through culverts by species from the families Salmonidae, Cyprinidae, Percidae and Cottidae. Also, Ensign (Kennesaw State University) recorded low rates of movements by small fishes through culverts compared to reaches with clear-span bridges over a 1-month period on six streams in the Etowah River system in northwest Georgia (reported in Norman et al. 2006).

While these studies have provided insight into the effects of road-stream crossings on the movement of small stream fishes, none have accounted for varying capture probability in their estimates of fish movement. If capture (or resight) probability is not constant, any estimates for rates or probabilities of movement may be biased. For example, consider an estimate of movement rates from one stream reach, A, to a second reach, B, based on the proportion of recaptured individuals (originally marked and released in A) observed in A and B. If a fish are recaptured in A and b fish are recaptured in B, then a commonly used estimate of proportional movement from A to B is $b/(a+b)$. However, for a fish to be recaptured in A or B, it must (1) remain in the study area, with probability S, (2) either move to B (with probability ψ) or remain in A (probability $1-\psi$), and (3) be recaptured (with probability p_a in A or probability p_b in B). Proportional movement can be thus expressed as:

$$b/(a+b) = S\,\psi\,p_b / [(S\,(1-\psi)\;p_a) + (S\,\psi\,p_b)]$$

$$= \psi\,[p_b/((1-\psi)\,p_a + \psi\,p_b)]$$

Thus, the observed proportional movement is an unbiased estimate of ψ, movement probability, if and only if $p_a = p_b$. However, capture probabilities are expected to vary with variation in fish behavior (Larimore 1961) and stream habitats (Peterson et al. 2005), and thus any unbiased estimation of movement probabilities should incorporate capture probability.

We build on the work of these previous studies, and recent work by Bill Ensign and colleagues (Kennesaw State University) in the Etowah, by using multi-state models that incorporate capture probabilities in the estimation of transition probabilities for fish movement. We conducted open population mark-recapture studies using resident fish assemblages in streams with culverts expected either to prevent or allow fish passage, and used a series of multi-state models to estimate transition probabilities for upstream and downstream movement of benthic and water column fishes between stream reaches, both through and not through culverts, to test the following hypotheses:

1. Fish are more likely to move short distances than long distances, and are less likely to move in the presence of a culvert.
2. Fish are more likely to move through embedded culverts than through culverts perched above the streambed or water surface.
3. Water column fishes are more likely to move and move through culverts than benthic fishes.

Our results provide further insight into the effects of culverts on the movement of non-game and small stream fishes, and demonstrate the application of multi-state models in mark-recapture and fish movement studies.

Methods

Study Sites and Site Selection

Study sites were located within the upper Etowah River Basin, GA (Fig. 3.1). The Etowah River Basin lies on the northern edge of the Atlanta metropolitan area and drains 4823 km^2 in parts of eleven counties in northwest Georgia. The Etowah River basin has been the focus of habitat conservation planning for nine imperiled fish species (www.etowahhcp.org), including the Cherokee darter, *Etheostoma scotti*, a federally Threatened species native to Etowah river tributaries. The Etowah River Basin has relatively high fish species diversity and endemism, harboring 76 extant native species that include four endemic species and seventeen imperiled species (Burkhead et al. 1997), many of which are small bodied fishes with limited ranges that could be detrimentally affected by blocked fish passage at road-stream crossings.

To select sites for our study, we began with a set of 610 candidate road-stream crossing sites in the highest conservation priority areas in the Etowah River Basin with watershed sizes less than 52 km^2 (Fig. 3.1). Road-stream crossing sites with watershed sizes greater than 52 km^2 were not considered because, in Georgia, streams draining an area greater than 52 km^2 are typically bridged. Watershed size was calculated using digital U.S. Geological Survey (USGS) topographic maps in ArcView® 3.3 geographic information systems (GIS) software. We used a combination of GIS analysis, fish survey databases, road-stream crossing surveys and site visits to select four sites with relatively high fish abundance and comparable habitat that extended upstream and downstream of the crossing with no nearby tributaries or extra potential barriers to fish passage, and where we could set block nets and sample effectively. We selected sites on Canton and Bluff Creeks – both of which were RPM sites (Sites 1 and 6 in Chapter 2) – in Cherokee County, Champion Creek in Pickens County, and Nimblewill Creek in Lumpkin

County. We conducted mark-recapture studies at each of these sites to estimate fish movement rates between reaches that were and were not separated by the culvert. The sites had watershed sizes that ranged from 4.09 km^2 to 28.06 km^2 and physical and water quality characteristics of the sites were similar (Table 3.1). Descriptions of the road-stream crossings and culvert characteristics for each site are shown in Table 3.2.

Open Population Mark-Recapture Study

To measure fish movement, we conducted open population mark-recapture studies at each of our four sites during August and September 2006. Upstream and downstream fish movement was assessed over four weeks at Canton, Bluff and Champion Creeks, and over six weeks at Nimblewill Creek. Canton Creek was sampled five times (four marking and four recapture occasions, with the first and last visits being marking-only and recapture-only, respectively, and the middle three visits involving mark and recapture). The first mark and mark-recapture occasions at Canton Creek occurred over 24 hours to ensure that we could capture and recapture an adequate number of fish for analysis. After confirming our capture and recapture rates were sufficient, we sampled Canton Creek three more times at weekly intervals. Because we were confident that our rates of capture and recapture would be sufficient at the remaining sites, we planned to sample Bluff, Champion and Nimblewill Creeks four times (three marking and three recapture occasions) at weekly intervals. We were able to sample Bluff and Champion Creeks weekly, but due to rain events, we were unable to sample Nimblewill Creek on a weekly basis. As a result, time between sampling occasions at Nimblewill Creek was fourteen days, eleven days and nine days on return visits.

Each site was divided into ten reaches with five reaches upstream and five downstream of the culvert. Moving upstream and downstream from the culvert, we designated three 25 m (+/- 3 m) reaches which were separated by two 'false' culvert reaches. The false culvert reaches were approximately equal to the length of the culvert at each site (range 17.6 m – 21 m) (Fig. 3.2a). Three weeks prior to the first sampling occasion, the individual reaches were measured and flagged at each site so that the length and position of each reach would remain constant throughout the study.

Fish sampling started below the culvert in the most downstream reach and proceeded upstream on each visit. Prior to fish sampling, all five downstream reaches and the culvert inlet were blocked off using block nets (3.2 mm mesh, 2 m to 2.4 m deep by 6 m to 20 m long) (Fig. 3.2b). Once we completed sampling of the downstream reaches, all block the nets were removed and set in place at the ends of the upstream reaches before fish sampling upstream of the culvert began. In each reach, we sampled fish in a single pass using a Smith Root model 12-B POW backpack electrofisher and kick-sets with a 2.4 m by 2 m seine (3.2 mm mesh) (Fig. 3.2c). We recorded stream temperature, conductivity and turbidity on each sampling occasion using a Hydrolab and a Hach turbidity meter.

All fish collected were anesthetized using a 30 mg/L dilution of Aqui-S® (Aqui-S New Zealand, Ltd), identified to species, and measured for standard length to the nearest millimeter. With the exception of *Campostoma oligolepis* and *Hypenteilum etowanum*, fish ≥30 mm and ≤120 mm standard length (SL) from each individual reach were given a mark using visible implant elastomer (VIE) tags (Northwest Marine Technology, Inc). Federally threatened *E. scotti* were marked using VIE tags with permission under federal permit number SA 02-11. VIE tags were administered to indicate both marking occasion and the reach where the fish was

located when it was captured. Marking occasion was indicated by the color used (yellow, orange, pink or blue), and position in the stream was indicated by the body position used for marking (Table 3.3). VIE tags have been shown to have no significant effect on the growth or survival rates of fishes (Olsen and Vollestad 2001, Roberts and Angermeier 2004, Astorga et al. 2005), although tag retention and the ability to correctly identify tags can vary depending upon the species, marking position on the body, and color combinations used (Frederick 1997, Close and Jones 2002, Goldsmith et al. 2003, Walsh and Winkelman 2004, Brennan et al. 2005, Hartman and Janney 2006). We avoided using combinations of colors that have been reported as difficult to separately identify (Curtis 2006, Northwest Marine Technology 2006).

We found that VIE tags were not easily identifiable at some body positions and with some colors on *C. oligolepis* and *H. etowanum*, so they were given unique partial fin clips on the caudal, pectoral and dorsal fins on each sampling occasion to indicate marking occasion and position in the stream at time of marking (Table 3.4). Because of the lack of unique fin clip combinations, we were only able to distinguish between *C. oligolepis* and *H. etowanum* captured either upstream or downstream of the culvert on each sampling occasion. Caudal and pectoral fin clips have been used in previous mark-recapture studies without significant bias (Riley et al. 1992, Hohausova 2000, Dietrich and Cunjak 2006, Lukey et al. 2006) and have been shown to have no significant effect on fish swimming performance (Webb 1973, Webb 1977, Parsons et al. 2003).

Before being released, we held fish in stream water containing a standard dilution (0.1mg/L) of Kayamycin Sulfate Powder (National Fish Pharmaceuticals®) to aid recovery from the anesthesia and to help prevent infection of the VIE injection site. All fish were released in the middle of the reach in which they were captured on each occasion. Once we released all the

fish, they were free to move among reaches and through the culvert until the reaches were blocked at the start of the next sampling occasion.

Modeling fish movement and culvert effects on transition probabilities

We used the Arnason-Schwartz multi-state model (Arnason 1973, Schwartz et al. 1993) as implemented in Program MARK (White and Burnham 1999) to estimate transition probabilities for fish movement between and among stream reaches as well as through culverts at our study sites. The Arnason-Schwartz multi-state model is an extension of Cormack-Jolly-Seber single state models (Cormack 1964, Jolly 1965, Seber 1965) used to model survival and capture probabilities, and it permits stochastic transitions among states, such as animals moving stochastically from one location to another (Williams et al. 2002). Multi-state and single-state population models both use maximum likelihood methods to estimate survival and recapture probabilities. However, in multi-state models, survival probabilities (φ_i^{rs}) incorporate the probabilities of transition from one state or, for our purposes, location to another, and capture probabilities (p_i^r) are state or location specific (Williams et al. 2002).

As defined by Williams et al. (2002), the parameter φ_i^{rs} is the probability of being alive and in location *s* at time $i + 1$ for a marked animal alive in location *r* at time *i*, and p_i^r is the probability that an animal in location *r* at time *i* is captured or observed. Under the assumption that survival from time *i* to time $i + 1$ does not depend on the location at time $i + 1$, then survival probabilities and transition probabilities can be estimated separately. Therefore, φ_i^{rs} can be written as $S_i^r \psi_i^{rs}$, where: S_i^r is the probability that an animal in location *r* at sampling period *i* survives and remains in the sampling area until $i + 1$; and ψ_i^{rs} is the probability that an animal in

location *r* at time *i* is in location *s* at time *i* + 1, given that the animal is alive at *i* + 1 (Williams et al. 2002, Cooch and White 2006).

For our study, we assumed that survival from time *i* to time *i* + 1 did not depend on the, location in the stream at time *i* + 1. Therefore, we were able to estimate survival and transition probabilities separately as well as reach specific capture probabilities. Because a fully saturated ten-state model (i.e., all parameters being time-dependent, with reach- specific survival and recapture probabilities and all possible combinations of transition probabilities estimated separately) would have generated between 660 and 880 parameters, we were forced to start the modeling process with a series of reduced, time-independent models (i.e., survival, capture and transition probabilities held constant across all sampling dates) that were chosen *a priori* to test a set of biologically reasonable hypotheses about fish movement among and between reaches and through culverts. We developed two sets of candidate models: one set of site-specific models with four possible general models and one constrained model, and one set of models that grouped all sites together with four possible general models and two constrained models. Since our models were not fully time-dependent (i.e., all parameters being estimated separately for each sampling occasion), we were unable to use goodness-of-fit (GOF) tests to evaluate lack of fit in our models, as GOF tests are only available for Arnason-Schwartz multi-state models that are fully time-dependent (Pradel et al. 2003, Cooch and White 2006). Nevertheless, none of the ranking orders (based on QAICc values) for models in our candidate model sets were very sensitive to incremental adjustments in the variance inflation factor, $\hat{c}$ (Anderson et al. 1998), indicating that our models likely fit the data relatively well (Cooch and White 2006).

For the site-specific models, reach-dependent capture probabilities were included in all candidate models. Individual fish were categorized as either benthic or water column species,

and parameters were estimated for each group separately. We began building our general models by first assembling our global model, hereafter referred to as the *Full Movement* (FM) *Grouped* model (Table 3.5, Fig. 3.3), which treated upstream and downstream movement transition probabilities separately, and combined transition probabilities for culvert-free movement into three distance groupings: movement to adjacent reaches, movement approximately equal to one culvert length, and movement >1 culvert length. Transition probabilities for movement upstream and downstream through a culvert were combined into two distance groupings: movement through a culvert and movement through a culvert + ≥1 culvert length. The competing *Reduced Movement* (RM) *Grouped* model (Table 3.5, Fig. 3.4) treated upstream and downstream movement transition probabilities separately and combined transition probabilities for culvert-free movement into two distance groupings: movement to adjacent reaches and movement ≥1 culvert length. Transition probabilities for movement upstream and downstream through a culvert were combined into a single distance grouping: movement through a culvert plus movement through a culvert + ≥1 culvert length. Survival probabilities in the Arnason-Schwartz model do not separate mortality from emigration, and because it is reasonable to assume that fishes closer to edges of the study site would be more likely to move out of the study site, survival estimates in our FM and RM models were paired by upstream and downstream reaches that were equidistant from the ends of our study sites (Fig. 3.5). *Full Movement Grouped with Constant Survival* (FMCS) and *Reduced Movement Grouped with Constant Survival* (RMCS) models were also assembled (Table 3.5). In FMCS and RMCS models, transition probability combinations were identical to those in the FM and RM models, and survival (i.e., survival and emigration) was treated as constant amongst all reaches (Fig. 3.6).

We selected the best supported general model by pair-wise comparisons of model parsimony using Akaike Information Criteria corrected for small sample size (AICc) (Akaike 1974, Burnham and Anderson 2002) (Table 3.5). FM and RM model parsimony were compared, and the best supported model was selected for comparison with the corresponding constant survival model (FMCS or RMCS). The best supported general model from the candidate set was then chosen and compared with a constrained model with the same model structure except transition probabilities were constrained as a function of individual standard length to test for an effect of fish length on probability of movement.

The candidate set of general models for the combined sites consisted of models with the same model structures as the site-specific models (Table 3.6), but observed capture histories were grouped by site instead of by benthic and water column fishes. Thus, the combined-site models estimated parameters separately for each site but treated all fish at a site as a single group. We used the same model selection process for the combined-site models as we used for the site-specific model selection. In the constrained models, we used the minimum height a culvert was perched above the water surface (*min PAWS*) as a covariate with movement upstream through a culvert. In addition, two of the culverts had concrete aprons that spanned the stream bottom downstream from the culvert, with an elevated lip at the downstream end of the apron. Thus, we used the height of the apron lip above the culvert bottom (*Apron*) as a covariate with movement downstream through a culvert. The most general constrained model contained both covariates while the reduced competing model contained only the *min PAWS* covariate. Movement of *C. oligolepis* and *H. etowanum* was modeled according to the same process and using the same sets of candidate models described above, but *C. oligolepis* and *H. etowanum* models only consisted of two states (i.e., upstream and downstream of the culvert), as opposed to

ten, and survival probabilities for the upstream and downstream reaches were estimated separately in the FM and RM models and as one parameter in the FMCS and RMCS models.

Multi-state models with more than two states are known to be prone to converging to a local, rather than the global, maximum, which can lead to faulty parameter estimates (Lebreton and Pradel 2002, Cooch and White 2006). Because there is no known simple diagnostic for potential global optimization problems *a priori*, we used the alternative global optimization process, simulated annealing (Goffe et al. 1994), as implemented in Program MARK to find the global maxima for our multi-state models. The simulated annealing algorithm periodically makes a totally random jump to a new parameter value, which is what allows the algorithm more flexibility in finding the global maximum rather than a local maximum (Goffe et al. 1994, Cooch and White 2006).

We used transition probabilities from the best supported model at each site to calculate the 'effect size' of the culvert on movement of benthic and water column fishes at each site. Effect size was calculated by taking the arithmetic difference between the parameter estimates for movement through a culvert and culvert-free movement. Corresponding 95% confidence limits were also calculated for each effect size value in the following manner (Cooch and White 2006):

$$SE = \sqrt{var(\psi_{\text{culvert-free move}}) + var(\psi_{\text{move thru-culvert}}) - cov(\psi_{\text{culvert-free move,}}\ \psi_{\text{move thru-culvert}})}$$

$$95\%\ CI = (\text{effect size} \pm 2SE)$$

We were only able to calculate effect size where there were transition probabilities for both culvert-free movement and movement through a culvert in the same direction (upstream or downstream) and over the same distance grouping.

Maximum stage height and stage at time of sampling

We installed maximum stage recorders constructed of 7.6 cm-diameter PVC pipe and fine ground cork and coffee grinds (Gordon et al. 2004) at each site. On each visit the maximum stage that occurred between sampling occasions was recorded, as was the stage height at the time of fish sampling. We used the stage height at the time of sampling and the maximum stage height between sampling occasions to calculate the change in stage height for the time interval between each sampling occasion. We also calculated a value for *effective min PAWS* for the time interval between each sampling occasion by taking the difference between *min PAWS* (explained above) and the change in stage height for each time interval at each site. Because we were not able to use fully time-dependent multi-state modeling in our study, we were not able to explicitly model the effects of stage and *effective min PAWS* on recapture or transition probabilities. Rather, *min PAWS* was used as a covariate on transition probabilities in the constrained combined-site models (explained above).

Results

The sites varied in watershed size and stream width, but all were cool water streams with gravel-dominated beds and relatively low turbidity and conductivity (Table 3.1). Culvert characteristics were different at each site, although all culverts were similar in length (Table 3.2). A total of 5805 individual fish were marked at the four sites, representing 30 species (Table 3.7, Appendix B).

Overall recapture rates ranged between 0.29 and 0.37 (Table 3.8) with 33% of individuals caught across all sites being recaptured on more than one date. Recaptures mostly occurred in the same reaches where individuals were initially marked; 6% to 11% of recaptured individuals

moved among reaches, and more fish were observed to move upstream than downstream at all four sites (Table 3.9). Of the individuals that moved among reaches, 63% to 87% of moves were to adjacent reaches (Table 3.10). We observed at least one fish that moved through the culvert at each site. Only one fish was observed to move through the culvert at Canton Creek, but several taxa were found to move through the culvert at the other sites (Table 3.10).

The best supported site-specific models generally were the FM models, except for Canton Creek where the best supported model was the RMCS model. The best supported model for Bluff and Champion Creeks was the FMCS model, while the FM model was best supported for Nimblewill Creek (Table 3.11). There were problems with numerical convergence in the constrained model for Bluff Creek, so the model constrained by standard length was not included in model comparisons. However, adding standard length as a covariate did not improve model fit at the other three sites (Table 3.11).

Survival and recapture probability estimates were similar for benthic and water column fishes, and across sites, although reach-specific recapture probabilities varied by a factor of 2 to over 10 within sites and fish groups (Table 3.12, Appendix B). Transition probabilities (ψ) for movement were generally small ($0.0054 < \psi < 0.16$) with wide confidence intervals across the sites. At Canton Creek (Table 3.13), benthic fishes were most likely to move downstream to an adjacent reach ($\psi = 0.068$, [95% confidence interval: 0.025 – 0.17]) while water column fishes were most likely to move upstream to an adjacent reach ($\psi = 0.16$, [0.12 – 0.21]). The highest probability of movement for benthic fishes at Bluff Creek was downstream through the culvert ($\psi = 0.095$, [0.012 – 0.47]) whereas water column fishes were most likely to move to adjacent reaches, with similar transition probabilities for upstream ($\psi = 0.13$, [0.087 – 0.19]) and downstream ($\psi = 0.14$, [0.091 – 0.21]) movement (Table 3.14). Transition probabilities at

Champion Creek were highest for upstream movement to adjacent reaches for benthic fishes (ψ = 0.083, [0.046 – 0.15]) and for downstream movement to adjacent reaches for water column fishes (ψ = 0.14, [0.099 – 0.19]) (Table 3.14). Both benthic and water column fishes were most likely to move upstream to adjacent reaches (ψ = 0.097, [0.058 – 0.16] and ψ = 0.081, [0.056 – 0.11], respectively) at Nimblewill Creek (Table 3.14).

Where we observed movements through the culvert and between reaches not separated by the culvert for the same distance and direction (i.e., upstream or downstream) categories, the estimated effect size of the culverts was either small or negative, with a confidence interval that included no effect (0). At Canton Creek, benthic fishes were about 1.4 times more likely to move upstream over ≥1 culvert length (CL) than through the culvert + ≥1 CL, although confidence intervals on all estimates were wide (Tables 3.13 and 3.15). Benthic fishes at Bluff Creek were 3.7 times more likely to move downstream through the culvert than move ≈1 CL where there was no culvert (Table 3.14). There was no significant culvert effect on water column fishes moving upstream through the culvert + ≥1 CL at Bluff Creek, and transition probabilities for movement upstream through the culvert were about 4.2 times greater than moving the same distance where no culvert was present (Tables 3.14 and 3.15). At Champion Creek, water column fishes were about 7 times more likely to move upstream through the culvert than ≈1 CL where there was no culvert (Table 3.14). However, water column fishes were about 4.7 times more likely to move upstream >1 CL where there was no culvert than move through the culvert + ≥1 CL (Table 3.14). The transition probability for water column fishes moving downstream through the culvert + ≥1 CL at Nimblewill Creek was similar to the transition probability for water column fishes moving >1 CL where there was no culvert (Table 3.14).

Fish movements through the culverts at the sites with perched culverts (Champion and Nimblewill Creeks) were only recorded following runoff events that raised the water level over the perched level. Stage height increased at all sites during the time intervals between each sampling occasion (Table 3.16). We recaptured fishes that had moved through the culverts at Canton, Champion and Nimblewill Creeks only after the time interval where the increase in stage height was greatest and *effective min PAWS* was the least: interval 3 at Canton (which was not perched) and Champion Creeks and interval 2 at Nimblewill Creek (Table 3.16). Fishes were observed to move through the culvert at Bluff Creek during each time interval (Table 3.16).

For the combined-site models where sites were treated as groups, the FM model and the FM model constrained by *min PAWS* ($FM^{min\ PAWS}$) had identical support, and were therefore indistinguishable in terms of fit (Table 3.17). Transition probability estimates from the FM and $FM^{min\ PAWS}$ models were identical to within six significant digits. The FM and $FM^{min\ PAWS}$ model had approximately 2.8 times the support of the next best fitting model, the FM model constrained by both *min PAWS* and *Apron* ($FM^{min\ PAWS\ +\ Apron}$) (Table 3.17). The transition probability estimates from the FM and $FM^{min\ PAWS}$ models were intermediate with estimates of transition probabilities for benthic and water column fishes from the site-specific models, although the estimates tended to be skewed more towards the estimates for water column fishes (Appendix B).

We observed *C. oligolepis* and *H. etowanum* movement at Bluff Creek only, and transition probabilities through the culvert were low. *C. oligolepis* was only observed to move upstream through the culvert (n = 1; ψ = 0.011, SE(0.013), [0.0010 – 0.11]), while *H. etowanum* was observed to move both upstream (n = 1; ψ = 0.010, SE(0.010), [0.0014 – 0.070]) and downstream (n = 2; ψ = 0.018, SE(0.013), [0.0044 – 0.073]) through the culvert. The best

supported model for *C. oligolepis* and *H. etowanum* movement at Bluff Creek was the FM model (AICc weight = 0.787). *C. oligolepis* recapture rate was also low, 0.14, with an estimated capture probability (downstream only) of 0.078. Upstream and downstream survival probabilities were 0.96 and 0.077 for *C. oligolepis*. *H. etowanum* recapture rate was higher, 0.43, with estimated upstream and downstream capture probabilities of 0.45 and 0.34, respectively. Survival probability estimates were 0.75 upstream and 0.74 downstream for *H. etowanum.*

Discussion

Fish movement

The recaptured fish at all four sites were not highly mobile and tended to remain in the same reach or move to adjacent reaches 96% to 99% of the time. This result is similar to the findings of other studies on movement by small stream fishes (e.g. Hill and Grossman 1987, Johnston 2000, Schafer et al. 2003). However because we did not sample to detect long range movement, we cannot not make any conclusions about the movement patterns of the 63% to 71% of marked individuals not recaptured during our study (Albanese et al. 2003). We documented fish movement in the upstream and downstream directions, with more fish moving upstream than downstream at all sites. We also observed fish movement through the culvert at each study site. However, movement through the culvert only appeared unrestricted for benthic and water column fishes at the site with a bottomless box culvert (Bluff Creek). Only water-column fishes were observed to move through the two perched culverts (Champion and Nimblewill Creeks), and only following periods of runoff when water levels exceeded the perched level.

Contrary to our hypotheses, we observed the fewest number of fish move through the culvert at Canton Creek, an embedded pipe culvert constructed according to FWS RPM terms

and conditions. Only one *Percina kathae*, a benthic darter species, moved upstream through the culvert at Canton Creek. This result is not surprising for a couple of reasons. First, movement of small stream fishes through pipe culverts has been found to be an order of magnitude less than movement through box culverts, fords or natural reaches (Warren and Pardew 1998). Second, there was a small debris dam blocking part of the culvert inlet which may have deterred fishes from moving through the culvert at Canton Creek. Bluff Creek (with a bottomless box culvert also constructed according to RPM terms and conditions) was the only site where we observed both upstream and downstream movement through the culvert by both water column and benthic fishes, including the federally Threatened Cherokee darter, *Etheostoma scotti*. Including *C. oligolepis* and *H. etowanum*, about the same number of fish were observed to move upstream (n = 11) and downstream (n = 10) through the culvert. At Champion and Nimblewill Creeks, only water column fishes were observed to move through the culvert, either upstream (n = 7, at Champion only) or downstream (n = 5, at Nimblewill only).

Moves through the culvert composed 24% of the total moves observed at Bluff Creek, and only 1%, 8% and 4% at Canton, Champion and Nimblewill Creeks, respectively. Bluff Creek was the only site where the culvert appeared to have no impact on fish movement, as fishes passed upstream and/or downstream of the culvert during each time interval between sampling occasions. The crossings at Canton and Bluff Creeks were the only crossings through which benthic fishes were observed to move, while the crossings at Champion and Nimblewill Creeks appeared to be barriers to both the upstream and downstream movement of benthic fishes. Culverts at Champion and Nimblewill Creeks also seemed to serve as barriers to passage by water column fishes until stage height increased sufficiently to submerge the culvert outlets, suggesting temporal variation in stage height may play a significant role in determining the

ability of fishes to pass through crossings where culverts are perched above the water surface. Given the degree to which the crossing at Nimblewill Creek was perched above the water surface, and because we observed no fish moves upstream through the crossing, it is likely that under most conditions, the crossing at Nimblewill Creek is a barrier to upstream fish passage.

Application of multi-state models

Estimates for transition probabilities from our models had wide confidence intervals for most estimates as a result of the low number of observed fish moves at our study sites. Despite lack of parameter estimate precision, use of multi-state models allowed us to quantify transition probabilities for movement by small bodied fishes both through and not through culverts while incorporating reach and group specific (benthic vs. water column fishes) capture probabilities. Being able to account for group and reach specific capture probability is important because it is expected that rates of capture for benthic and water column fishes will vary with group (e.g., because of behavioral and habitat differences; Larimore 1961, Reynolds 1983) and in-stream habitat heterogeneity (Peterson et al. 2005). Furthermore, multi-state models do not require a separate estimation of capture probability, which has logistical benefits (e.g., it's not necessary to leave block nets in the stream for extended periods) and biological benefits for the fish (i.e., more time is allowed between sampling occasions, reducing sampling- and handling-induced stress). In addition to explicitly modeling capture probability, the use of multi-state models also allowed us to include individual standard length and culvert characteristics (*min PAWS* and *Apron*) as covariates on movement probabilities. We saw no apparent effect of standard length on fish movement, but length has been shown to be a correlate of movement for some fishes, although the effect is not pervasive (Gatz and Adams 1994, Albanese et al. 2004).

As suggested above, the ability of fish to pass through crossings at our study sites likely varied with changes in stage height through time. Because we designed our study to maximize the potential for observing individual movement among reaches (i.e., by sampling 10 reaches at each site), time-dependent models were not feasible given the hundreds of possible parameters fully time-dependent models would have generated. Had we been able to include time-dependence in our models, we would have gained the ability to explicitly model changes in stage height as a covariate on movement probabilities to test whether or not the fish passage through culverts varied through time as a function of stage height (or *effective min PAWS*). Since we were unable to include time-dependence in our models, probabilities of movement were in effect averaged across all time intervals. Therefore, it was not surprising that *min PAWS* and *Apron* did not improve model fit or change estimates for movement probabilities through the culverts at Champion or Nimblewill Creeks.

Implications

Our results indicate that multiple-barrel box culverts with bottomless middle barrels and continuous streambed substrate throughout should provide for fish passage comparable to that of natural stream reaches. As far as multiple-barrel embedded pipe and perched box culverts are concerned, our results were less conclusive in terms of the effects on fish passage. While it appeared that embedded multiple-barrel pipe culverts do not prevent fish passage, fish passage through pipe culverts may be reduced compared to natural reaches, especially if debris barriers form at the inlet of one or more pipes. Multiple-barrel box culverts that are perched above the water surface may not be complete barriers to fish passage, but likely serve as barriers to the passage of benthic fishes more so than water column fishes. It is likely that benthic and water

column fishes are more likely to be able to move downstream than upstream through culverts once the culvert outlet becomes perched above the streambed or the water surface. We suspect the more perched a culvert outlet becomes above the water surface, the more likely that culvert is to prohibit upstream passage by benthic fishes over any range of flows, especially where streambed substrate is not present inside the culvert. Water column fishes may still be able to move upstream and/or downstream through perched box culverts, but not with constant success or at a constant rate. Rather, the ability to pass upstream will likely depend on the degree to which the culvert is perched above the water surface, which is expected to vary temporally with changes in stage height. Furthermore, the rate at which fishes pass upstream through perched box culverts (and probably any other type of perched culvert) when stage height increases sufficiently to submerge the culvert outlet will likely depend on the ability (and tendency) of fishes to swim upstream against the increased water velocities and discharge associated with increases in stage height.

More work needs to be done in order to fully understand and quantify the effects of different road-stream crossing types on fish passage and habitat and population fragmentation, and how those effects vary among benthic and water column species and along ecological gradients. Use of mark-recapture studies and multi-state models provides a promising framework for future research into the effects of road-stream crossings on fish passage and population fragmentation. Future studies should be conducted over longer time spans using fewer reaches to maximize the potential for observing variation in passage ability through time and to minimize the number of states involved in the modeling process, thereby not precluding the inclusion of time-dependence and time-varying covariates. Modeling time-dependent fish movement as a function of variables (both time-varying and constant) predicted to determine the

ability of fishes to pass through crossings and/or across natural reaches (e.g. changes in stage height, *effective min PAWS*, storm flow velocities, habitat and substrate characteristics) could help us begin quantifying the extent to which road-stream crossings actually reduce or prevent passage and cause fragmentation. Moving beyond hypotheses and cautionary predictions to systematically quantifying crossing effects could help improve efforts to maintain habitat and population connectivity for stream fishes, as well as other aquatic fauna potentially affected by road-stream crossings.

Given the pervasiveness of roadways around the globe, with more than 6.4 million miles of roadways in the U.S. alone (NRC 2005), road-stream crossings have the potential to affect the connectedness of stream habitat and fish populations over extensive areas. A more thoroughly quantified understanding of the effects of road-stream crossings on fish passage and fragmentation is important if informed and biologically sound decisions are to be made about the replacement of existing road-stream crossings as well as the design and construction of new crossings. This becomes especially important in areas where resources are limited and decisions must be made about how to invest in the replacement of existing 'problem' crossings and/or the design of crossings that are less likely to fragment streams and prevent fish passage in the future. Because crossing replacement can have unforeseen hydrologic and geomorphic effects and requires careful planning (RSCP 2006), and because the design of crossings that simulate stream conditions can be more expensive than traditional crossings, quantified information about the effectiveness of different crossing types is crucial to ensuring that crossings are replaced and designed to obtain maximum biological benefit without wasting economic and ecological resources.

Acknowledgements

Mary Freeman, Megan Hagler and Bud Freeman were unbelievably helpful with study design, field work, and data analysis and interpretation, and have my sincere gratitude. Jim Peterson also gave valuable insight and advice on study design and data analysis, and helped provide field equipment necessary to complete this study. Megan Hagler, Mary Freeman, Jeffery Garnett, Christina Baker, Nicole Pontzer, Rachel Katz, Jane Argentina, Allison Price, Carrie Straight and Rebecca Coleman assisted in field work, and deserve special thanks. We wish to thank Judy Meyer and David Leigh for suggestions that improved this work. Funding was provided by the U.S. Fish and Wildlife Service to Byron J. Freeman, through Research Work Order 79, USGS GA Cooperative Fish and Wildlife Research Unit.

Literature Cited

Akaike, H. 1974. A new look at the statistical model identification. IEEE Transactions on Automatic Control 19: 716-723.

Albanese, B., P.L. Angermeier, and S. Dorai-Raj. 2004. Ecological correlates of fish movement in a network of Virginia streams. Canadian Journal of Fisheries and Aquatic Sciences 61: 857-869.

Albanese, B., P. L. Angermeier, and C. Gowan. 2003. Designing mark-recapture studies to reduce effects of distance weighting on movement distance distributions of stream fishes. Transactions of the American Fisheries Society 132: 925-939.

Allan, J.D. 1995. Stream ecology: structure and function of running waters. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Anderson, D.R., K.P. Burnham, and G.C. White. 1998. Comparisons of Akaike Information Criterion for model selection and statistical inference from capture-recapture studies. Journal of Applied Statistics 25: 263-282.

Arnason, A.N. 1973. The estimation of population size, migration rates and survival in a stratified population. Researches on Population Ecology 15: 1-8.

Astorga, N., J.M. Afonso, M.J. Zamorano, D. Montero, V. Oliva, H. Fernandez, and M.S. Izquierdo. 2005. Evaluation of visible implant elastomer tags for tagging juvenile gilthead seabream (*Sparus auratus* L.); effects on growth, mortality, handling time and tag loss. Aquaculture Research 36: 733-738.

Belford, D.A., and W.R. Gould. 1989. An evaluation of trout passage through six highway culverts in Montana. North American Journal of Fisheries Management 9: 437-445.

Bergstedt, L.C., and E.C. Bergersen. 1997. Health and movements of fish in response to sediment sluicing in the Wind River, Wyoming. Canadian Journal of Fisheries and Aquatic Sciences 54: 312-319.

Blakely, T.J., J.S. Harding, A.R. McIntosh, and M.J. Winterbourn. 2006. Barriers to recovery of aquatic insect communities in urban stream. Freshwater Biology 51: 1634-1645.

Brennan, N.P., K.M. Leber, H.L. Blankenship, J.M. Ransier, and R. DeBruler. 2005. An evaluation of coded wire and elastomer tag performance in juvenile common snook under field and laboratory conditions. North American Journal of Fisheries Management 25: 437-445.

Burkhead, N.M, S.J. Walsh, B.J. Freeman, and J.D. Williams. 1997. Status and restoration of the Etowah River, and imperiled southern Appalachian river ecosystem. Pages 375-344 *in* G.A. Benz and D.E. Collins , editors. Aqautic fauna in peril: the southern perspective. Lenz Design & Communications, Decatur, GA.

Burnham, K.P., and D.R. Anderson. 2002. Model selection and multimode inference: A practical-theoretic approach, 2nd ed. Springer-Verlag, New York, NY.

Clapp, D.F., R.D. Clark, Jr., and J.S. Diana. 1990. Range, activity, and habitat of large, free-ranging brown trout in a Michigan stream. Transactions of the American Fisheries Society 119: 1022-1034.

Close, T.L., and T.S. Jones. 2002. Detection of visible implant elastomer in fingerling and yearling rainbow trout. North American Journal of Fisheries Management 22: 961-964.

Coffman, J.S. 2005. Evaluation of a predictive model for upstream fish passage through culverts. M.S. Thesis, James Madison University, Harrisonburg, VA.

Cooch, E., and G.C. White. 2006. Program MARK: a gentle introduction, 5th ed. http://www.phidot.org/software/mark/docs/book/

Cormack, R.M. 1964. Estimates of survival from sighting of marked animals. Biometrika 51: 429-438.

Curtis, J.M.R. 2006. Visible implant elastomer color determination, tag visibility and tag loss: potential sources of error for mark-recapture studies. North American Journal of Fisheries Management 26: 327-333.

Dietrich, J.P., and R.A. Cunjak. 2006. Evaluation of the impacts of Carlin tags, fin clips, and Panjet tattoos in juvenile Atlantic salmon. North American Journal of Fisheries Management 26: 163-169.

Fausch, K.D., and M.K. Young. 1995. Evolutionary significant units and movements of resident stream fishes: a cautionary tale. Pages 360-370 *in* J.L. Nielson, editor. Evolution and the Aquatic System: Defining Unique Units in Population Conservation. American Fisheries Society Symposium 17.

Fausch, K.D., C.E. Torgersen, C.V. Baker, and H.W. Li. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. BioScience 52: 483-498.

Frederick, J.L. 1997. Evaluation of fluorescent elastomer injection as a method for marking small fish. Bulletin of Marine Science 61: 399-408.

Gatz, A.J., Jr., and S.M. Adams. 1994. Patterns of movement of centrarchids in two warmwater streams in eastern Tennessee. Ecology of Freshwater Fishes 3: 35-48.

Gibson, R.J., R.L. Haedrich, C.M. Wernerheim. 2005. Loss of fish habitat as a consequence of inappropriately constructed stream crossings. Fisheries 30: 10-17.

Goffe, W.L., G.D. Ferrier, and J. Rogers. 1994. Global optimization of statistical functions with simulated annealing. Journal of Econometrics 60: 65-99.

Goforth, R.R., and J.W. Foltz. 1998. Movements of yellowfin shiner, *Notropis lutipinnis*. Ecology of Freshwater Fish 7: 49-55.

Goldsmith, R.J., G.P. Closs, and H. Steen. 2003. Evaluation of visible implant elastomer for individual marking of small perch and common bully. Journal of Fish Biology 63: 631-636.

Gordon, N.D., T.A. McMahon, B.L. Finlayson, C.J. Gipple, and R.J. Nathan. 2004. Stream hydrology: An introduction for ecologists, 2nd ed. John Wiley and Sons, West Sussex, England.

Gowan, C., and K. D. Fausch. 1996. Mobile brook trout in two high-elevation Colorado streams: reevaluating the concept of restricted movement. Canadian Journal of Fisheries and Aquatic Sciences 53: 1370-1381.

Hartman, K.J., and E.C. Janney. 2006. Visual implant elastomer and anchor tag retention in large mouth bass. North American Journal of Fisheries Management 26: 665-669.

Hill, J., and G.D. Grossman. 1987. Home range estimates for three North American stream fishes. Copeia 2: 367-380.

Hohausova, E. 2000. Exchange rate and small-scale movement of fish between a river and its backwater. Archiv Fur Hydrobiologie 147: 485-504.

Jackson, S.D. 2003. Ecological considerations in the design of river and stream crossings. Pages 20-29 *in* C.L. Irwin, P. Garnett, and K.P. McDermott , editors. 2003 Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment. North Carolina State University, Raleigh, NC.

Johnston, C.E. 2000. Movement patterns of imperiled blue shiners (*Pisces: Cyprinidae*) among habitat patches. Ecology of Freshwater Fish 9: 170-176.

Jolly, G.M. 1965. Explicit estimates from capture-recapture data with both death and immigration – stochastic model. Biometrika 52: 225-247.

Kerby, J.L., S.P.D. Riley, L.B. Kats, and P. Wilson. 2005. Barriers and flow as a limiting factor in the spread of an invasive crayfish (*Procambarus clarkii*) in southern California streams. Biological conservation 126: 402-409.

Larimore, R.W. 1961. Fish population and electrofishing success in a warm-water stream. Journal of Wildlife Management 25: 1-12.

Larson, G. L., R. L. Hoffman, and S. E. Moore. 2002. Observations of the distribution of five fish species in a small Appalachian stream. Transactions of the American Fisheries Society 131: 791-796.

Lebreton, J.D., and R. Pradel. 2002. Multistate recapture models: modelling incomplete individual histories. Journal of Applied Statistics 29: 353-369.

Light, T. 2003. Success and failure in a lotic crayfish invasion: the roles of hydrologic variability and habitat alteration. Freshwater Biology 48: 1886-1897.

Lukey, J.R., A.J. Booth, R. Froneman. 2006. Fish population size and movement patterns in a small intermittently open South African estuary. Estuarine Coastal and Shelf Science 67: 10-20.

Meffe, G.K., and A.L. Sheldon. 1990. Post-defaunation recovery of fish assemblages in southeastern blackwater streams. Ecology 71: 657-667.

Norman, J.R., E. Franzen, H. Millington, B. Ensign, S. Wenger, M. Freeman, and M. Hagler. 2006. Stream crossing and culvert design policy. http://www.etowahhcp.org/research/documents/road_crossings_tech_rpt_2006_08_18.pdf.

Northwest Marine Technology. 2006. Visible implant elastomer tag project manual: Guidelines on planning and conducting project using VIE and associate equipment. http://www.nmt.us/support/appnotes/ape06.pdf.

NRC (National Research Council). 2005. Assessing and managing the ecological impacts of pave roads. National Academy Press, Washington, D.C.

Olsen, E.M., and L.A. Vollestad. 2001. An evaluation of visible implant elastomer for marking age-0 brown trout. North American Journal of Fisheries Management 21: 967-970.

Parsons, G.R., J.J. Hoover, and K.J. Killgore. 2006. Effect of pectoral fin ray removal on station-holding ability of shovelnose sturgeon. North American Journal of Fisheries Management 23: 742-747.

Peterson, J.T., N.P. Banish, and R.F. Thurow. 2005. Are block nets necessary?: movement of stream-dwelling salmonids in response to three common survey methods. North American Journal of Fisheries Management 25: 732-743.

Power, M.E, W.J. Matthews, and A.J. Stewart. 1985. Grazing minnows, piscivorous bass and stream algae: dynamics of a strong interaction. Ecology 66: 1446-1456.

Pradel, R., C.M.A. Winterbertm, and O. Gimenez. 2003. A proposal for a goodness-of-fit test to the Arnason-Schwartz multisite capture-recapture model. Biometrics 59: 43-53.

Pringle, C.M. 1997. Exploring how disturbance is transmitted upstream: going against the flow. Journal of the North American Benthological Society 16: 425-438.

Resh, V.H. 2005. Stream crossings and the conservation of diadromous invertebrates in South Pacific island streams. Aquatic Conservation 15: 313-317.

Reynolds, J. B. 1983. Electrofishing. Pages 147-163 *in* L. A. Nielsen and D. L. Johnson , editors. Fisheries Techniques. American Fisheries Society, Bethesda, MD.

Riley, S.C., K.D. Fausch, and C. Gowan. 1992. Movement of brook trout (*Salvelinus fontinalis*) in four small subalpine streams in northern Colorado. Ecology of Freshwater Fish 1: 112-122.

Roberts, J.H., and P.L. Angermeier. 2004. A comparison of injectable fluorescent marks in two genera of darters: Effects on survival and retention rates. North American Journal of Fisheries Management 24: 1017-1024.

RSCP (River and Stream Continuity Partnership) (Univ. of Massachusetts Amherst, MA Riverways Program, The Nature Conservancy). 2006. Massachusetts River and Stream Crossing Standards. http://www.streamcontinuity.org/pdf_files/MA_Crossing_Stds_3-1-06.pdf.

Schafer, J.F., E. Marsh-Matthews, D.E. Spooner, K.B. Gido, and W.J. Matthews. 2003. Effects of barriers and thermal refugia on local movement of the threatened leopard darter, *Percina pantherina*. Environmental Biology of Fishes 66: 391-400.

Schlosser, I.J., and P.L. Angermeier. 1995. Spatial variations in the demographic processes of lotic fishes: conceptual models, empirical evidence, and implications for conservation. Pages 392-401 *in* J.L. Nielson , editor. Evolution and the Aquatic System: Defining Unique Units in Population Conservation. American Fisheries Society Symposium 17.

Schmetterling, D.A., and S.B. Adams. 2004. Summer movements within the fish community of a small montane stream. North American Journal of Fisheries Management 24: 1163-1172.

Schrank, A.J., and F.J. Rahel. 2004. Movement patterns in inland cutthroat trout (*Oncorhynchus clarki utah*): management and conservation implications. Canadian Journal of Fisheries and Aquatic Sciences 61: 1528-1537.

Schwartz, C.J., J.F. Schweigert, and A.N. Arnason. 1993. Estimating migration rates using tag-recovery data. Biometrics 49: 177-193.

Seber, J.A.F. 1965. A note on the multiple recapture census. Biometrika 52: 249-259.

Slatkin, M. 1987. Gene flow and geographic structure of natural populations. Science 236: 787-792.

Toepfer, C.S., W.L. Fisher, and J.A. Haubelt. 1999. Swimming performance of the threatened leopard darter in relation to road culverts. Transactions of the American Fisheries Society 128: 155-161.

Utzinger, J., C. Roth, and A. Peter. 1998. Effects of environmental parameters on the distribution of bullhead *Cottus gobio* with particular consideration of the effects of obstructions. Journal of Applied Ecology 35: 882-892.

Voelz, N.J., J.V. McAuthur, and R.B. Rader. 1998. Upstream mobility of the Asiatic clam *Corbicula fluminea*: identifying potential dispersal agents. Journal of Freshwater Biology 13: 39-45.

Walsh, M.G., and D.L. Winkelman. 2004. Anchor and visible implant elastomer tag retention by hatchery rainbow trout stocked into an Ozark stream. North American Journal of Fisheries Management 24: 1435-1439.

Ward, J.V. 1998. Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. Biological Conservation 83: 269-278.

Warren, M.L., and M.G. Pardew. 1998. Stream crossings as barriers to small-stream fish movement. Transactions of the American Fisheries Society 127: 637-644.

Webb, P.W. 1973. Effects of partial caudal-fin amputation on the kinematics and metabolic rate of underyearling sockeye salmon (*Oncorhynchus nerka*) at steady swimming speeds. Journal of Experimental Biology 59: 565-581.

Webb, P.W. 1977. Effects of median-fin amputation on fast-start performance of rainbow trout (*Salmo gairdneri*). Journal of Experimental Biology 68:123-135.

Wheeler, A.P., P.L. Angermeier, and A.E. Rosenberger. 2005. Impacts of new highways and subsequent landscape urbanization on stream habitat and biota. Reviews in Fisheries Science 13: 141-164.

Wenger, S.J. 2006. Predicting and preventing losses of imperiled fish species in an urbanizing environment. Ph.D Dissertation, University of Georgia, Athens, GA.

White, G.C., and K.P. Burnham. 1999. Program MARK: survival rate estimation from both live and dead encounters. Bird Study 46(Supplement): S120-S139.

Williams, B.K., J.D. Nichols, and M.J. Conroy , editors. 2002. Analysis and management of animal populations. Academic Press, San Diego, CA.

Wofford, J.E.B., R.E. Gresswell, and M.A. Banks. 2005. Influence of barriers to movement on within-watershed genetic variation of coastal cutthroat trout. Ecological Applications 15: 628-637.

Table 3.1. Comparison of physical and water quality characteristics of mark-recapture study sites. Watershed size (WS) is drainage area above the site; turbidity (Turb), conductivity (Cond), and water temperature (Temp) values are averaged across all sampling dates at each site. Dominant Bed Sed shows bed sediment types in order of prevalence, based on visual estimates.

Site	WS (km^2)	Mean Width (m)	Mean Turb (NTU)	Mean Temp (C°)	Mean Cond (SpC)	Dominant Bed Sed[1]
Canton	4.09	2.7	5.59	22.18	68.98	Gr-Sd-Cb-Sl-Bdr
Bluff	18.15	4.54	8.96	21.5	38.7	Gr-Sd-Cb-Sl-Bdr
Champion	28.06	3.88	1.65	21.3	36.9	Gr-Cb-Sd-Bdr-Sl
Nimblewill	26.05	5.93	3.13	16.51	8.3	Gr-Cb-Sd-Bdr-Sl

[1] Bdr = bedrock, Cb = cobble, Gr = gravel, Sd = sand, Sl = silt

Table 3.2. Description of the road-stream crossing and culvert characteristics for study sites. Length (L) measurements include the length of the apron where applicable. 'Embeddedness' refers to the degree to which the culvert is embedded below the streambed. *min PAWS* and *max PAWS* refer to the minimum and maximum height (i.e., across multiple barrels) the culvert outlet is perched above the water surface, and 'Apron Lip' refers to the height of the apron lip above the apron/culvert bottom.

Site	**Description**	**Width x Height or Diameter (m)**	**L (m)**	**Embeddedness Depth (m) / Percent**	***min PAWS* (m)**	***max PAWS* (m)**	**Apron Lip (m)**
Canton	3-barrel pipe culvert, corrugated metal	3.18	18.6	1.19 / 38%	–	–	–
Bluff	3-barrel box culvert, concrete with headwalls; middle barrel bottomless	3.25 x 3.75 (each barrel)	19.6	Bottomless	–	–	–
Champion	4-barrel box culvert, concrete with headwalls and downstream apron	3 x 1.85 (each barrel)	18.2	–	0.040	0.390	0.090
Nimblewill	4-barrel box culvert, concrete with headwalls and downstream apron	3.5 x 3 (each barrel)	18.6	–	0.231	0.60	0.110

Table 3.3. Marking scheme for all fishes except *C. oligolepis* and *H. etowanum*. The body position where a fish was marked specified the reach in which it was captured.

DS Reaches	Body Position	US Reaches	Body Position
A	R Anterior Dorsal	F	L Anterior Dorsal
B	R Posterior Dorsal	G	L Posterior Dorsal
C	R Posterior Anal	H	L Posterior Anal
D	R Anterior Anal	I	L Anterior Anal
E	Upper Caudal	J	Lower Caudal

Table 3.4. Marking scheme for *C. oligolepis* and *H. etowanum*. The different fin clips taken to specify location upstream or downstream of the culvert for each marking occasion are shown.

Marking Occasion	Upstream	Downstream
1	Lower caudal fin lobe	Upper caudal fin lobe
2	Left pectoral fin	Right pectoral fin
3	Anal fin	Dorsal fin

Table 3.5. Description of site-specific candidate models and model selection procedure. Arrows denote that the best supported model in each pair of models was used as the alternative for the next model comparison; (CL = culvert length, SL = standard length).

Assumptions	Model	Description
Adjacent moves are more likely than short moves and long moves, and short moves are more likely than long moves. Culvert-free moves are more likely than moves through a culvert (Fig. 3.3, 3.5)	S(pairs) p(reach) ψ(g1)	Full Movement (FM) Grouped model – reach-paired survival, reach-specific recapture rate, and transition probabilities grouped by adjacent moves, moves ≈1 CL, moves >1 CL, thru-culvert moves, and moves thru-culvert + ≥1 CL.
Adjacent moves are more likely than longer moves. Culvert-free moves are more likely than moves through a culvert (Fig. 3.4, 3.5)	S(pairs) p(reach) ψ(g2)	Reduced Movement (RM) Grouped model – reach-paired survival, reach-specific recapture rate, and transition probabilities grouped by adjacent moves, moves ≥ 1 CL, and moves thru-culvert plus moves thru-culvert + ≥1 CL.
↓		
Constant survival provides a better fit than reach paired survival. (Fig. 3.3, 3.5, 3.6)	S(.) p(reach) ψ(g1)	Full Movement Grouped with Constant Survival (FMCS) instead of reach paired survival.
	OR	
Constant survival provides a better fit than reach paired survival. (Fig. 3.4, 3.5, 3.6).	S(.) p(reach) ψ(g2)	Reduced Movement Grouped with Constant Survival (RMCS) instead of reach paired survival.
↓		
Standard length is a good predictor of movement (use best supported model *X* from above).	S(*x*) p(reach) $\psi(x)^{SL}$	Model *X* with movement constrained as a function of SL.

Table 3.6. Description of combined-site candidate models and model selection procedure. Arrows denote that the best supported model in each pair of models was used as the alternative for the next model comparison; (US = upstream, DS = downstream, *min PAWS* = minimum height culvert outlet is perched above water surface, *Apron* = height of apron lip on upstream side of culvert outlet)

Assumptions	Model	Description
Adjacent moves are more likely than short moves and long moves, and short moves are more likely than long moves. Culvert-free moves are more likely than moves through a culvert (Fig. 3.3, 3.5)	S(pairs) p(reach) ψ(g1)	Full Movement (FM) Grouped model – reach paired survival, reach specific recapture rate, and transition probabilities grouped by adjacent moves, moves ≈1 CL, moves >1 CL, thru-culvert moves, and moves thru-culvert + ≥1 CL.
Adjacent moves are more likely than longer moves. Culvert-free moves are more likely than moves through a culvert (Fig. 3.4, 3.5)	S(pairs) p(reach) ψ(g2)	Reduced Movement (RM) Grouped model – reach paired survival, reach specific recapture rate, and transition probabilities grouped by adjacent moves, moves ≥ 1 CL, and moves thru-culvert plus moves thru-culvert + ≥1 CL.
↓		
Constant survival provides a better fit than reach paired survival. (Fig. 3.3, 3.5, 3.6)	S(.) p(reach) ψ(g1)	Full Movement Grouped with Constant Survival (FMCS) instead of reach paired survival.
	OR	
Constant survival provides a better fit than reach paired survival. (Fig. 3.4, 3.5, 3.6).	S(.) p(reach) ψ(g2)	Reduced Movement Grouped with Constant Survival (RMCS) instead of reach paired survival.
↓		
min PAWS is a good predictor of US thru-culvert movement and *Apron* is a good predictor of DS thru-culvert movement (use best supported model *X* from above).	$S(x)\ p(\text{reach})\ \psi(x)^{Min\ PAWS\ +\ Apron}$	*X* model with US thru-culvert movement constrained as a function of *min PAWS* and DS thru-culvert movement as a function of height of apron lip (*Apron*).
min PAWS is a good predictor of upstream thru-culvert movement (use best supported model *X* from above).	$S(x)\ p(\text{reach})\ \psi(x)^{Min\ PAWS}$	*X* model with US thru-culvert movement constrained as a function of *min PAWS*.

Table 3.7. List of benthic and water column fish species caught at mark-recapture study sites, showing the total number of individuals marked for each species at each site.

Genus species	Canton	Bluff	Champion	Nimblewill
Benthic fishes				
Amieurus melas	1			
Amieurus sp.	1			
Campostoma oligolepis	76	81	213	16
Cottus sp. cf. carolinae		53	224	469
Etheostoma sp. cf. brevirostrum				1
Etheostoma scotti	51	23	92	
Etheostoma stigmaeum		1		
Hypentelium etowanum	13	315	163	130
Noturus leptacanthus		39	41	
Percina kathae	40	7		
Percina sp. cf. macrocephala			1	
Percina nigrofasciata	31	89	27	
Percina palmaris			36	11
Water column fishes				
Cyprinella callistia		131	257	
Fundulus stellifer		2		
Gambusia holbrooki	3			
Lepomis auritus	296	9	3	
Lepomis cyanellus		1	4	
Lepomis gulosus	4			
Lepomis macrochirus	344	46	7	
Lepomis microlophus	4			
Lepomis punctatus			1	
Micropterus coosae	2	12	19	
Micropterus punctulatus		1		
Micropterus salmoides	5	3		
Moxostoma sp.		25		
Nocomis leptocephalus		39	1	177
Notropis lutipinnis		172		1268
Notropis xaenocephalus	8	83	494	
Semotilus atromaculatus	2	99	13	18

Table 3.8. The overall recapture rate and the number of fish captured 1, 2, 3 and 4 times at each study site.

Site	Recap rate	1 capture	2 captures	3 captures	4 captures
Canton Creek	0.37	572	266	58	7
Bluff Creek	0.31	842	312	55	7
Champion Creek	0.37	1009	425	143	19
Nimblewill Creek	0.29	1476	509	93	12

Table 3.9. The total number and percentage of moving and non-moving fish, and the total number of fish moving upstream (US) and downstream (DS), at each site and the sites combined (excluding *C. oligolepis* and *H. etowanum*).

Site	**Total Non-movers**	**Total Movers**	**% Non-movers**	**% Movers**	**US Moves**	**DS Moves**
Canton	725	89	89%	11%	47	42
Bluff	738	82	90%	10%	45	37
Champion	1133	87	93%	7%	51	36
Nimblewill	1832	112	94%	6%	67	45
TOTAL	4428	370	92%	8%	210	160

Table 3.10. Summary of fish movements over the course of the study for each site. The number of fish that moved over each distance category is shown, as well as the percentage of total moves in each distance category and the species of fish that moved through the culvert. Movements through the culvert are recorded for (1) transitions between the reaches immediately upstream and downstream of the culvert ('Thru-culvert'), and (2) transitions through the culvert that involved movements over additional reaches ('Thru-culvert>1'). Thru-culvert moves are labeled for upstream (US), downstream (DS) or a combination of upstream and downstream movements (US/DS).

Site	Distance Moved	Numbers	% of total moves	Thru-culvert moves
Canton Creek	Adjacent	77	87%	**Thru-culvert:** None
	≈1 CL	8	9%	
	>1 CL	3	3%	**Thru-culvert >1:** Benthic – *P. kathae*
	Thru-culvert	0	–	
	Thru-culvert + ≥1 CL	1 (US)	1%	
Bluff Creek	Adjacent	52	63%	**Thru-culvert:** Water column – *N. leptacanthus, N. lutipinnis, N. xaenocephalus*
	≈1 CL	12	15%	
	>1 CL	1	1%	**Thru-culvert >1:** Benthic – *E. scotti, P. kathae*
	Thru-culvert	5 (US/DS)	6%	
	Thru-culvert + ≥1 CL	12 (US/DS)	15%	Water column – *C. callistia, N. leptocephalus, N. lutipinnis, S. atromaculatus*
Champion Creek	Adjacent	61	70%	**Thru-culvert:** Water column – *N. xaenocephalus*
	≈1 CL	12	14%	
	>1 CL	7	8%	**Thru-culvert >1:** Water column – *C. callistia, N. xaenocephalus*
	Thru-culvert	3 (US)	3%	
	Thru-culvert + ≥1 CL	4 (US)	5%	
Nimblewill Creek	Adjacent	79	71%	**Thru-culvert:** None
	≈1 CL	21	19%	
	>1 CL	7	6%	**Thru-culvert >1:** Water column – *N. lutipinnis*
	Thru-culvert	0	–	
	Thru-culvert + ≥1 CL	5 (DS)	4%	

Table 3.11. AICc values and number of estimated parameters for site-specific candidate models.

Canton Creek			
Model	**AICc**	**AICc Weight**	**Estimated Parameters**
S(.) p(g) ψ(g2)	2589.316	0.581	29
S(pairs) p(g) ψ(g2)	2590.167	0.379	35
S(pairs) p(g) ψ(g1)	2594.866	0.0362	38
S(.) p(g) ψ(g2)^{SL}	2599.362	0.004	29
Bluff Creek			
Model	**AICc**	**AICc Weight**	**Estimated Parameters**
S(.) p(g) ψ(g1)	2034.593	0.992	36
S(pairs) p(g) ψ(g1)	2044.225	0.008	44
S(pairs) p(g) ψ(g2)	2065.238	0.000	42
Champion Creek			
Model	**AICc**	**AICc Weight**	**Estimated Parameters**
S(.) p(g) ψ(g1)	3648.511	0.965	35
S(pairs) p(g) ψ(g1)	3655.421	0.0305	41
S(.) p(g) ψ(g1)^{SL}	3659.588	0.00380	35
S(pairs) p(g) ψ(g2)	3663.716	0.000	37
Nimblewill Creek			
Model	**AICc**	**AICc Weight**	**Estimated Parameters**
S(pairs) p(g) ψ(g1)	4476.298	0.854	38
S(pairs) p(g) ψ(g1)^{SL}	4480.635	0.0977	38
S(pairs) p(g) ψ(g2)	4482.322	0.0420	36
S(.) p(g) ψ(g1)	4486.225	0.00597	32

Table 3.12. Parameter estimates for survival (S) and the range of parameter estimates for reach-specific recapture probabilities (p) for benthic (B) and water column (WC) fishes from the best-supported model for each site. A range of parameter estimates for survival is shown for Nimblewill Creek because the best-supported model for that site included reach-paired survival, whereas at all other sites, constant survival had the most support.

Parameters	**Canton Creek**		**Bluff Creek**		**Champion Creek**		**Nimblewill Creek**	
	B	**WC**	**B**	**WC**	**B**	**WC**	**B**	**WC**
S	0.82	0.82	0.84	0.81	0.81	0.94	0.64 – 0.93	0.55 – 0.84
p	0.11 – 0.40	0.080 – 0.33	0.054 – 0.37	0.054 – 0.36	0.18 – 0.39	0.025 – 0.47	0.14 – 0.35	0.11 – 0.47

Table 3.13. Transition probabilities (ψ) from RMCS model for benthic fishes (B) and water column fishes (WC) at Canton Creek. Standard error is shown in parentheses and 95% confidence intervals are shown in brackets for each ψ. (US = upstream movement, DS = downstream movement, CL = culvert length).

Direction and Distance of Move	**ψB**	**ψWC**
US Adjacent	**0.063** (0.045) [0.015, 0.23]	**0.16** (0.024) [0.12, 0.21]
US ≥1 CL	**0.0073** (0.016) [0.96^{E-04}, 0.36]	**0.012** (0.0057) [0.0044, 0.030]
US Thru-Culvert + ≥1 CL	**0.0054** (0.006) [0.73^{E-03}, 0.039]	**0**
DS Adjacent	**0.068** (0.034) [0.025, 0.17]	**0.091** (0.017) [0.063, 0.13]
DS ≥1 CL	**0**	**0.0055** (0.0034) [0.0016, 0.018]
DS Thru-Culvert + ≥1 CL	**0**	**0**

Table 3.14. Transition probabilities (ψ) for benthic fishes (B) and water column fishes (WC) at Bluff Creek (FMCS model), Champion Creek (FMCS model) and Nimblewill Creek (FM model). Standard error is shown in parentheses and 95% confidence intervals are shown in brackets for each ψ. (US = upstream movement, DS = downstream movement, and CL = culvert length).

Direction and Distance of Move	**Bluff Creek**		**Champion Creek**		**Nimblewill Creek**	
	ψB	**ψWC**	**ψB**	**ψWC**	**ψB**	**ψWC**
US Adjacent	**0.056** (0.033) [0.017, 0.17]	**0.13** (0.026) [0.087, 0.19]	**0.083** (0.025) [0.046, 0.15]	**0.092** (0.018) [0.062, 0.13]	**0.097** (0.025) [0.058, 0.16]	**0.081** (0.014) [0.056, 0.11]
US ≈1 CL	**0.022** (0.024) [0.0025, 0.17]	**0.026** (0.012) [0.010, 0.065]	**0.056** (0.026) [0.023, 0.13]	**0. 0072** (0.0052) [0.0017, 0.029]	**0.049** (0.021) [0.021, 0.11]	**0.015** (0.0069) [0.0062, 0.037]
US >1 CL	**0**	**0.010** (0.0074) [0.0025, 0.041]	**0.0097** (0.015) [0.45^{E-03}, 0.18]	**0.0080** (0.0042) [0.0028, 0.022]	**0**	**0.0089** (0.0054) [0.0027, 0.029]
US Thru-Culvert	**0**	**0.11** (0.056) [0.037, 0.27]	**0**	**0.051** (0.027) [0.017, 0.14]	**0**	**0**
US Thru-culvert + ≥1 CL	**0.011** (0.0078) [0.0025, 0.044]	**0.0066** (0.0034) [0.0024, 0.018]	**0**	**0.0017** (0.0012) [0.42^{E-03}, 0.0070]	**0**	**0**
DS Adjacent	**0.070** (0.035) [0.026, 0.18]	**0.14** (0.030) [0.091, 0.21]	**0.026** (0.012) [0.010, 0.063]	**0.14** (0.022) [0.099, 0.19]	**0.016** (0.0095) [0.0052, 0.050]	**0.067** (0.012) [0.047, 0.095]
DS ≈1 CL	**0.026** (0.026) [0.0036, 0.17]	**0.026** (0.016) [0.0075, 0.083]	**0.0063** (0.0068) [0.78^{E-03}, 0.050]	**0.015** (0.0084) [0.0050, 0.044]	**0**	**0.019** (0.0067) [0.0098, 0.038]
DS >1 CL	**0**	**0**	**0**	**0.0048** (0.0050) [0.62^{E-03}, 0.036]	**0**	**0.0067** (0.0040) [0.0021, 0.021]
DS Thru-Culvert	**0.095** (0.094) [0.012, 0.47]	**0**	**0**	**0**	**0**	**0**
DS Thru-culvert + ≥1 CL	**0**	**0.011** (0.0044) [0.0052, 0.024]	**0**	**0**	**0**	**0.0075** (0.0034) [0.0031, 0.018]

Table 3.15. Culvert effect size on transition probabilities for benthic (B) and water column (WC) fishes. Values are shown only where effect size was calculable. 95% confidence intervals are shown in parentheses. (*NCM* = no observed movement through the culvert, No Effect = transition probability for movement through the culvert ≥ culvert-free movement).

Site	**US culvert effect**		**US effect culvert >1**		**DS culvert effect**		**DS culvert effect >1**	
	B Fishes	WC Fishes	B Fishes	WC Fishes	B Fishes	WC Fishes	B Fishes	WC Fishes
Canton	*NCM*	*NCM*	0.0019 (-0.032, 0.036)	*NCM*	*NCM*	*NCM*	*NCM*	*NCM*
Bluff	*NCM*	No Effect	No Effect	0.0034 (-0.012, 0.019)	No Effect	*NCM*	*NCM*	No Effect
Champion	*NCM*	No Effect	*NCM*	0.0021 (-0.053, 0.058)	*NCM*	*NCM*	*NCM*	*NCM*
Nimblewill	*NCM*	*NCM*	*NCM*	*NCM*	*NCM*	*NCM*	*NCM*	No Effect

Table 3.16. Change in stage height (Δ Stage), *effective min PAWS* (Eff P), and the number fish that had moved upstream (US) or downstream (DS) through a culvert (# TC) after each time interval between sampling occasions. The number of fish that had moved through a culvert is the number of fish caught that had moved through a culvert *after* a given time interval, not necessarily the time interval over which an individual *actually* moved through the culvert. The numbers in brackets represent the number of fish that *actually* moved through a culvert *over* a particular time interval. (Eff P is not applicable to Canton and Bluff Creeks because the culverts at those sites were embedded below the stream bed).

Time	Canton Creek			Bluff Creek			Champion Creek			Nimblewill Creek		
Interval	**Δ Stage (m)**	**Eff P (m)**	**# TC (US/DS)**	**Δ Stage (m)**	**Eff P (m)**	**# TC (US/DS)**	**Δ Stage (m)**	**Eff P (m)**	**# TC (US/DS)**	**Δ Stage (m)**	**Eff P (m)**	**# TC (US/DS)**
1	0	–	0 0	0.11	–	1/0 **[1/0]**	0.056	-0.016	0 0	0.387	-0.156	0 0
2	0.10	–	0 0	0.40	–	3/3 **[3/2]**	0.091	-0.051	0 0	0.513	-0.282	0/4 0
3	0.075	–	0 0	0.041	–	7/7 **[4/5]**	0.212	-0.172	7/0 **[3/0]**	0.16	0.071	0/1 0
4	0.070	–	1/0 0	–	–	–	–	–	–	–	–	–

Table 3.17. AICc values and number of estimated parameters for combined-site candidate models.

Model	AICc	AICc Weight	Estimated Parameters
S(pairs) p(g) ψ(g1)	11631.20	0.370	89
S(pairs) p(g) ψ(g1)$^{Min\ PAWS}$	11631.20	0.370	89
S(pairs) p(g) ψ(g1)$^{Min\ PAWS\ +\ Apron}$	11633.26	0.132	89
S(.) p(g) ψ(g1)	11633.32	0.128	74
S(pairs) p(g) ψ(g2)	11653.17	0.000	80

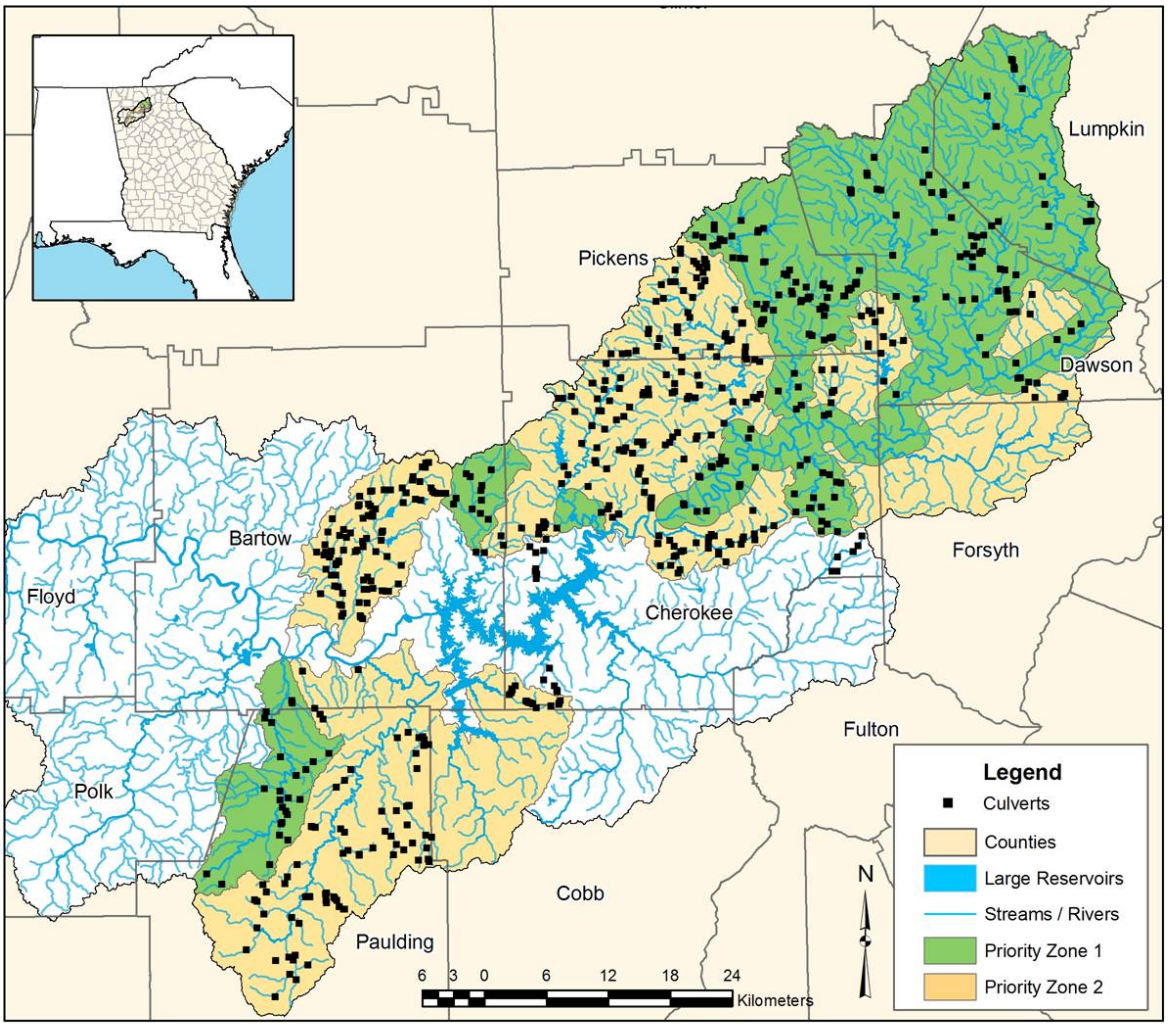

Fig. 3.1. Map of the Etowah River Basin showing the candidate road-stream crossings considered in site selection for mark-recapture studies. The location of the Etowah River Basin in the state of Georgia is also shown (lower left). Priority 1 zones encompass all of the habitat for the two Federally endangered species in the Etowah River Basin, *Etheostoma etowahae* (Etowah darter) and *Percina antesella* (Amber darter). Priority 2 zones cover the primary remaining habitat for the Federally threatened species *Etheostoma scotti* (Cherokee darters) (Wenger 2006).

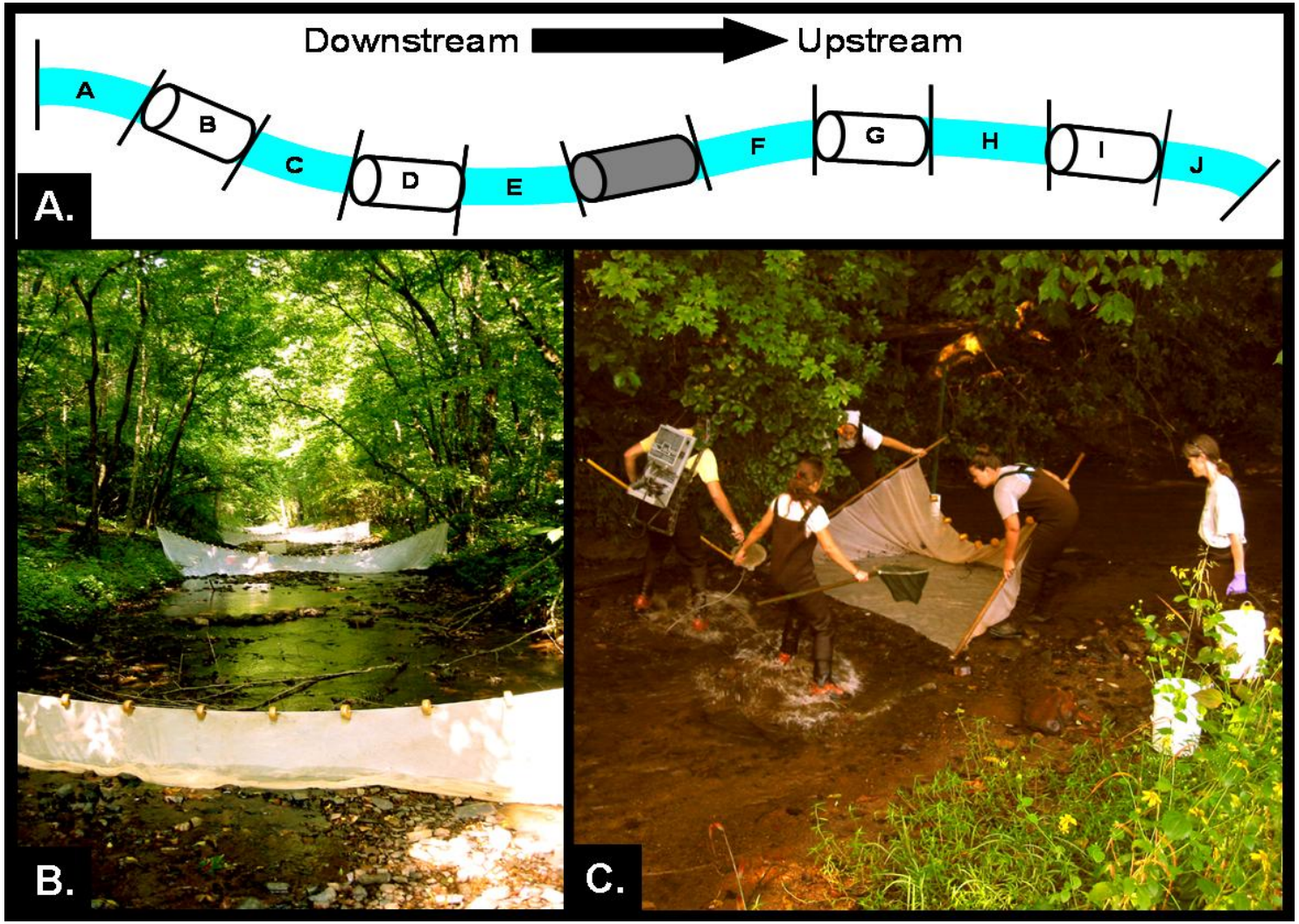

Fig. 3.2. Diagram and photographs of study design and sampling methods. A.) Diagram showing how study site reaches were designated. The road-stream crossing is shown as the shaded cylinder and the false culverts are shown as the lettered cylinders. B.) Photograph showing block nets in place at Champion Creek. C.) Photograph showing usage of backpack electroshocker and seine used to catch fish at study sites (pictured: Christina Baker, Jeffery Garnett, Rachel Katz, Nicole Pontzer and Mary Freeman).

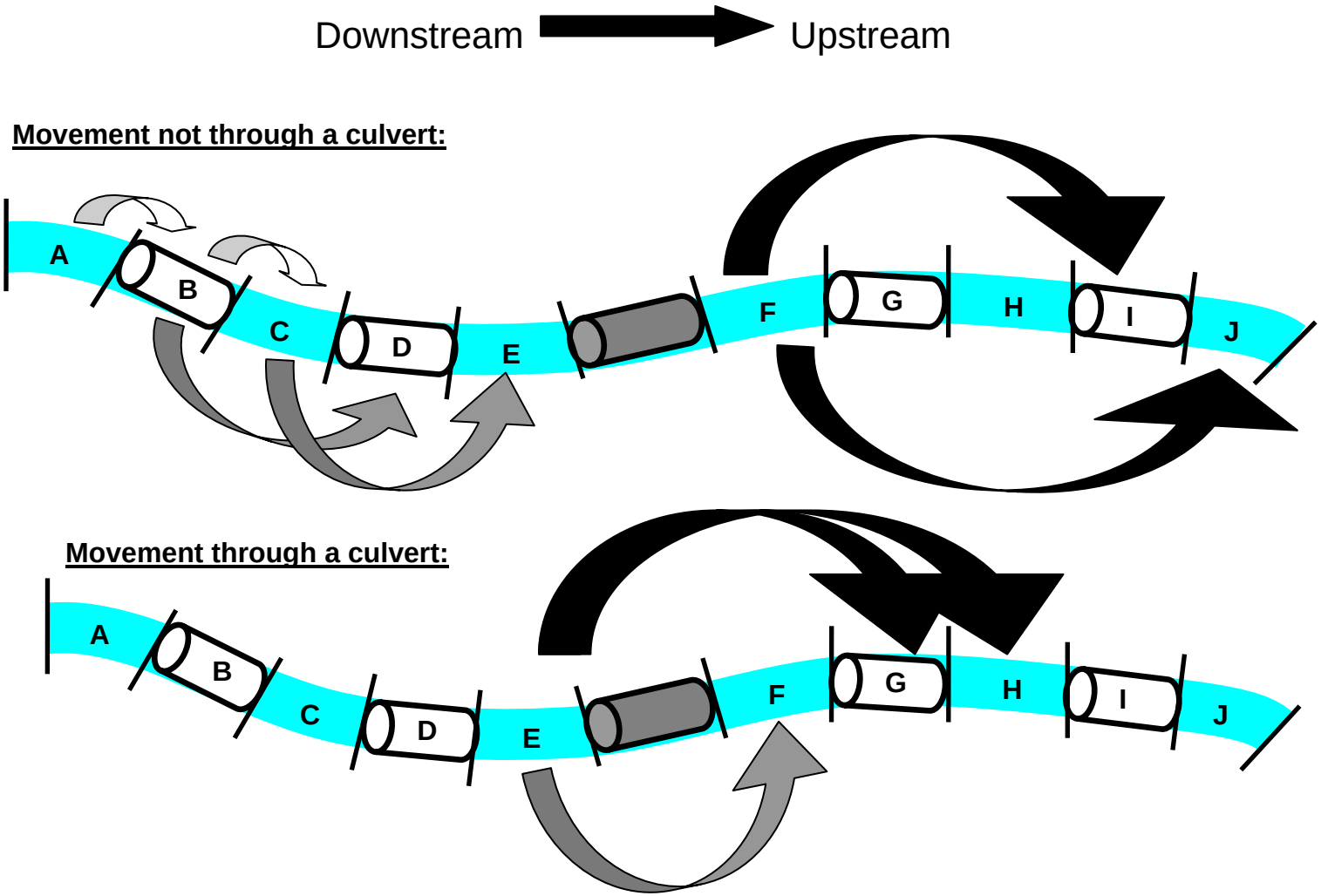

Figure 3.3. Diagram showing the *upstream* transition probabilities estimated in the Full Movement (FM) grouped model, not through the culvert (upper): between adjacent reaches (e.g., A to B or B to C), across an intervening reach (e.g., B to D or C to E), and across multiple reaches (e.g., F to I or F to J). Movement probabilities through the culvert (lower) were estimated for E to F *or* E to G, H, I or J. The corresponding *downstream* transition probabilities were also estimated; the road-stream crossing is the shaded cylinder; reaches designated as 'false culverts' are shown as lettered cylinders.

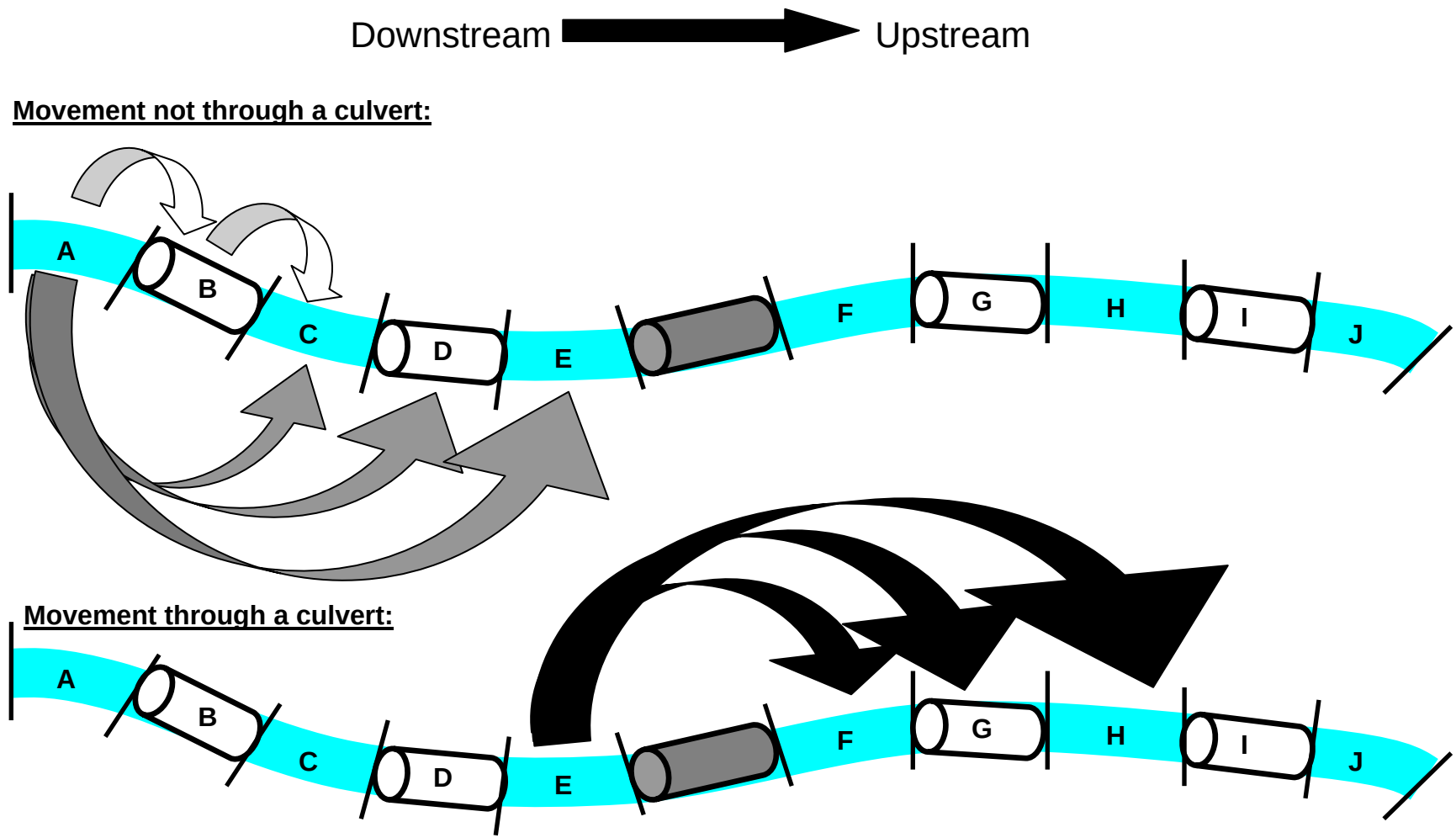

Fig. 3.4. Diagram showing the *upstream* transition probabilities estimated in the Reduced Movement (RM) grouped model, not through the culvert (upper): between adjacent reaches (e.g., A to B or B to C), and across an intervening reach *or* multiple reaches (e.g., A to C or A to D or A to E). A single movement probability through the culvert (lower) was estimated for E to F, G, H, I or J. The corresponding *downstream* transition probabilities were also estimated; the road-stream crossing is the shaded cylinder; reaches designated as 'false culverts' are shown as lettered cylinders.

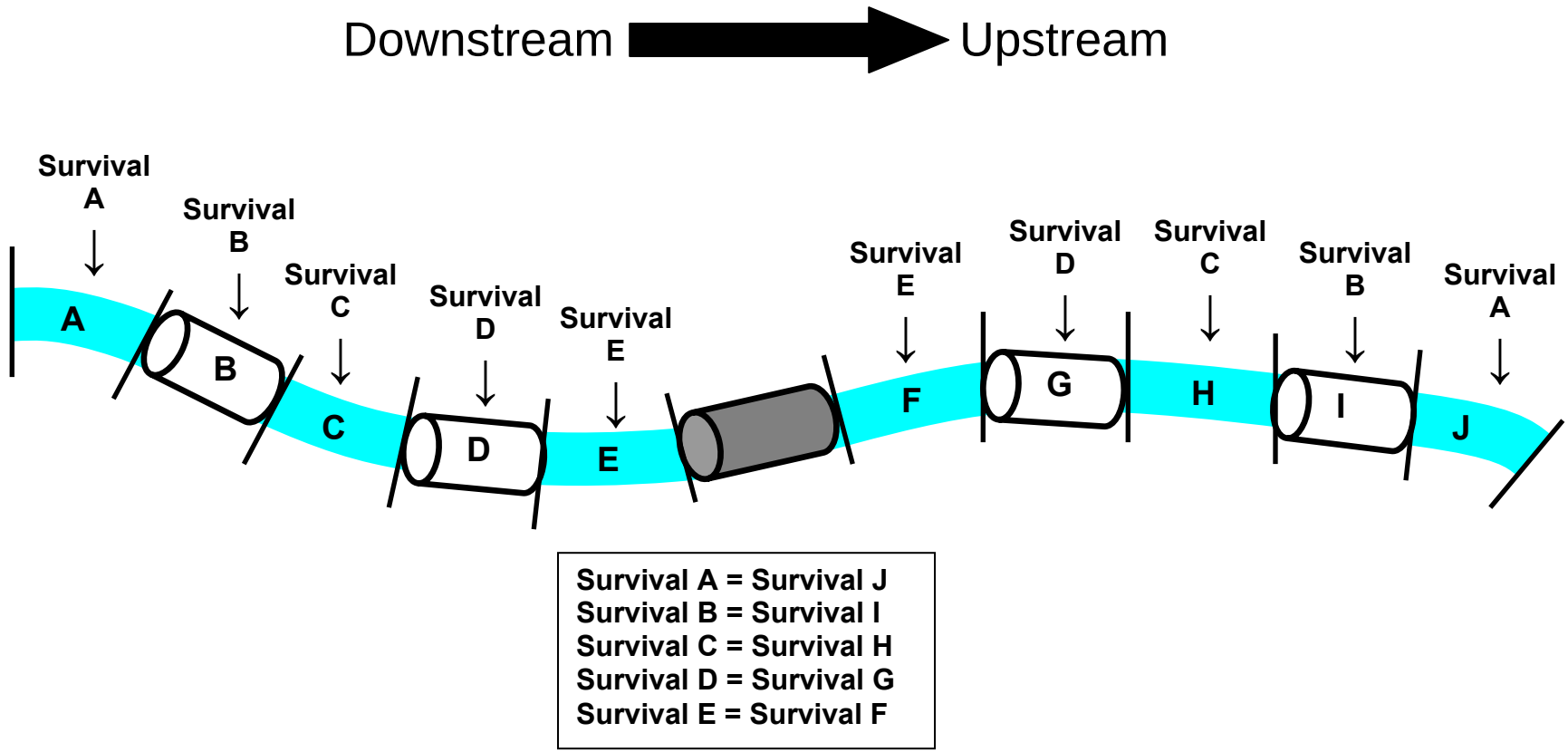

Fig. 3.5. Diagram showing assumptions of model with reach paired survival.

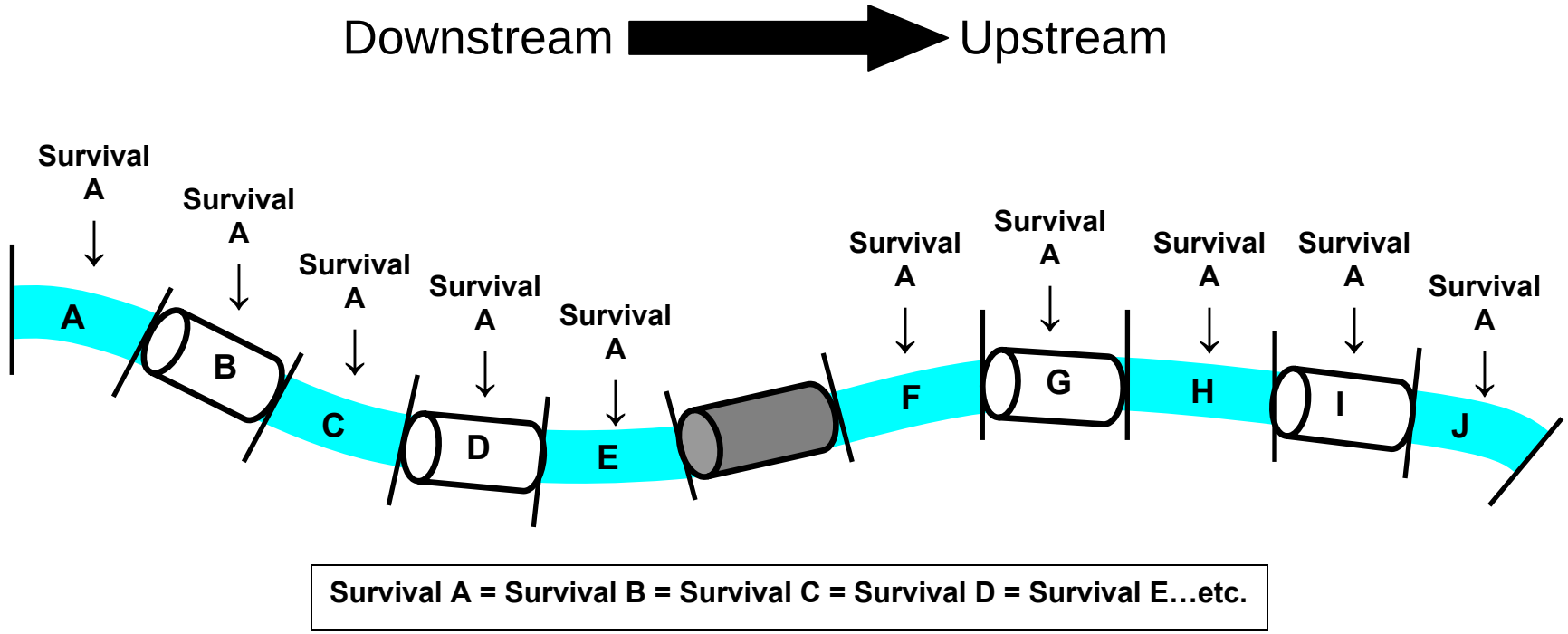

Fig. 3.6. Diagram showing assumptions of model with constant survival.

Chapter 4

Conclusion

In general, the road-stream RPMs evaluated in this research appeared to be relatively effective in preventing changes to channel geomorphology with respect to channel depth and bed sediment composition, but less effective in preventing channel width changes and blocked fish passage. Most of the changes to channel geomorphology at RPM crossings resulted from (1) unstable banks around the crossing, (2) excessive channel widening around the crossing during construction that was inadequately restored, and (3) culverts being undersized to pass high flows and high sediment and debris loads, while concerns about impeded fish passage at RPM crossings were due to (1) shallow depths inside the culvert or around the crossing, (2) lack of substrate inside the culvert, (3) debris dams at culvert inlets, and (4) culverts being perched above the streambed or water surface. The efficacy of the FWS road-stream RPMs in preventing changes to channel morphology and fish passage could be improved by placing more emphasis on bank and channel form restoration following construction of crossings and by adopting the stream simulation (Gubernick and Bates 2003, WDFW 2003, Norman et al. 2006, RSCP 2006) approach to culvert design. Greater enforcement of applicant compliance with RPM terms and conditions could also help improve the efficacy of FWS RPMs in preventing channel alterations and blocked fish passage.

We observed varying degrees of fish movement through the culverts at our mark-recapture study sites. None of the culverts were complete barriers to passage, however the two perched multiple-barrel box culverts were likely to impede upstream and/or downstream passage by both benthic and water column fishes. The multiple-barrel box culvert with a bottomless

middle barrel and continuous streambed substrate throughout appeared to provide for fish passage comparable to that of natural stream reaches. At an embedded multiple-barrel pipe culvert with debris partially obstructing the inlet, we observed only a single fish movement through the culvert.

Based on these observations, we predict that fish passage is likely to be reduced compared to natural reaches at pipe culverts, especially if debris barriers form at the inlet of one or more pipes, and at perched box culverts. We suspect that both benthic and water column fishes are more likely to be able to move downstream than upstream through culverts with the outlet perched above the streambed or the water surface, but that box culverts perched above the water surface likely serve as greater barriers to passage by benthic fishes than water column fishes. In addition, we predict that the more perched a culvert outlet becomes above the water surface, the more likely that culvert is to prohibit upstream passage by benthic fishes over any range of flows, especially where streambed substrate is not present inside the culvert. We observed that the ability of benthic and water column fishes to pass through box culverts perched above the water surface varied temporally in relation to increases in stage height that were sufficient to submerge the culvert outlets. Thus, the degree to which perched culverts impede upstream and/or downstream passage will depend on the extent to which the culvert is perched above the water surface as well as the ability and tendency of fishes to move upstream or downstream through the culvert when increased water velocities and discharge associated with increases in stage height occur.

As road-stream crossings have the potential to affect the connectedness of stream habitat and fish populations over large areas, more research is needed to fully understand and quantify the effects of road-stream crossing on fish passage and habitat and population fragmentation, and

how those effects vary among species and along ecological gradients. Use of mark-recapture studies and multi-state models provides a promising framework for future research attempting to systematically quantify the effects of road-stream crossings on fish passage and habitat fragmentation. A more thoroughly quantified understanding of the effects of road-stream crossings on fish passage and fragmentation is important to making decisions about the replacement of existing culverts and the design and construction of new road-stream crossings that provide the most biological benefit while maximizing economic efficiency.

APPENDICES

APPENDIX A: SUPPLEMENTAL MATERIALS FOR CHAPTER 2

The following figures show longitudinal profiles for basal channel width, thalweg depth and phi (raw data and running averages of four transects) at all eleven RPM road-stream crossing sites.

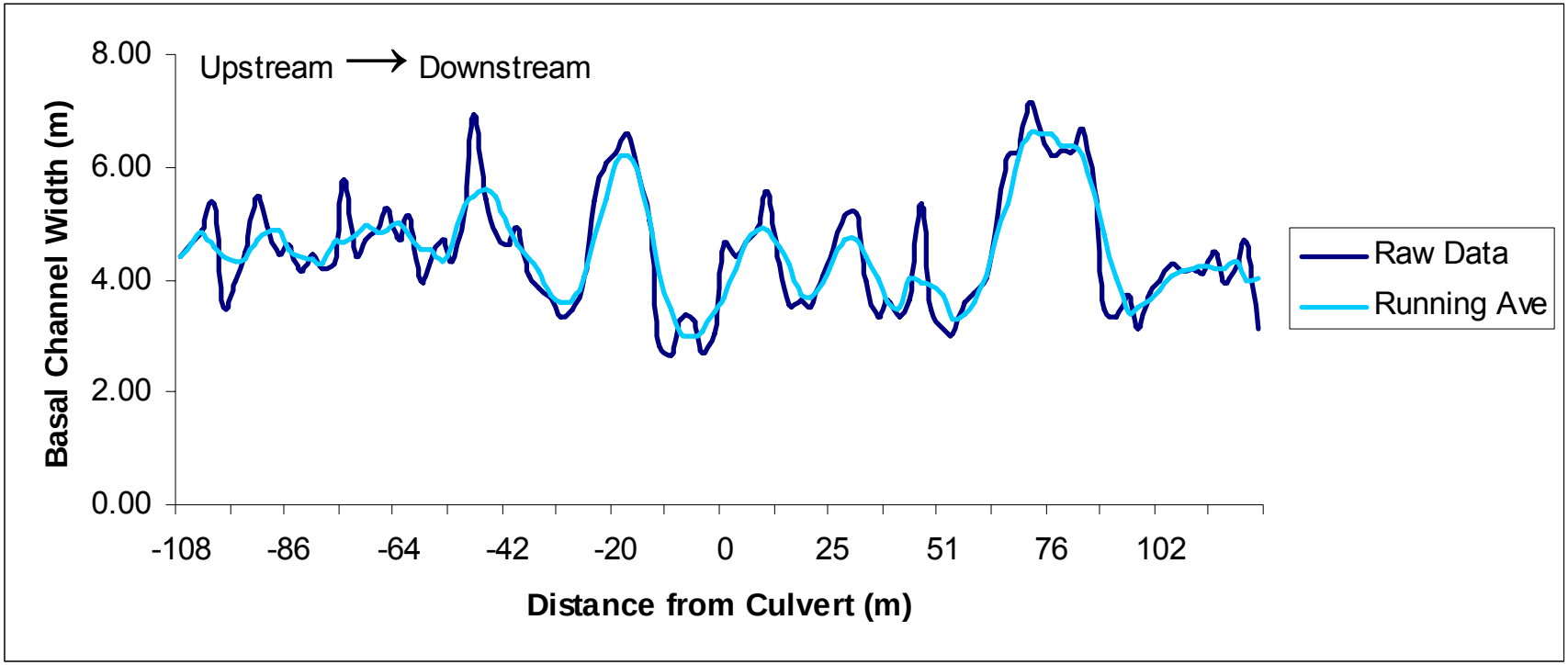

Fig. 1. Change in basal channel width along longitudinal transect at Site 1. Culvert is located at 0 on the x-axis.

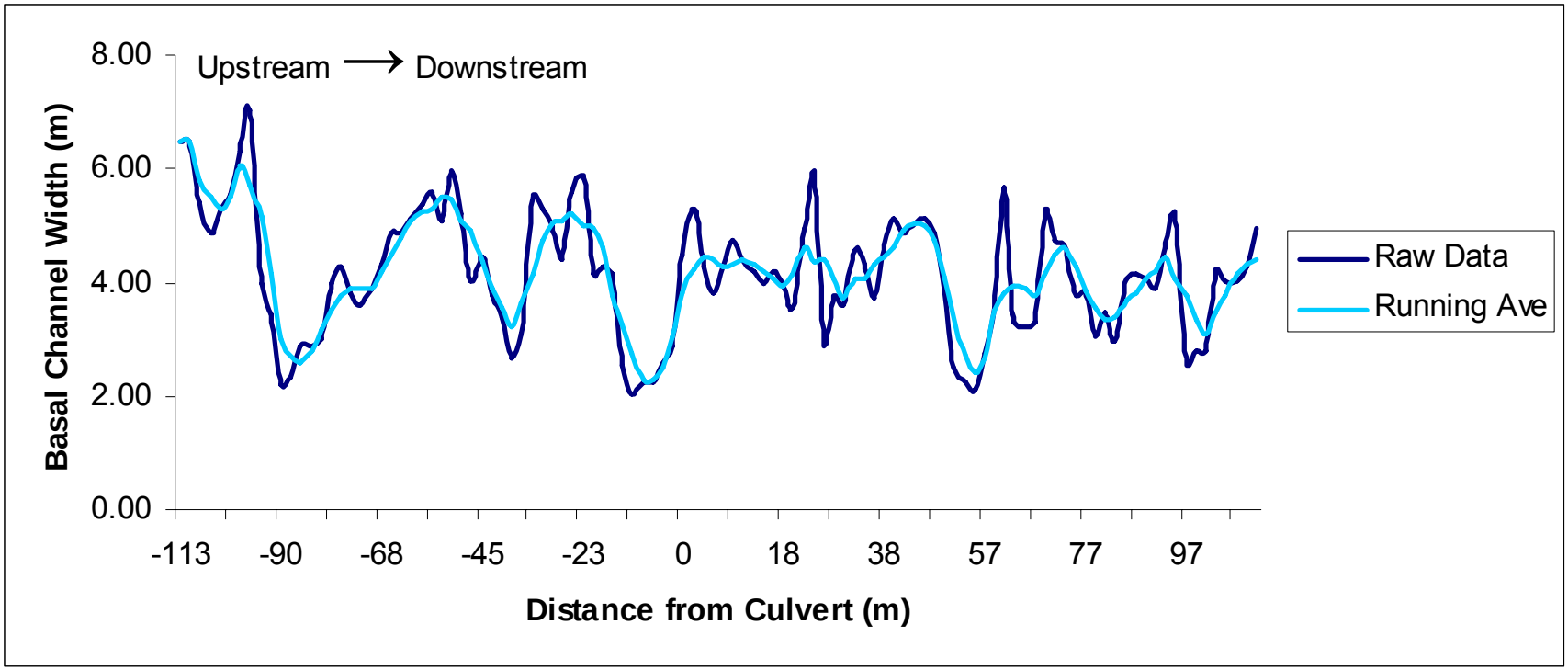

Fig. 2. Change in basal channel width along longitudinal transect at Site 2. Culvert is located at 0 on the x-axis.

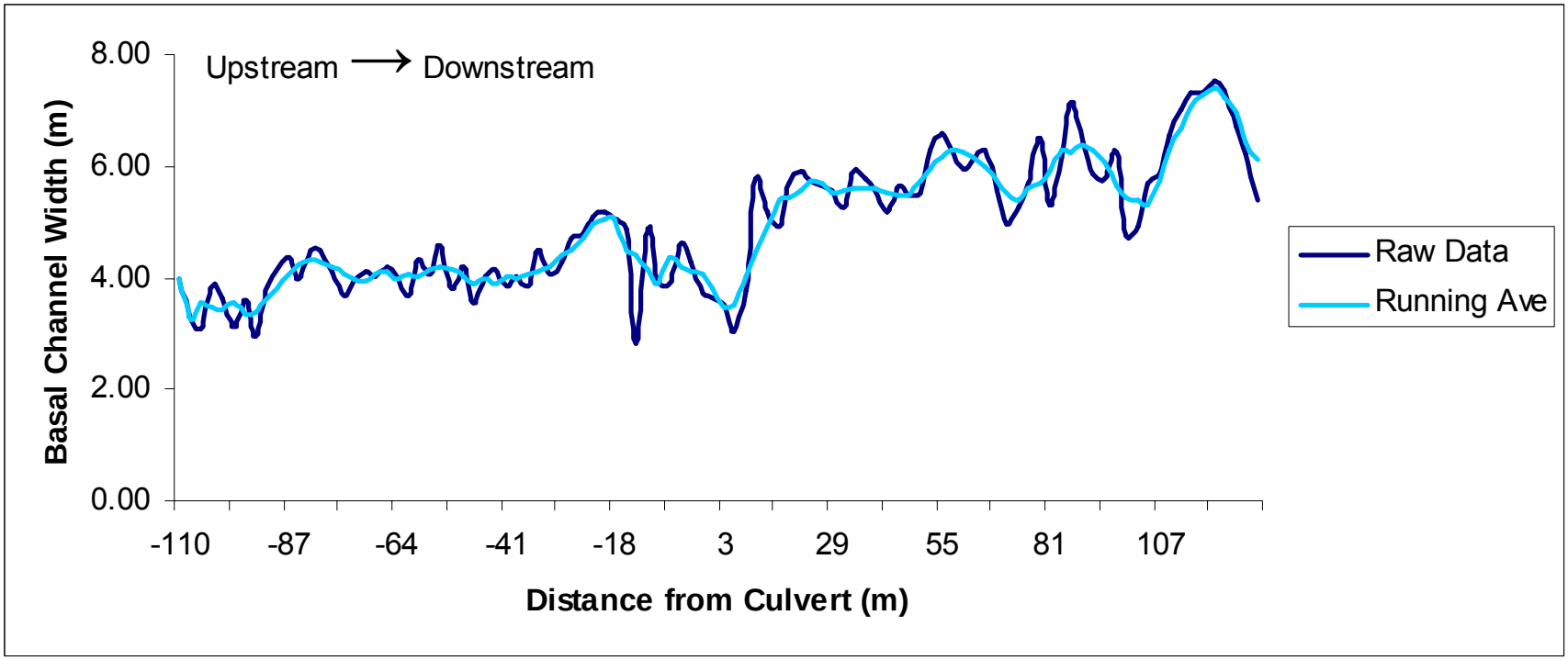

Fig. 3. Change in basal channel width along longitudinal transect at Site 3. Culvert is located at 0 on the x-axis.

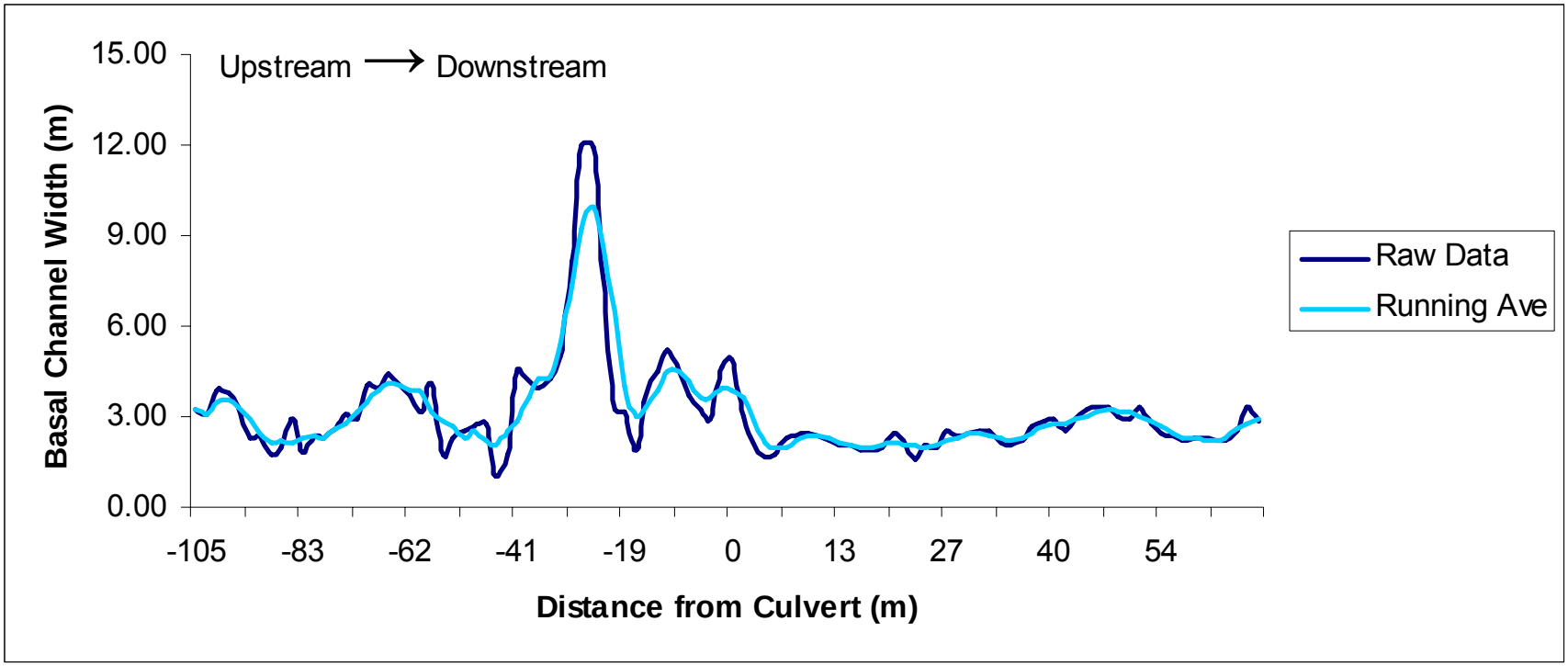

Fig. 4. Change in basal channel width along longitudinal transect at Site 4. Culvert is located at 0 on the x-axis.

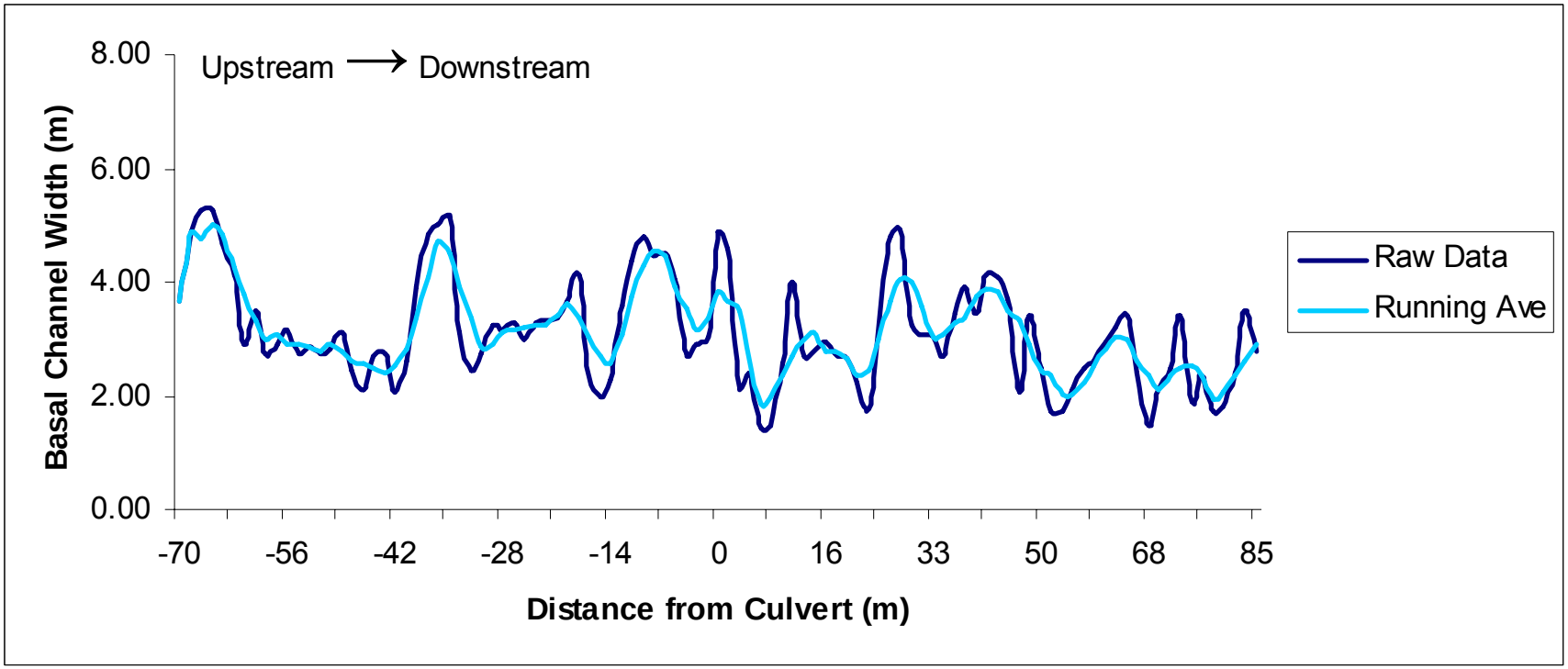

Fig. 5. Change in basal channel width along longitudinal transect at Site 5. Culvert is located at 0 on the x-axis.

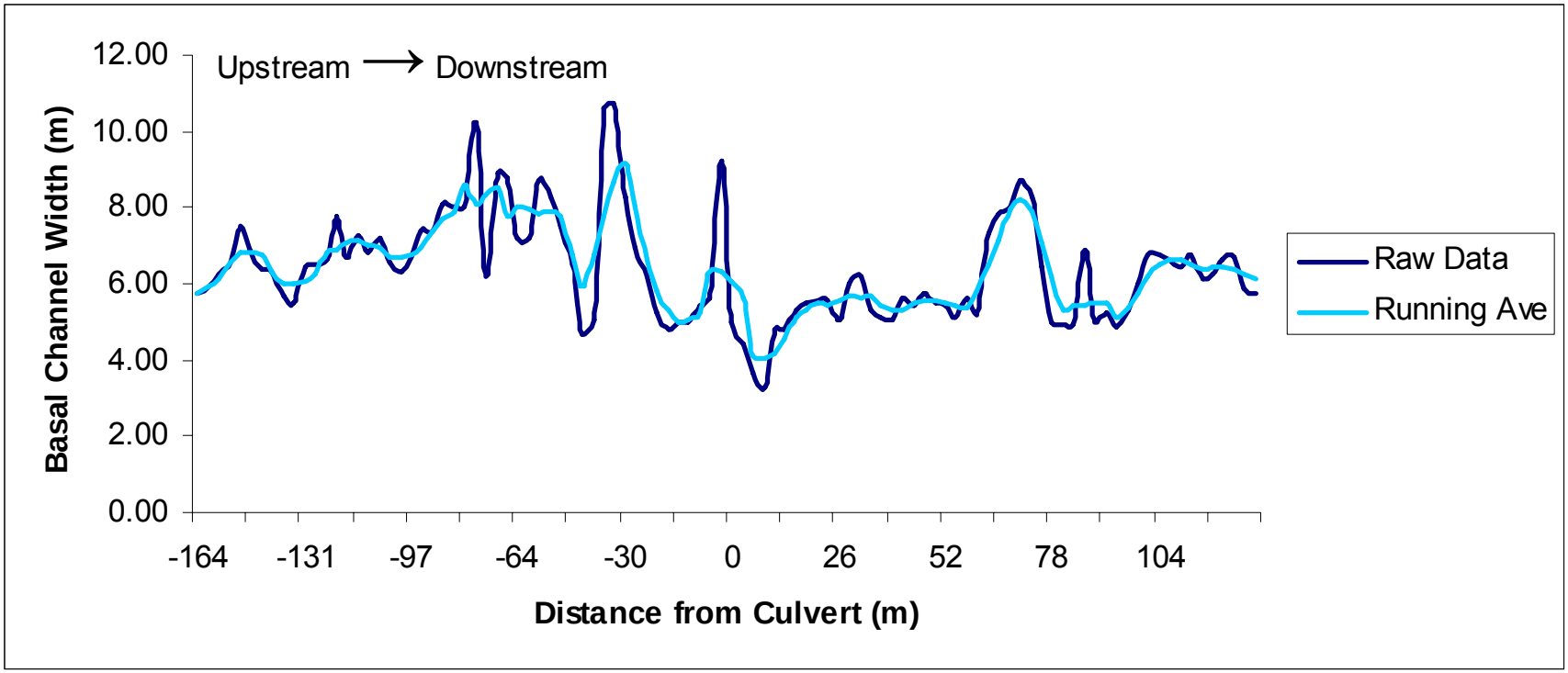

Fig. 6. Change in basal channel width along longitudinal transect at Site 6. Culvert is located at 0 on the x-axis.

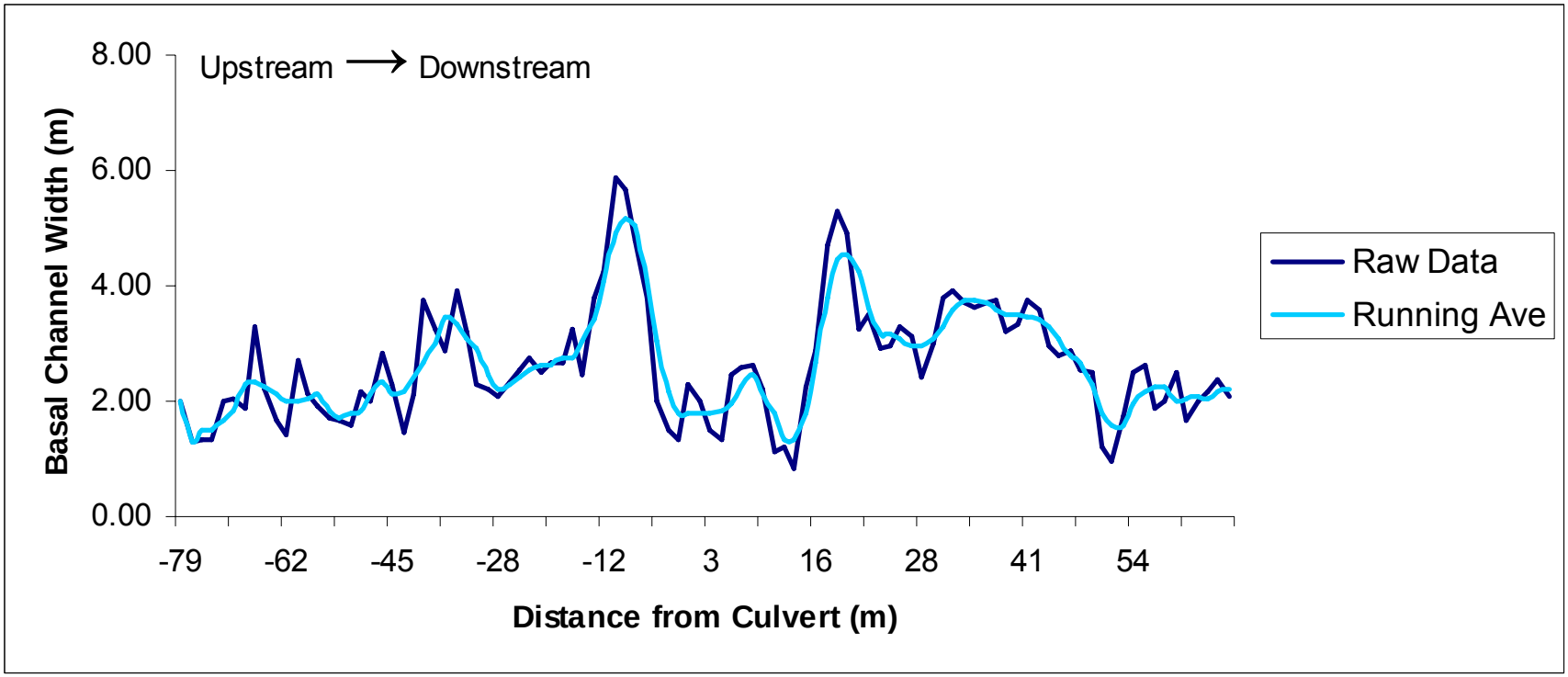

Fig. 7. Change in basal channel width along longitudinal transect at Site 7. Culvert is located at 0 on the x-axis.

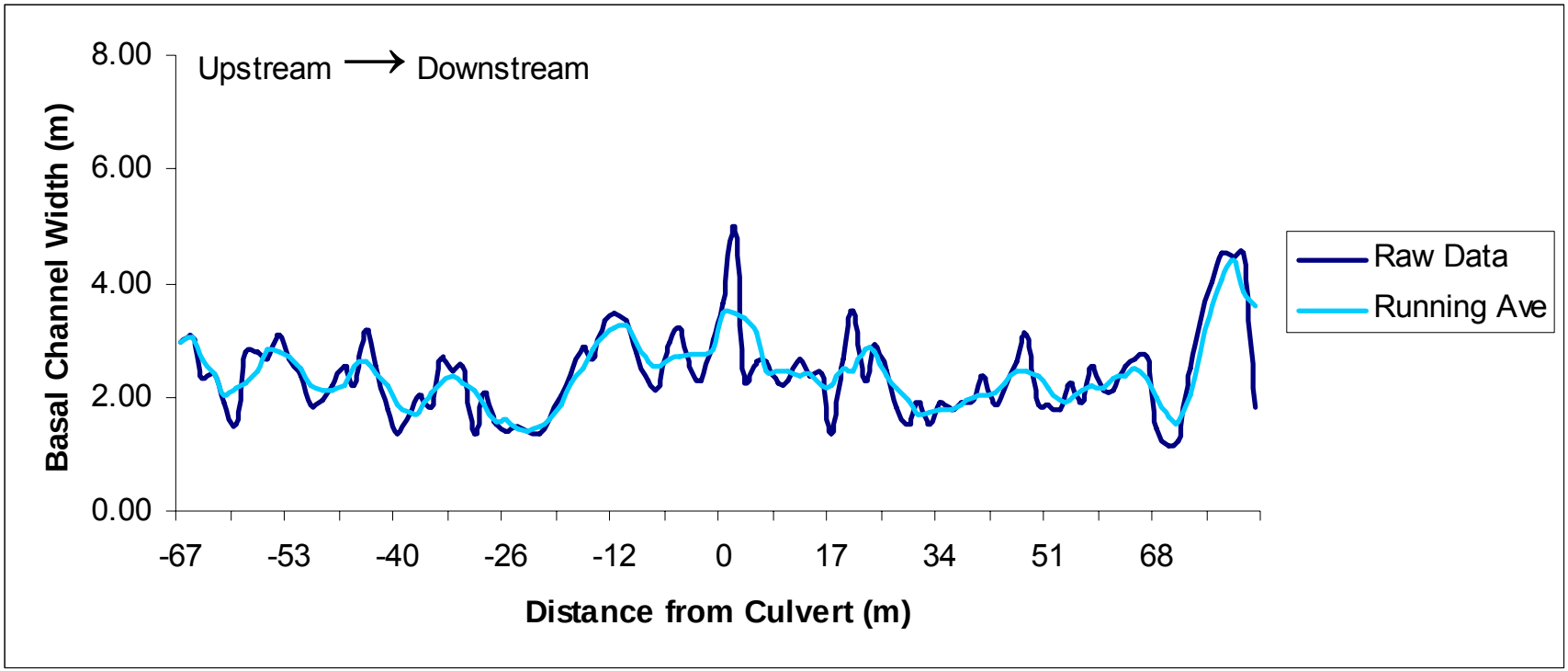

Fig. 8. Change in basal channel width along longitudinal transect at Site 8. Culvert is located at 0 on the x-axis.

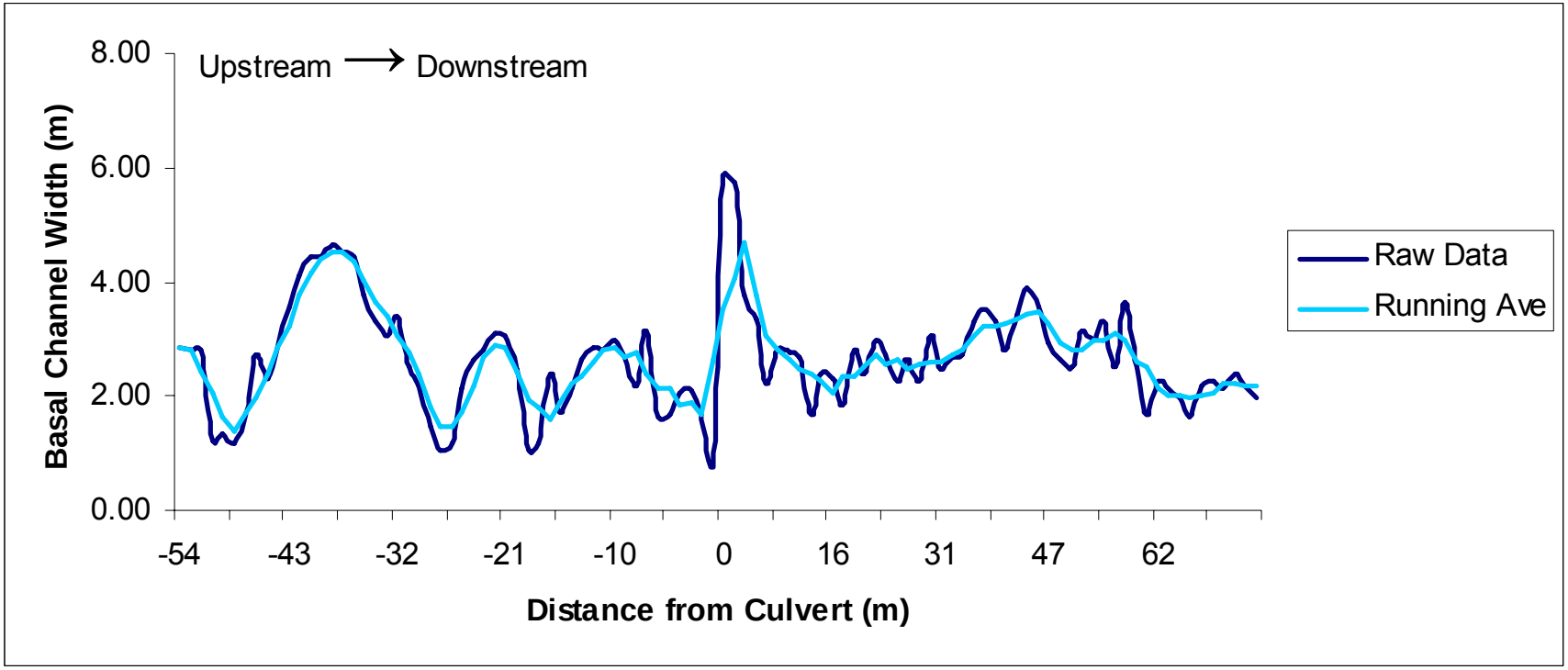

Fig. 9. Change in basal channel width along longitudinal transect at Site 9. Culvert is located at 0 on the x-axis.

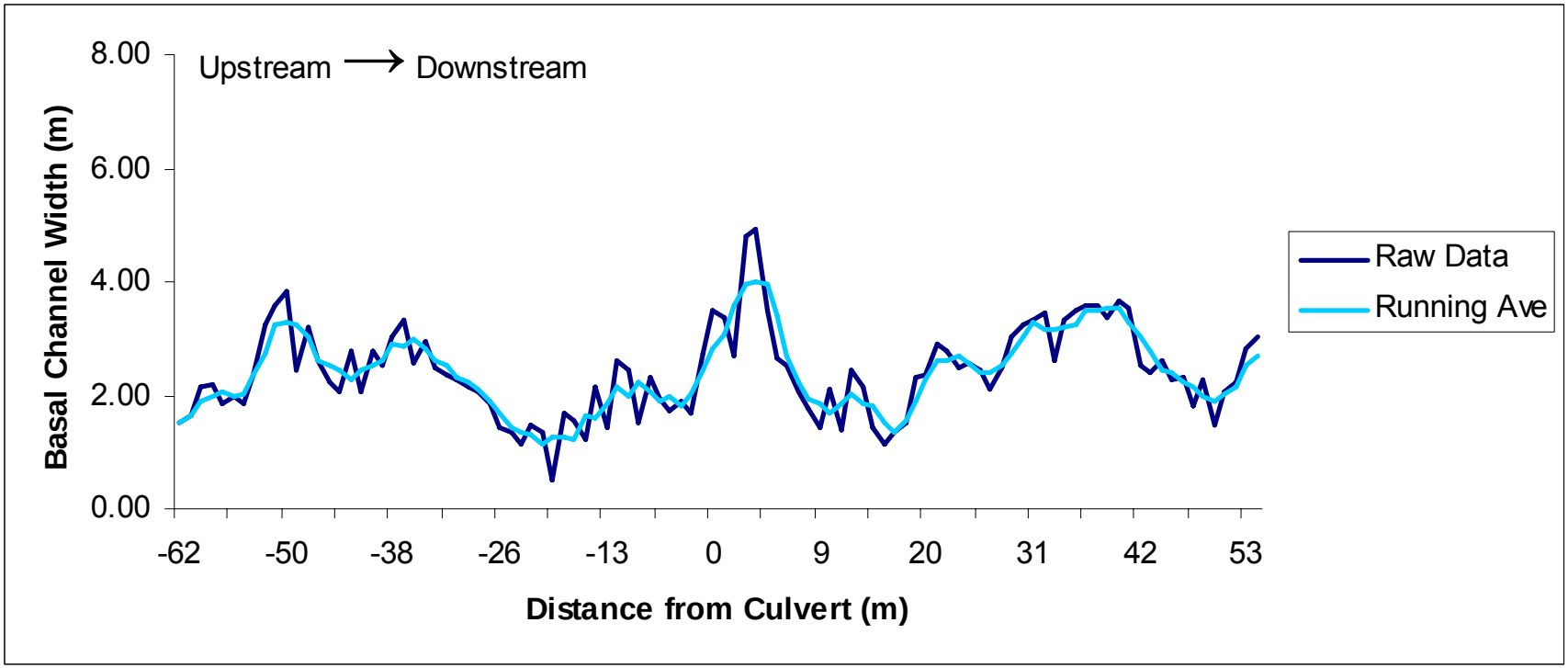

Fig. 10. Change in basal channel width along longitudinal transect at Site 10. Culvert is located at 0 on the x-axis.

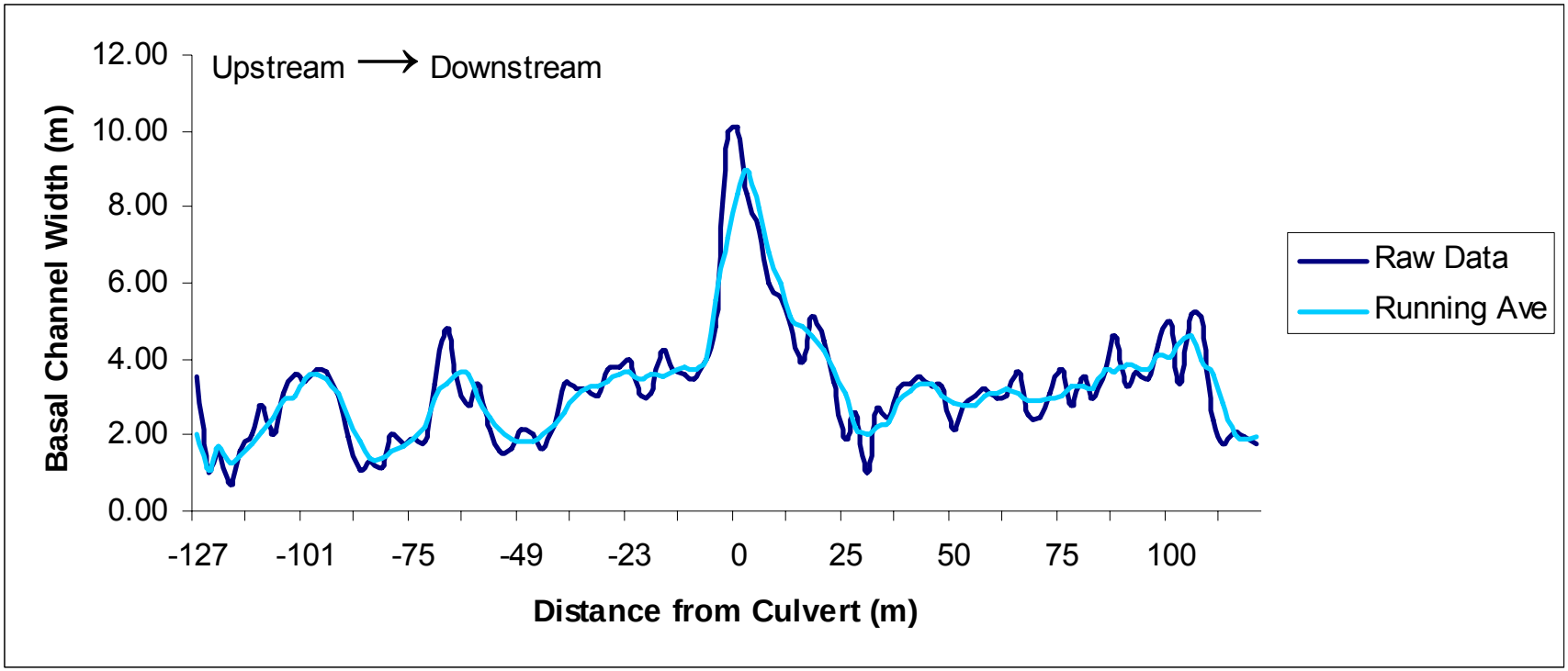

Fig. 11. Change in basal channel width along longitudinal transect at Site 11. Culvert is located at 0 on the x-axis.

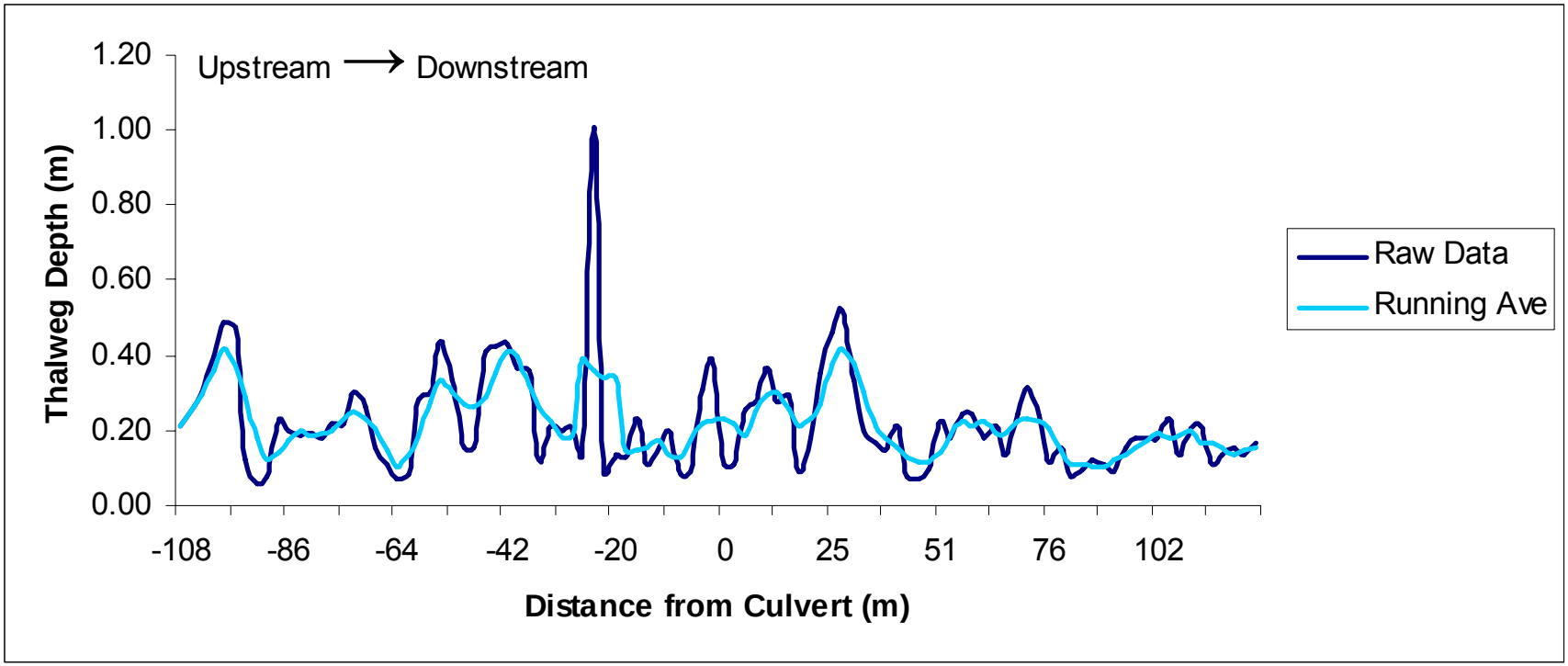

Fig. 12. Change in thalweg depth along longitudinal transect at Site 1. Culvert is located at 0 on the x-axis.

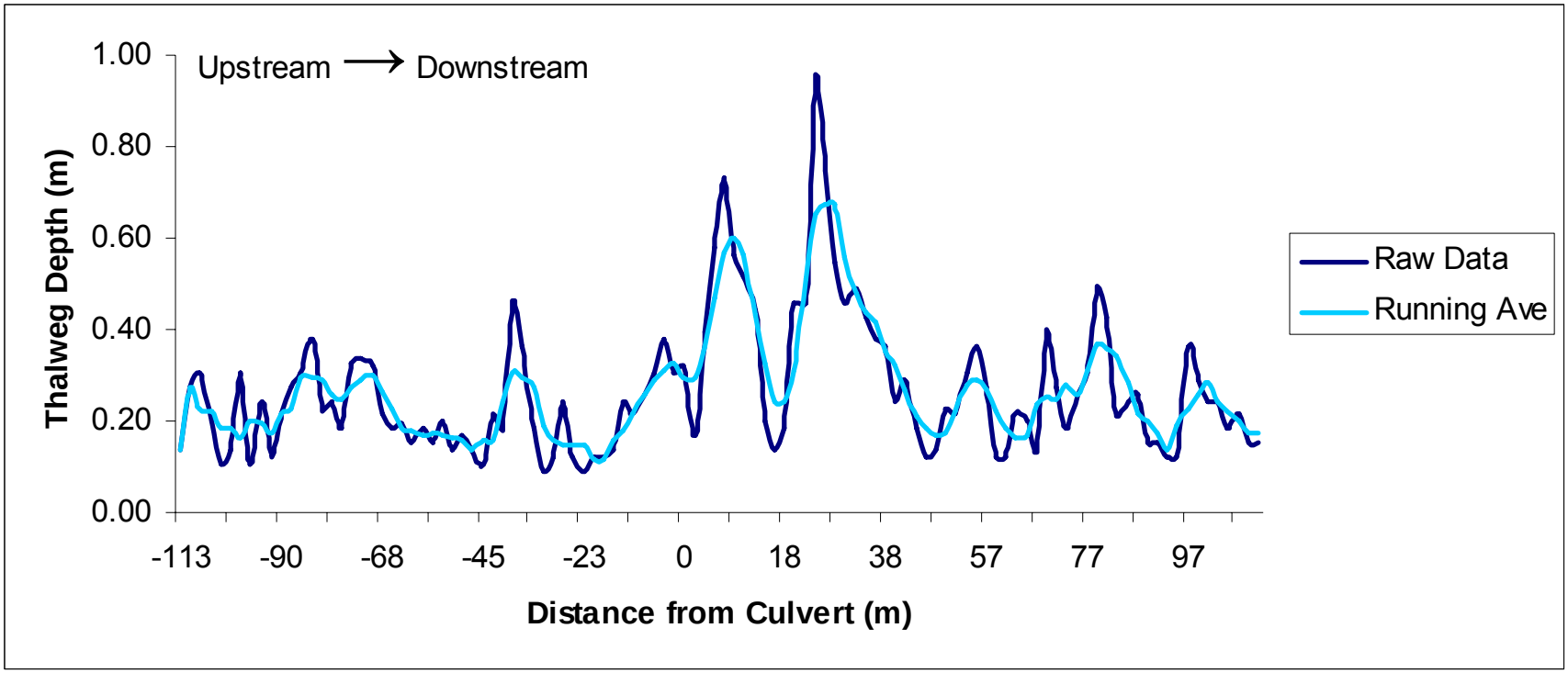

Fig. 13. Change in thalweg depth along longitudinal transect at Site 2. Culvert is located at 0 on the x-axis.

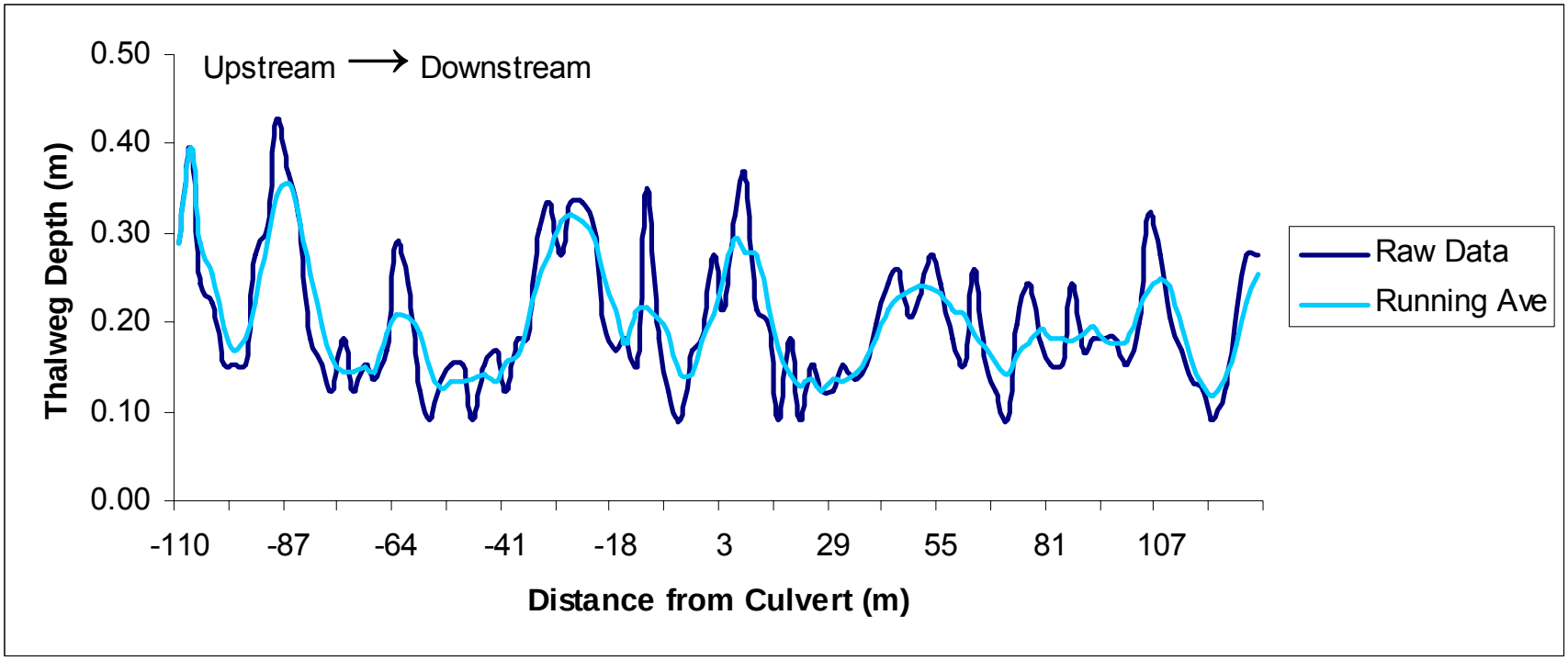

Fig. 14. Change in thalweg depth along longitudinal transect at Site 3. Culvert is located at 0 on the x-axis.

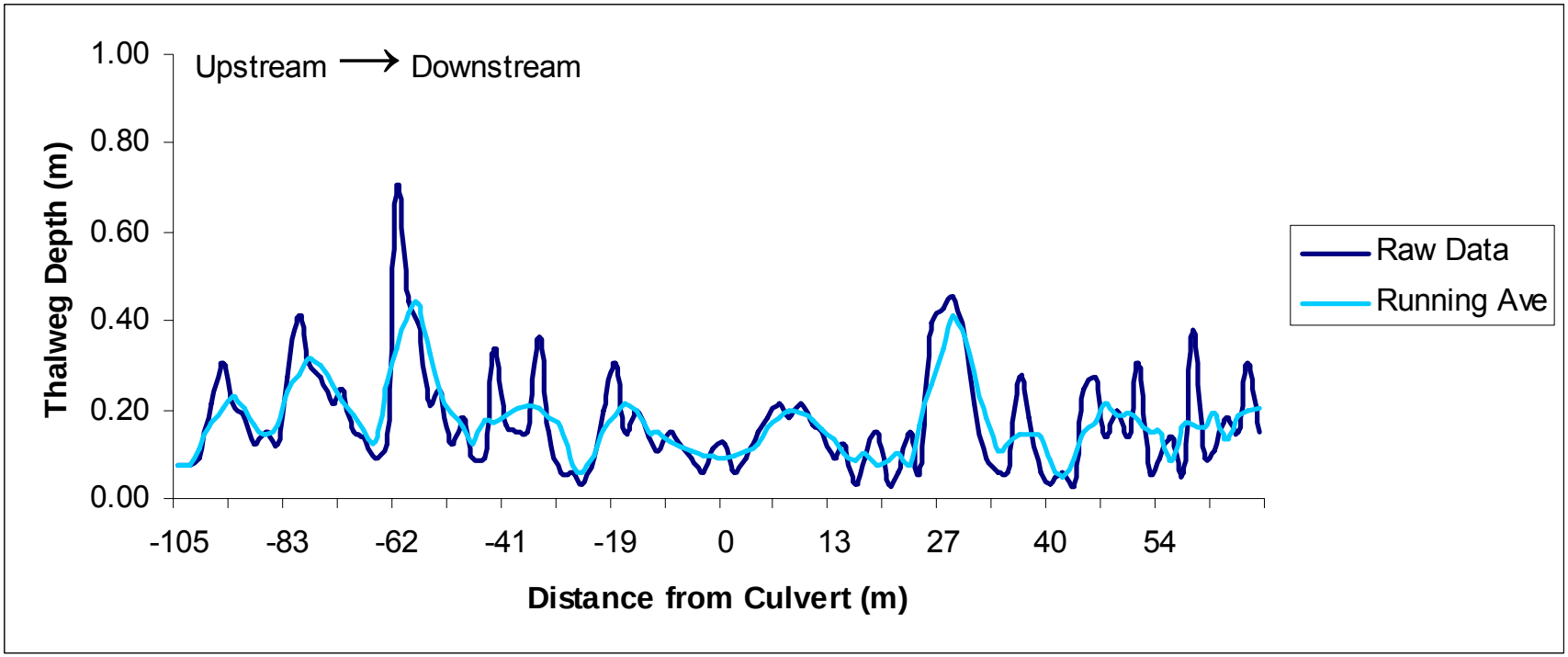

Fig. 15. Change in thalweg depth along longitudinal transect at Site 4. Culvert is located at 0 on the x-axis.

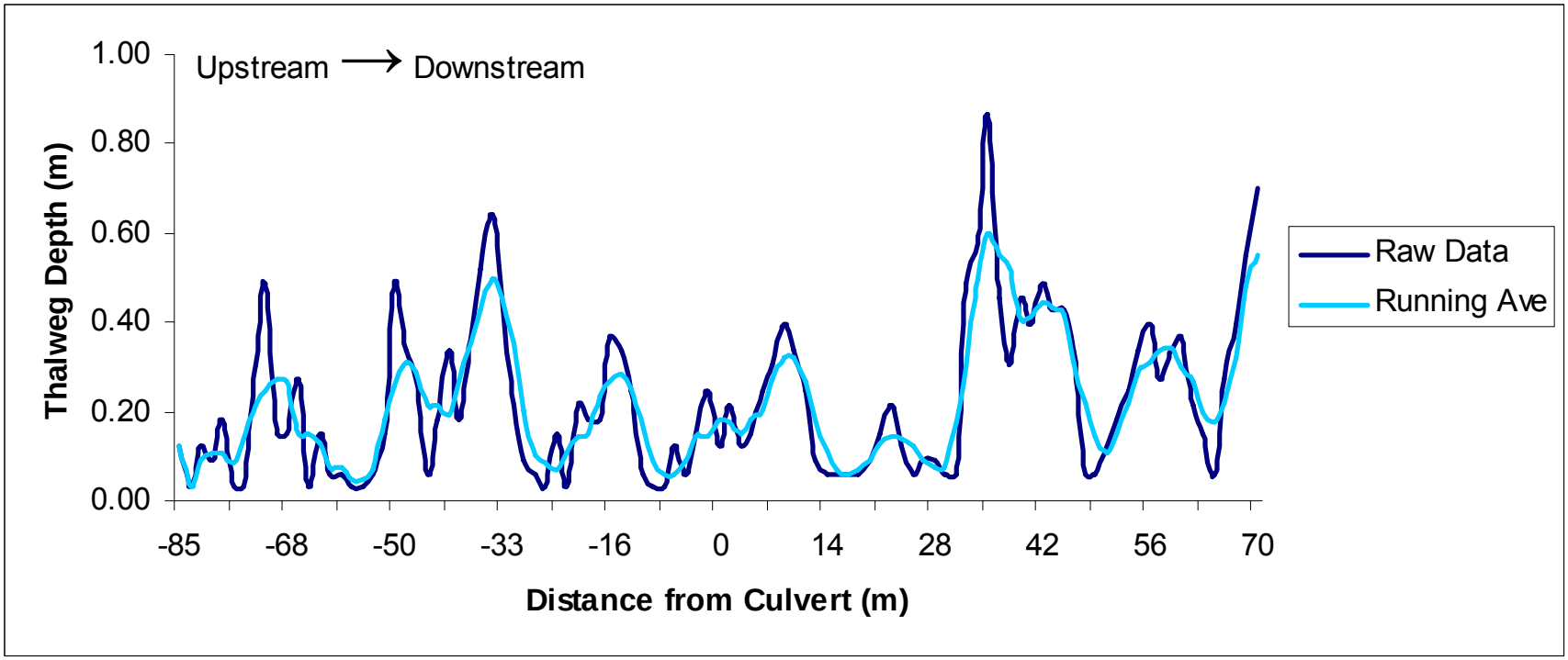

Fig. 16. Change in thalweg depth along longitudinal transect at Site 5. Culvert is located at 0 on the x-axis.

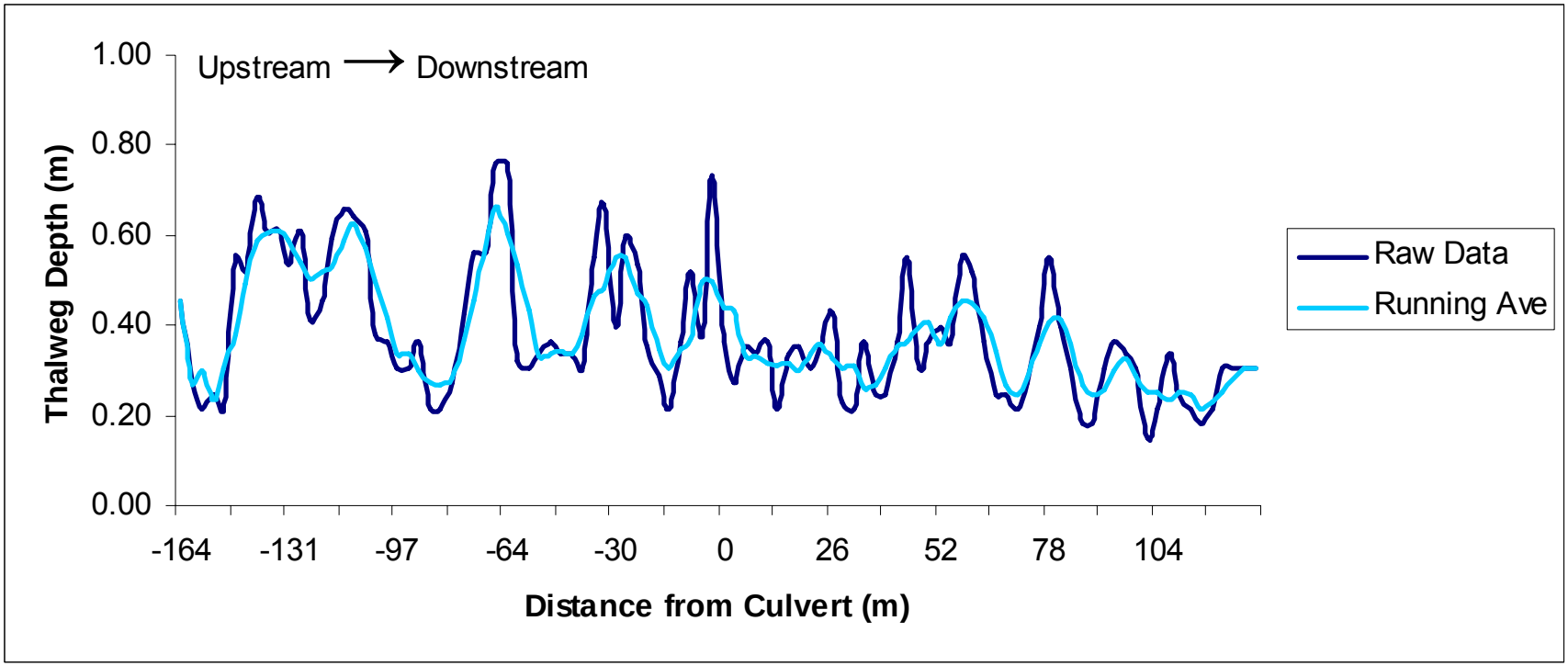

Fig. 17. Change in thalweg depth along longitudinal transect at Site 6. Culvert is located at 0 on the x-axis.

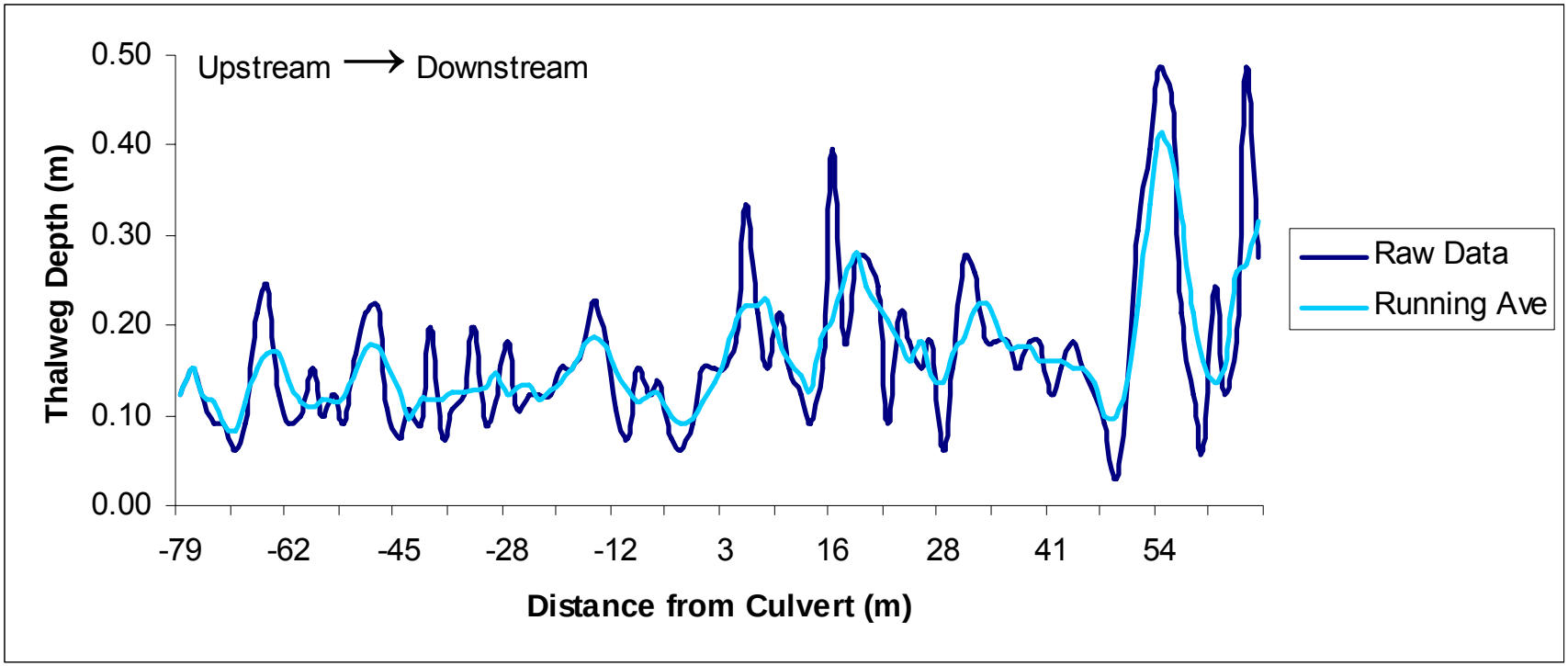

Fig. 18. Change in thalweg depth along longitudinal transect at Site 7. Culvert is located at 0 on the x-axis.

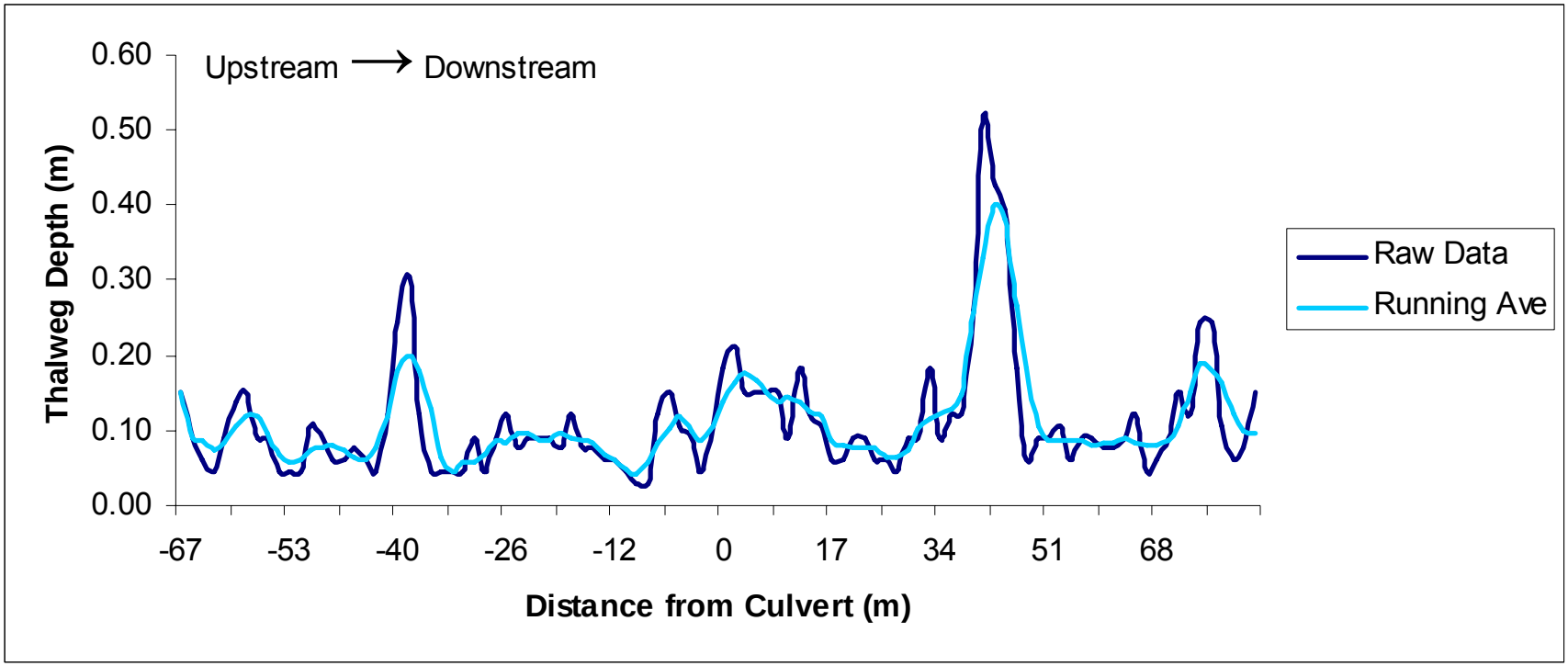

Fig. 19. Change in thalweg depth along longitudinal transect at Site 8. Culvert is located at 0 on the x-axis.

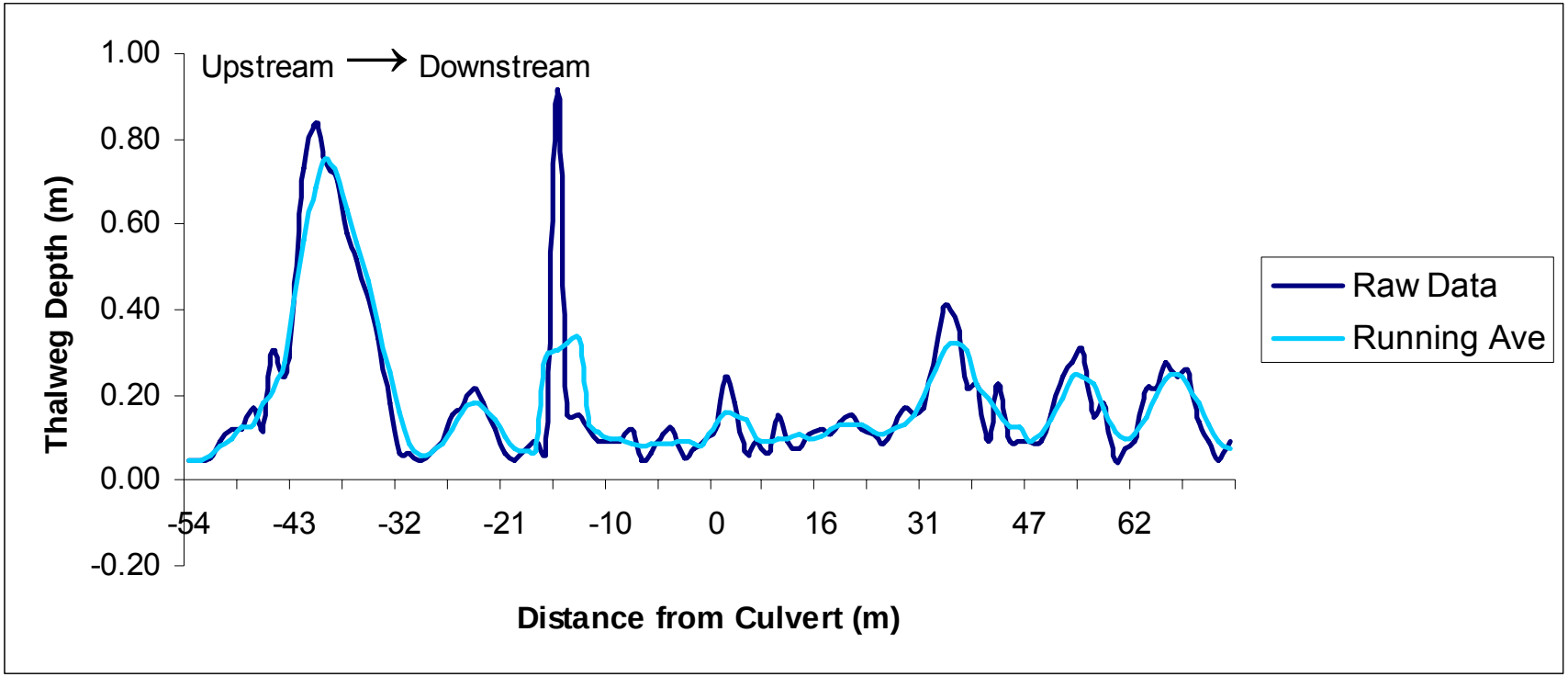

Fig. 20. Change in thalweg depth along longitudinal transect at Site 9. Culvert is located at 0 on the x-axis.

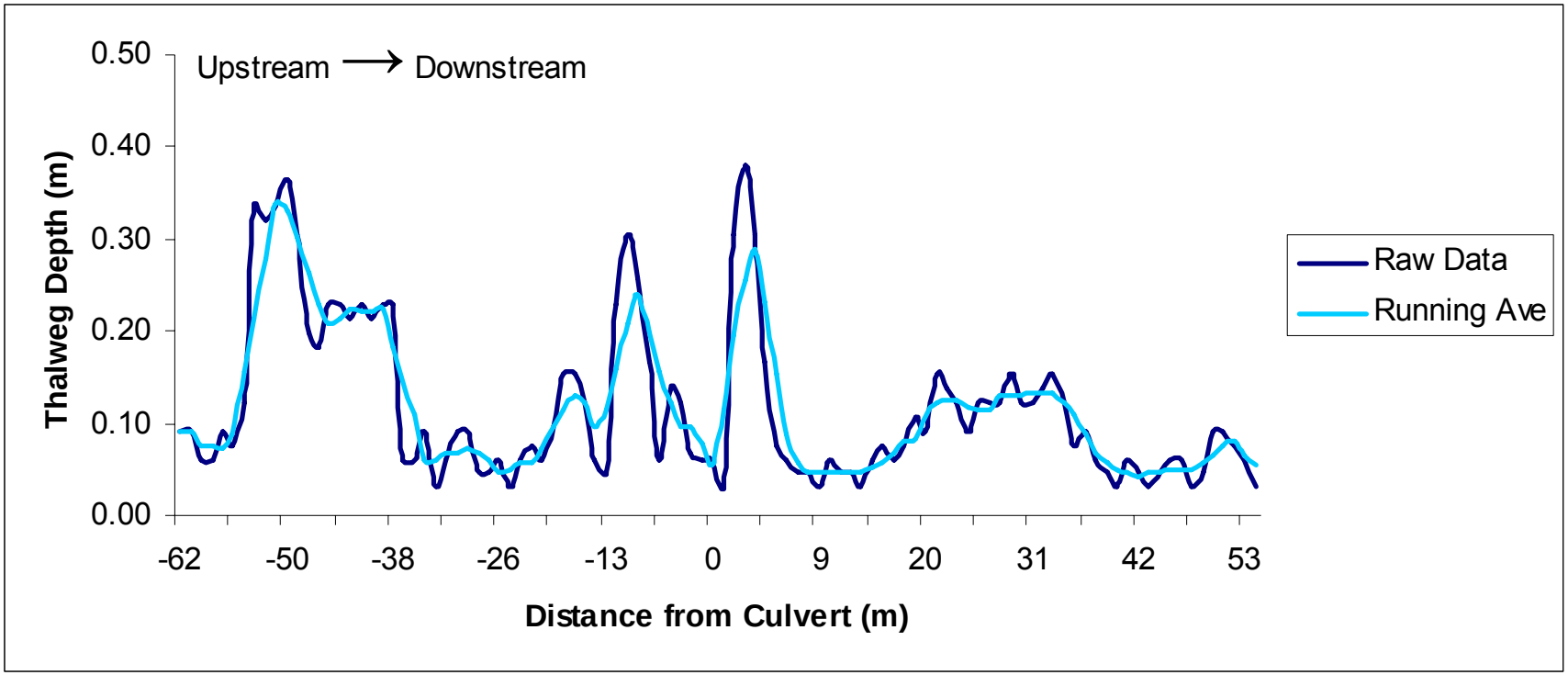

Fig. 21. Change in thalweg depth along longitudinal transect at Site 10. Culvert is located at 0 on the x-axis.

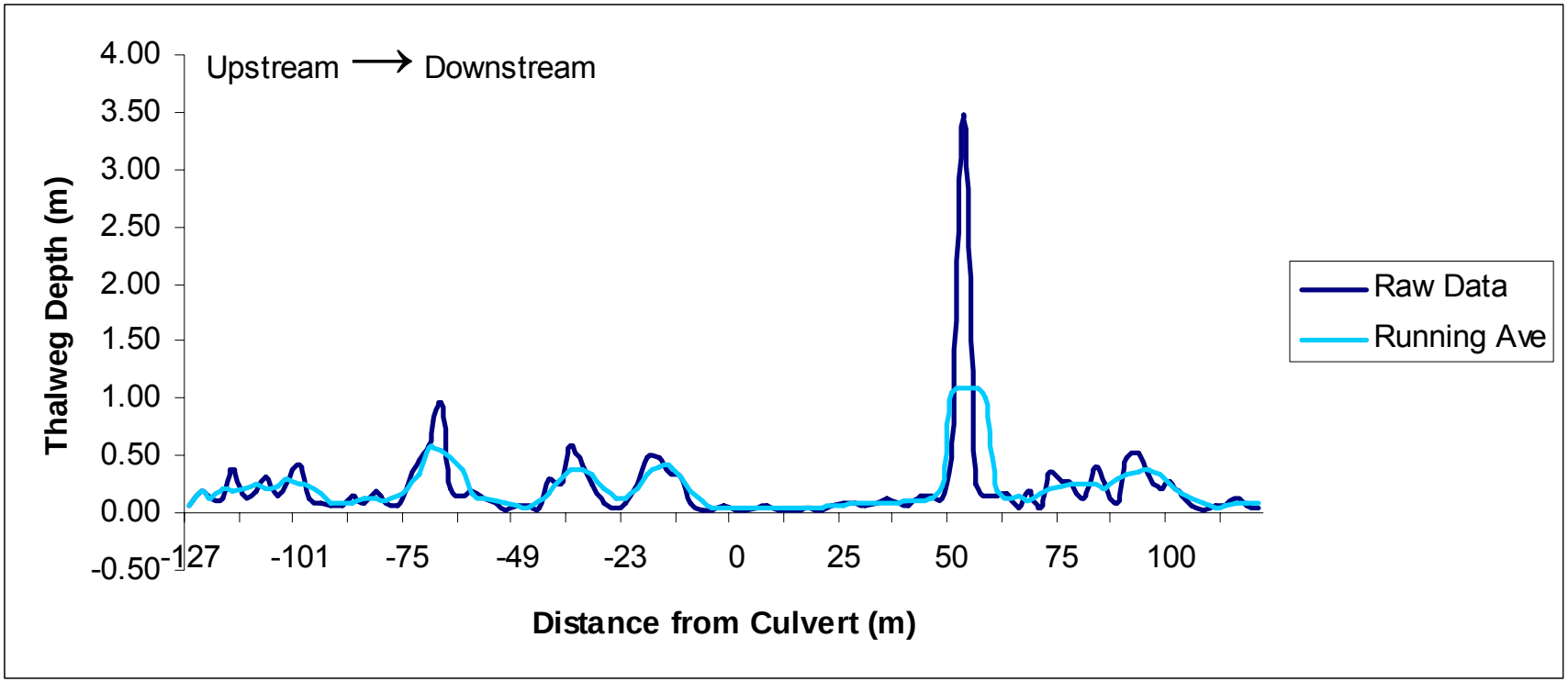

Fig. 22. Change in thalweg depth along longitudinal transect at Site 11. Culvert is located at 0 on the x-axis.

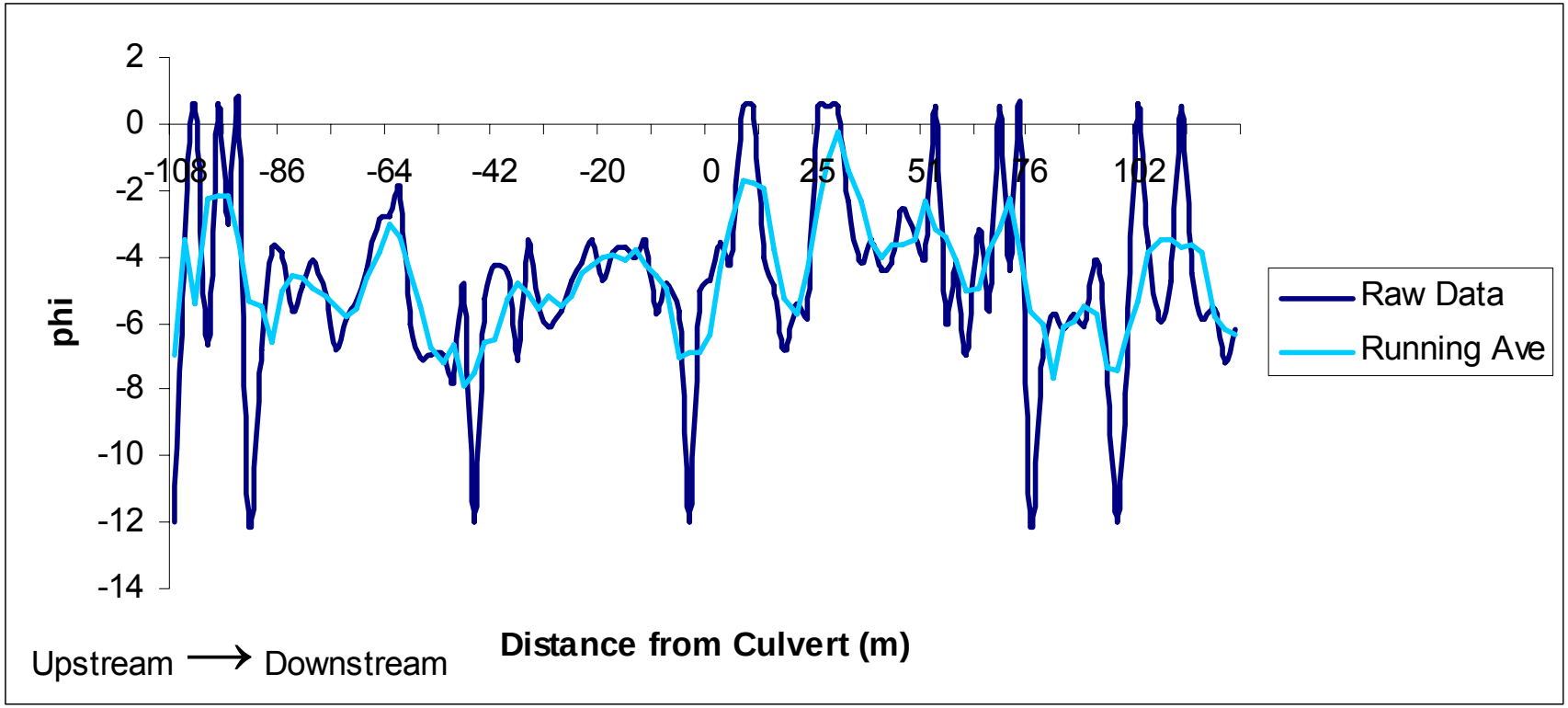

Fig. 23. Change in phi along longitudinal transect at Site 1. Culvert is located at 0 on the x-axis.

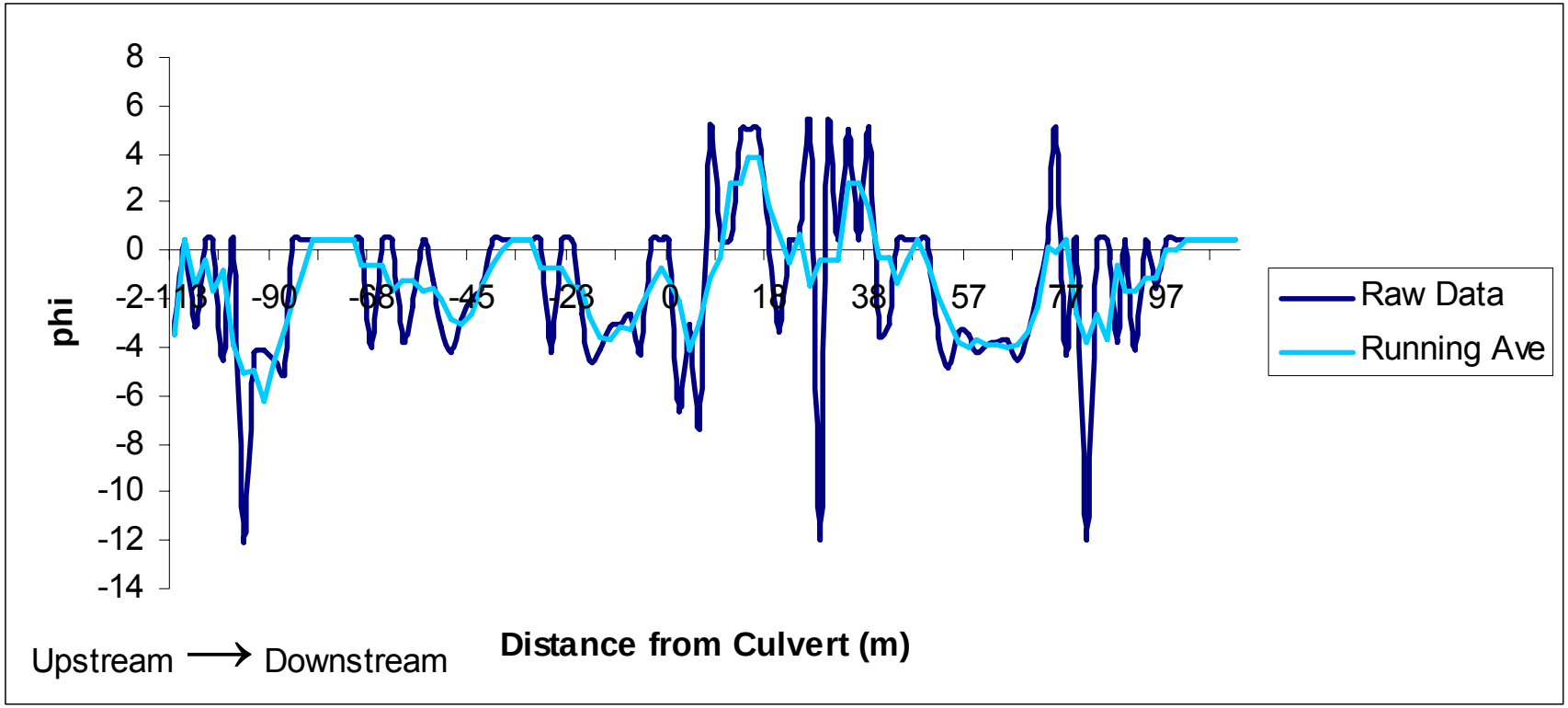

Fig. 24. Change in phi along longitudinal transect at Site 2. Culvert is located at 0 on the x-axis.

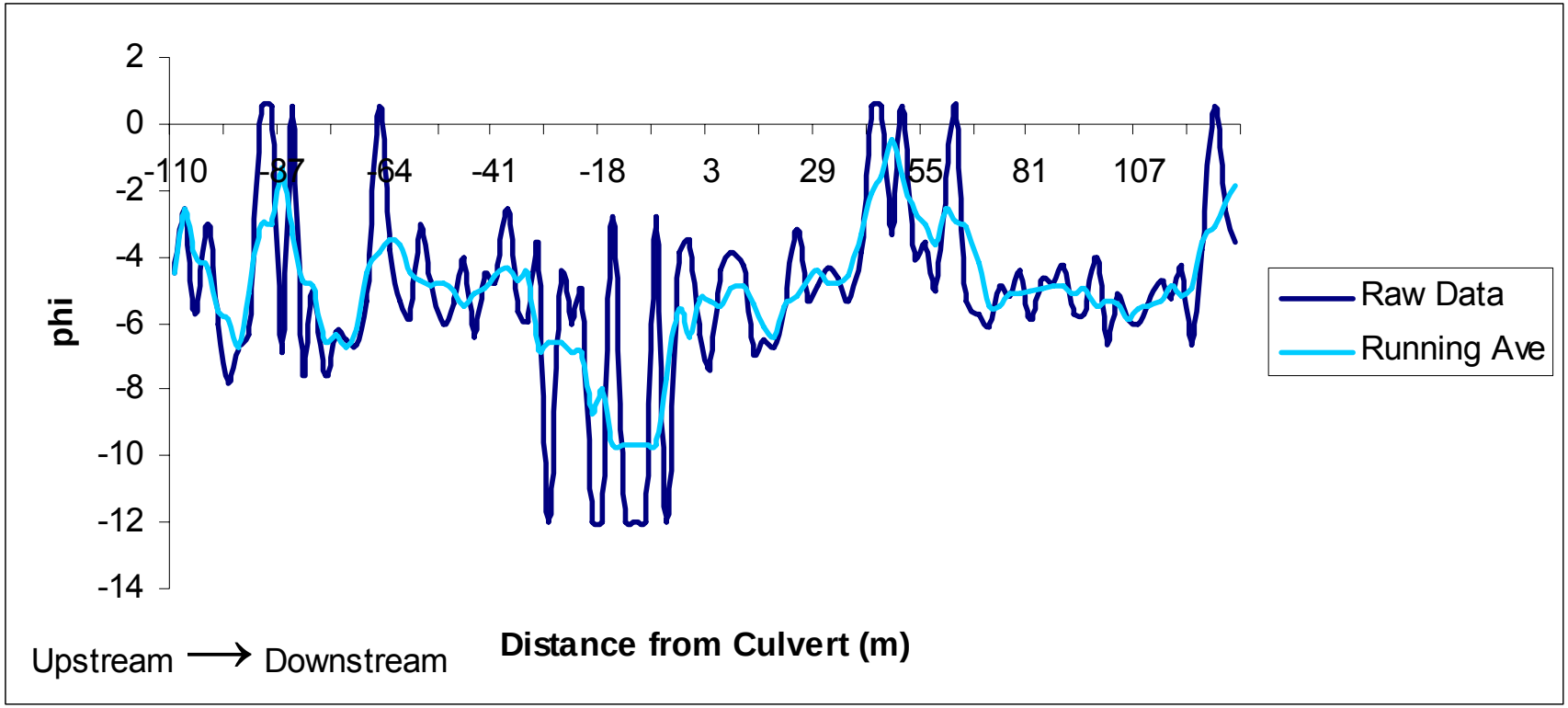

Fig. 25. Change in phi along longitudinal transect at Site 3. Culvert is located at 0 on the x-axis.

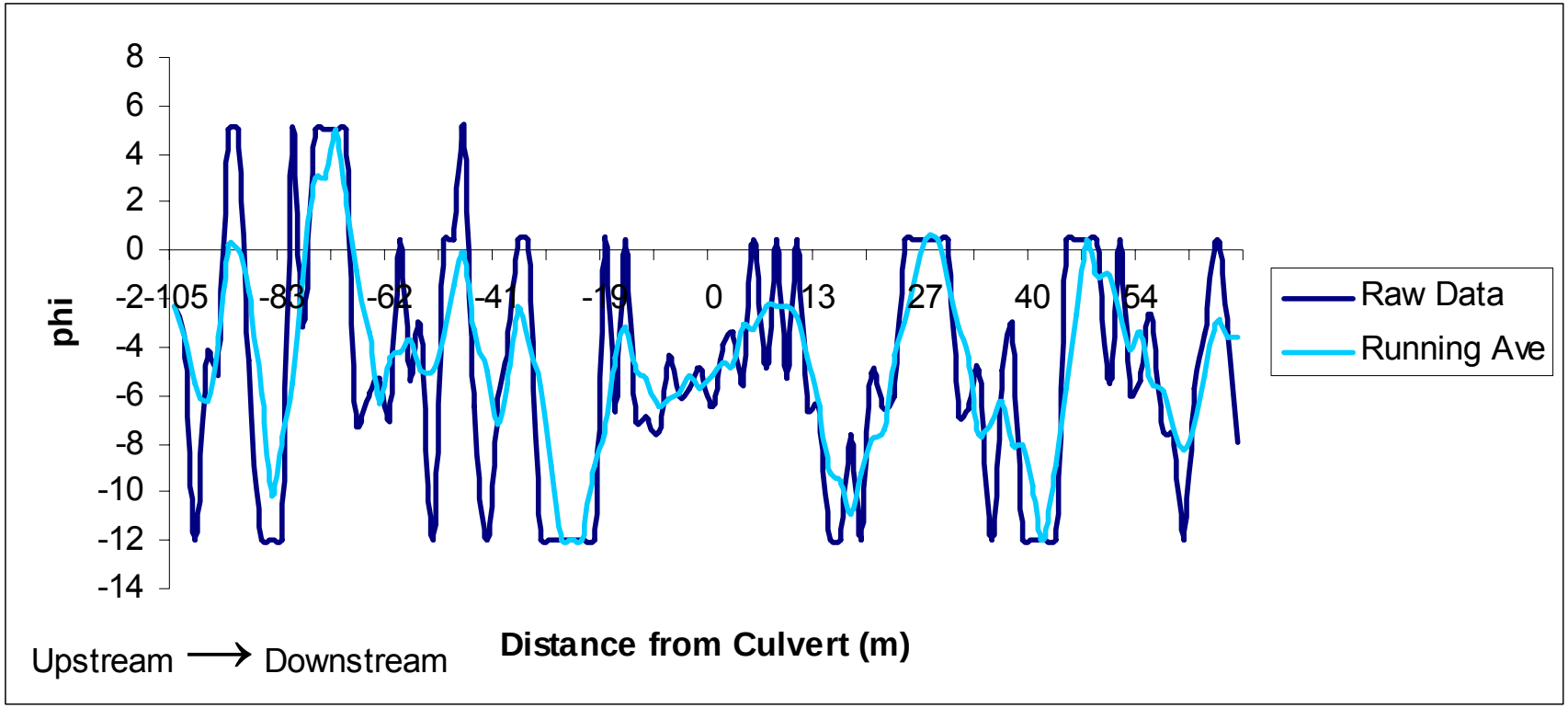

Fig. 26. Change in phi along longitudinal transect at Site 4. Culvert is located at 0 on the x-axis.

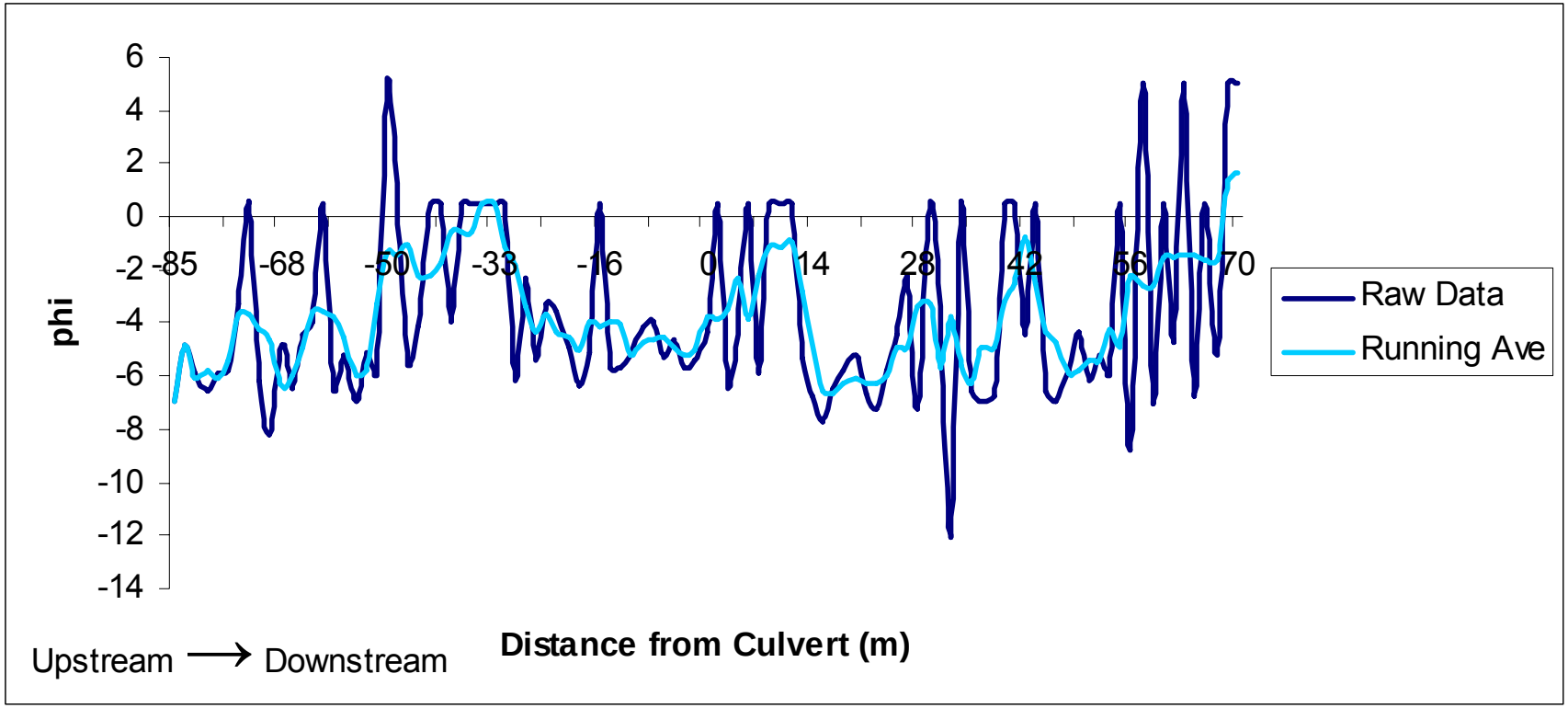

Fig. 27. Change in phi along longitudinal transect at Site 5. Culvert is located at 0 on the x-axis.

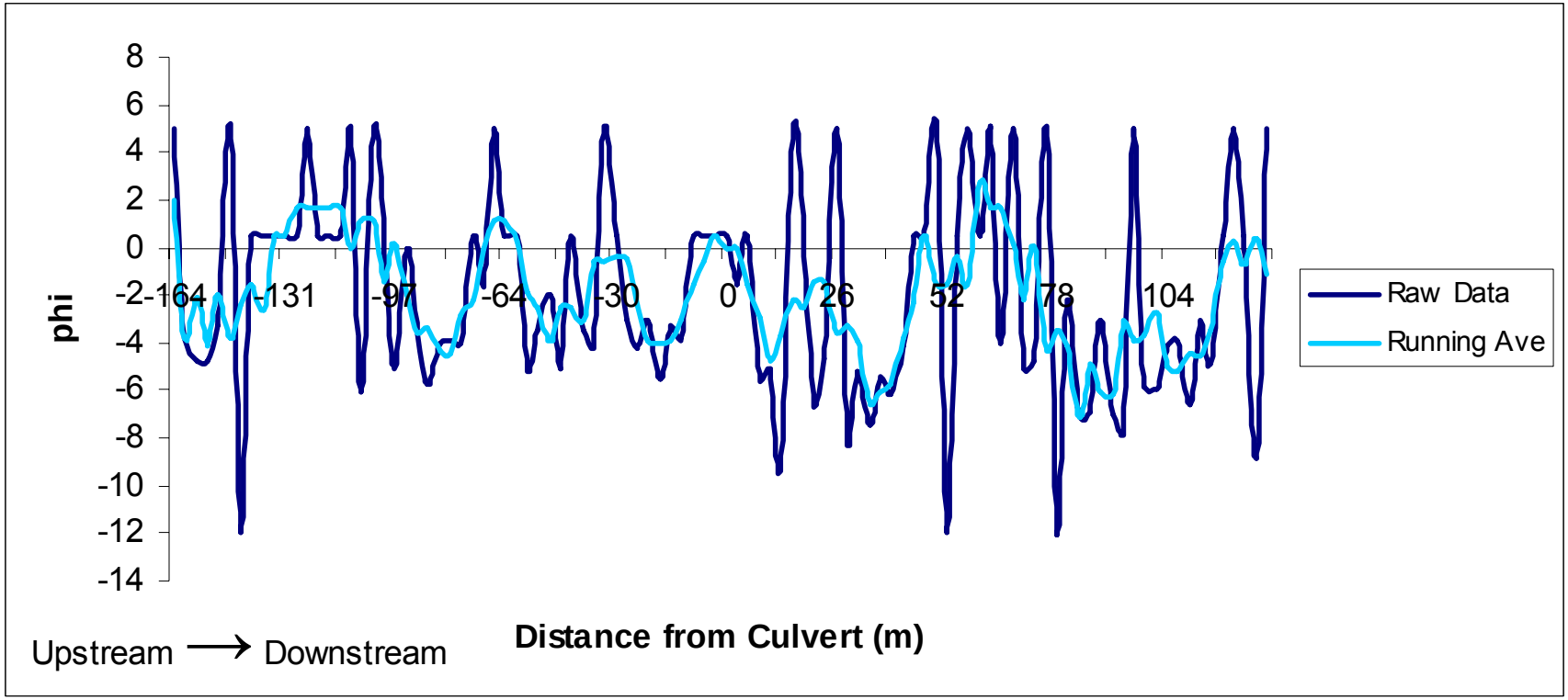

Fig. 28. Change in phi along longitudinal transect at Site 6. Culvert is located at 0 on the x-axis.

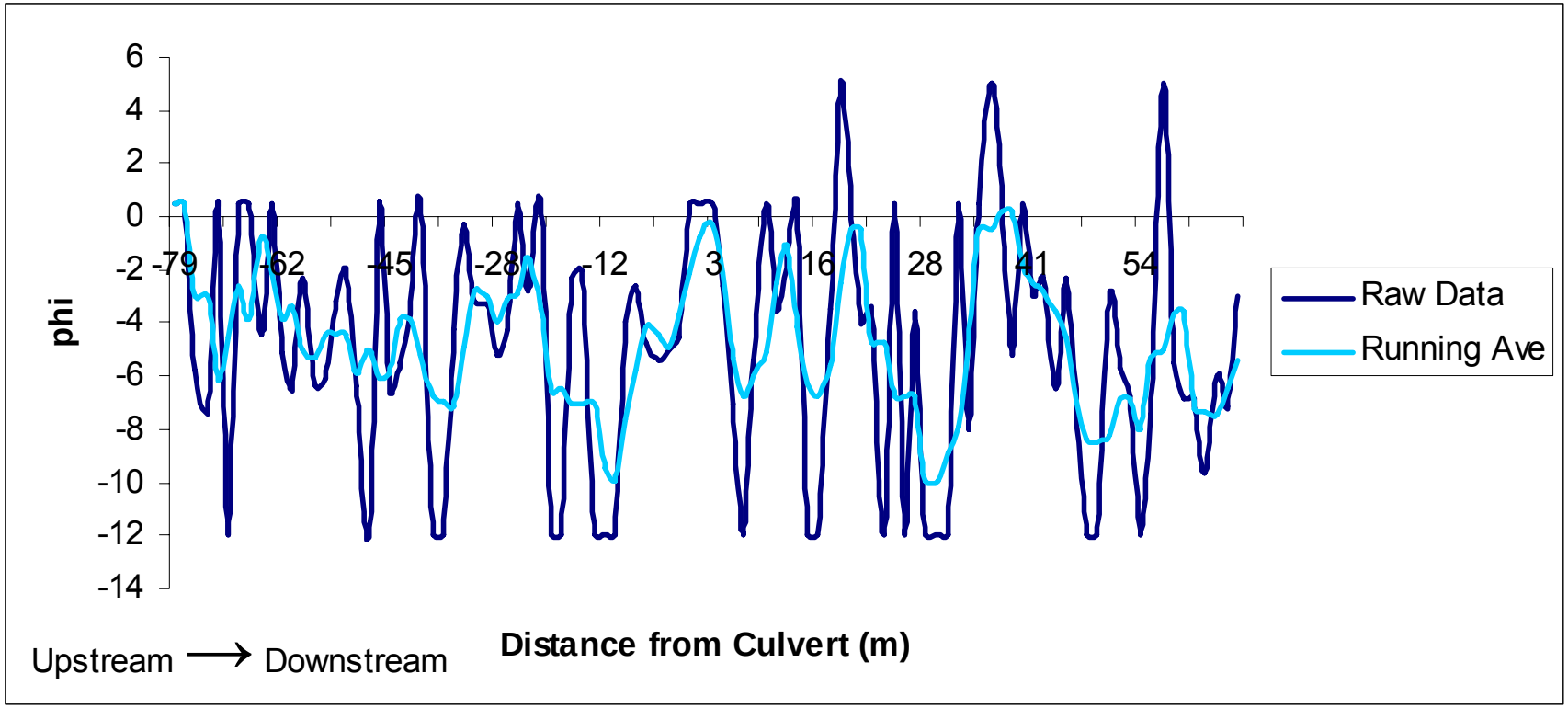

Fig. 29. Change in phi along longitudinal transect at Site 7. Culvert is located at 0 on the x-axis.

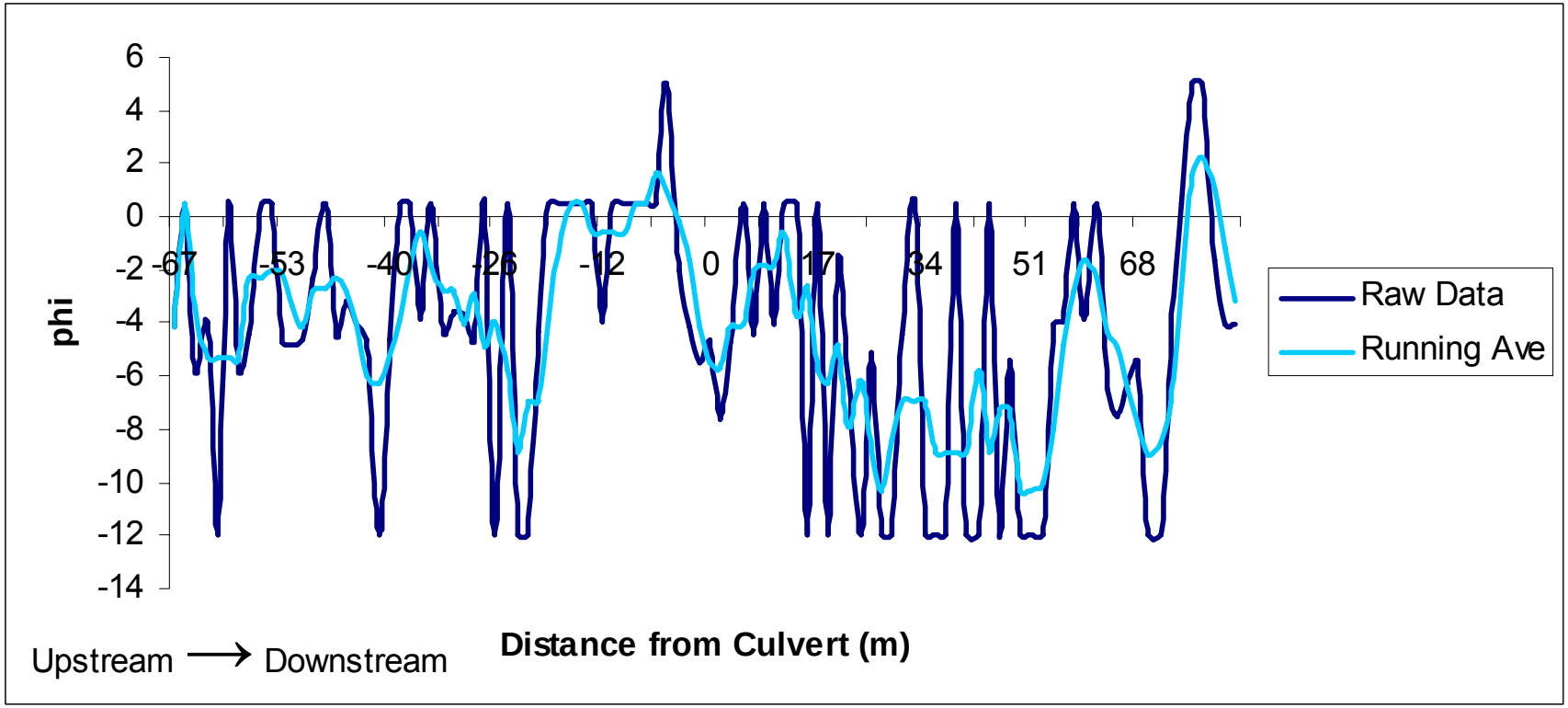

Fig. 30. Change in phi along longitudinal transect at Site 8. Culvert is located at 0 on the x-axis.

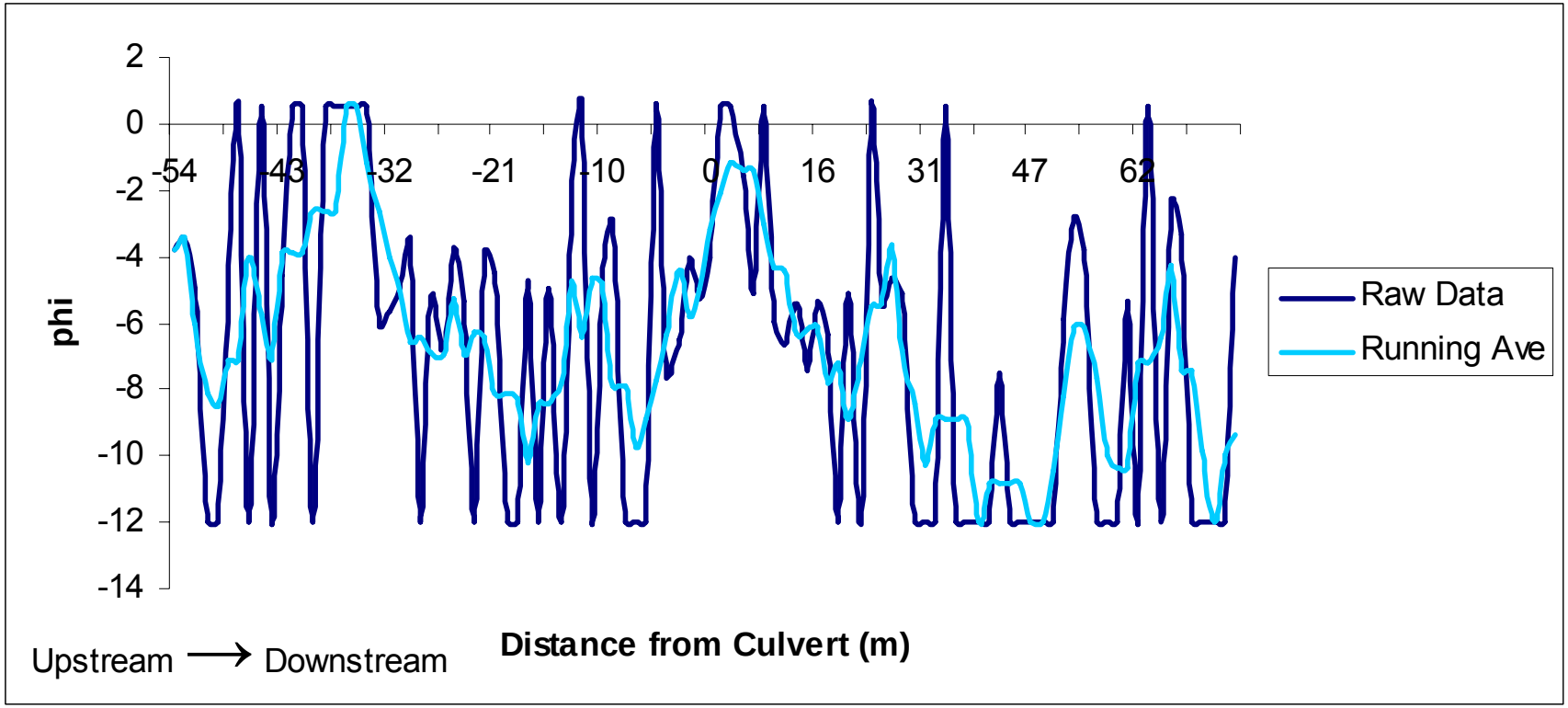

Fig. 31. Change in phi along longitudinal transect at Site 9. Culvert is located at 0 on the x-axis.

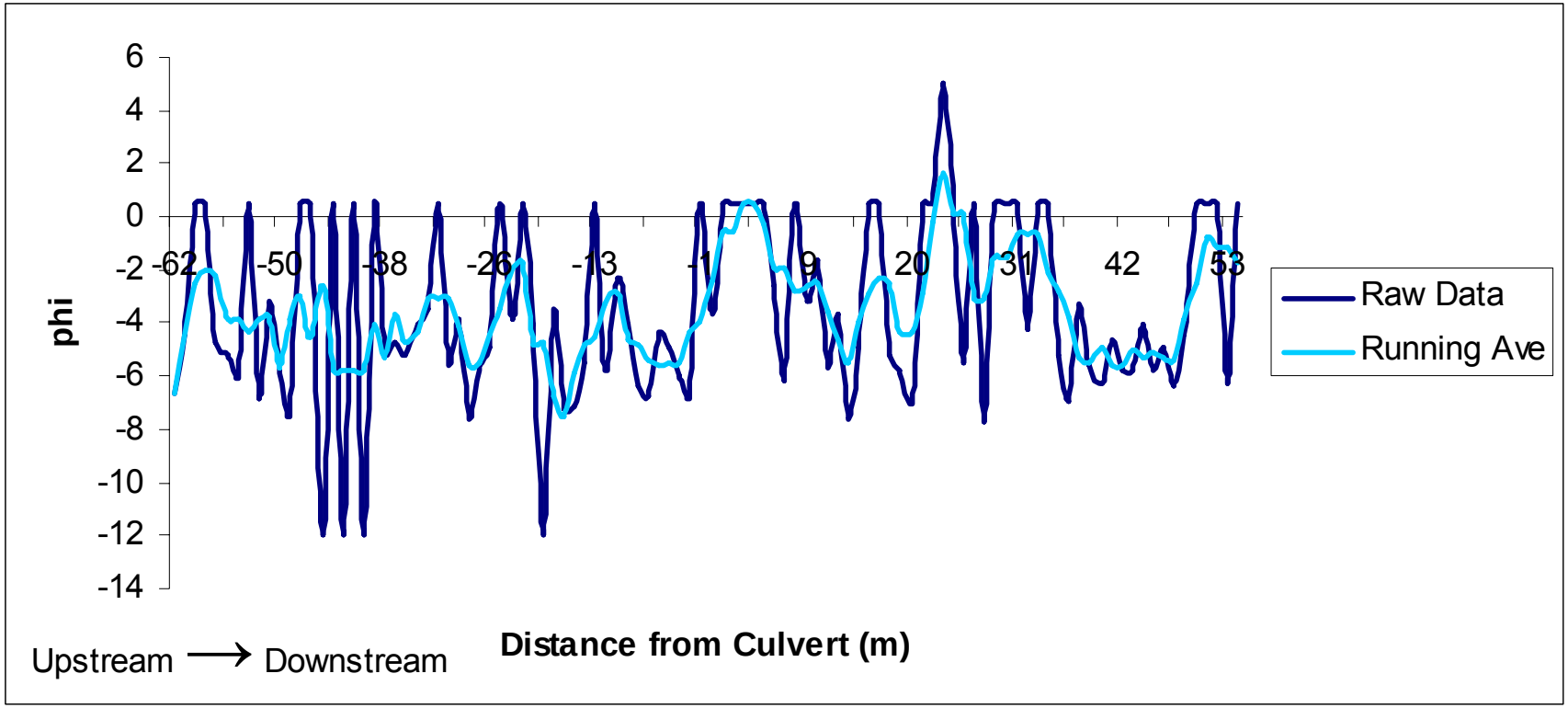

Fig. 32. Change in phi along longitudinal transect at Site 10. Culvert is located at 0 on the x-axis.

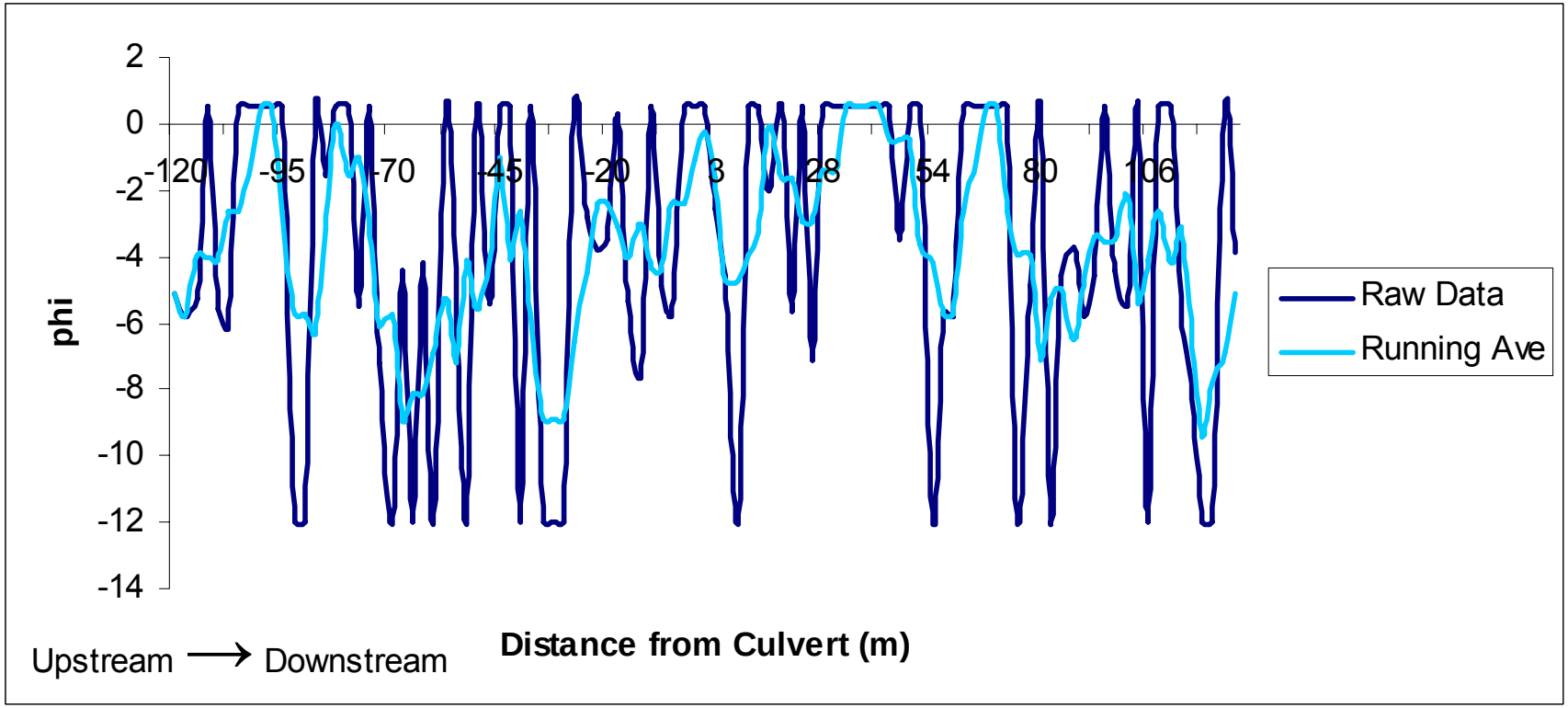

Fig. 33. Change in phi along longitudinal transect at Site 11. Culvert is located at 0 on the x-axis.

APPENDIX B: SUPPLEMENTAL MATERIALS FOR CHAPTER 3

Table 1. MARK output from the best supported model for Canton Creek (RMCS model) showing estimates for all parameters in the model. Parameter estimates are shown for both benthic fishes (first estimate in each group) and water column fishes (second estimate in each group). Parameters that could not be estimated have values listed as a number with a negative exponent or a value of 1. Parameters that could not be estimated were not included in the total number of parameters estimated for AIC calculations.

Parameter	Estimate	Standard Error	Lower	Upper
1:S A:A	0.8227364	0.0846427	0.5980777	0.9353863
2:S A:A	0.8162184	0.0337279	0.7408211	0.8734295
3:p A:A	0.2339016	0.1232484	0.0734745	0.5403327
4:p A:A	0.0797816	0.0413745	0.0279255	0.2073879
5:p B:B	0.1563607	0.0848449	0.0499156	0.3953447
6:p B:B	0.3318880	0.0555196	0.2331857	0.4479624
7:p C:C	0.1225270	0.0930273	0.0249743	0.4322168
8:p C:C	0.1291653	0.0381302	0.0709109	0.2237513
9:p D:D	0.3970585	0.1392572	0.1739569	0.6731278
10:p D:D	0.2638336	0.0511469	0.1762042	0.3751946
11:p E:E	0.1657084	0.0794659	0.0604852	0.3799545
12:p E:E	0.2541767	0.0401990	0.1836094	0.3405518
13:p F:F	0.2690194	0.0995697	0.1200434	0.4982032
14:p F:F	0.1717535	0.0723341	0.0711022	0.3597105
15:p G:G	0.2687467	0.1156833	0.1038880	0.5381185
16:p G:G	0.3330259	0.0585567	0.2294818	0.4556611
17:p H:H	0.2966507	0.1238891	0.1163886	0.5745614
18:p H:H	0.2062807	0.0408888	0.1374097	0.2977549
19:p I:I	0.2361100E-08	0.6287496E-05	-0.1232113E-04	0.1232585E-04
20:p I:I	0.2704143	0.0495435	0.1847161	0.3774639
21:p J:J	0.1089314	0.0814898	0.0230405	0.3878845
22:p J:J	0.2168349	0.0330820	0.1589537	0.2885614
23:Psi A to B	0.0639661	0.0451534	0.0153465	0.2305523
24:Psi A to B	0.1572473	0.0235932	0.1163148	0.2091748
25:Psi A to C	0.0073034	0.0160685	0.9552098E-04	0.3616772
26:Psi A to C	0.0116071	0.0057537	0.0043750	0.0304284
27:Psi A to F	0.0054133	0.0055013	0.7340712E-03	0.0387626
28:Psi A to F	0.2061374E-08	0.1378062E-05	-0.2698940E-05	0.2703063E-05
29:Psi B to A	0.0679281	0.0337594	0.0249882	0.1716648
30:Psi B to A	0.0911573	0.0172007	0.0625899	0.1309421
31:Psi C to A	0.2077064E-08	0.4134281E-05	-0.8101114E-05	0.8105268E-05
32:Psi C to A	0.0054975	0.0034133	0.0016234	0.0184458
33:Psi F to A	0.2074201E-08	0.2286786E-05	-0.4480027E-05	0.4484176E-05
34:Psi F to A	0.2061206E-08	0.1224965E-05	-0.2398871E-05	0.2402993E-05

Table 2. MARK output from the best supported model for Bluff Creek (FMCS model) showing estimates for all parameters in the model. Parameter estimates are shown for both benthic fishes (first estimate in each group) and water column fishes (second estimate in each group). Parameters that could not be estimated have values listed as a number with a negative exponent or a value of 1. Parameters that could not be estimated were not included in the total number of parameters estimated for AIC calculations.

Parameter	Estimate	Standard Error	Lower	Upper
1:S A:A	0.8390119	0.1275764	0.4500784	0.9707487
2:S A:A	0.8103276	0.0575567	0.6722005	0.8989966
3:p A:A	0.2208073	0.0960565	0.0866503	0.4584219
4:p A:A	0.1356707	0.0411194	0.0731684	0.2378619
5:p B:B	0.2194581	0.0795778	0.1016186	0.4113748
6:p B:B	0.2699602	0.0542622	0.1773409	0.3881296
7:p C:C	0.2141088	0.1203666	0.0628325	0.5254084
8:p C:C	0.2784129	0.0703221	0.1626802	0.4338216
9:p D:D	0.1854638	0.0992139	0.0591338	0.4520180
10:p D:D	0.1816033	0.0566126	0.0951675	0.3188780
11:p E:E	0.3052756	0.1296067	0.1171102	0.5927836
12:p E:E	0.3610011	0.0830545	0.2181114	0.5336150
13:p F:F	0.3680608	0.1362249	0.1559757	0.6473454
14:p F:F	0.3159728	0.0583203	0.2139603	0.4394322
15:p G:G	0.1510354	0.0941130	0.0405007	0.4285149
16:p G:G	0.1765631	0.0518400	0.0963313	0.3013353
17:p H:H	0.2395854	0.1091892	0.0886971	0.5049349
18:p H:H	0.3263662	0.0734416	0.2011043	0.4825255
19:p I:I	0.2551405	0.1005575	0.1082761	0.4914281
20:p I:I	0.1597377	0.0549284	0.0785447	0.2977414
21:p J:J	0.0539818	0.0553495	0.0067734	0.3231617
22:p J:J	0.0538324	0.0277567	0.0191776	0.1420408
23:Psi A to B	0.0560510	0.0330371	0.0171636	0.1679866
24:Psi A to B	0.1293063	0.0255572	0.0869047	0.1881336
25:Psi A to C	0.0219009	0.0239268	0.0025016	0.1666120
26:Psi A to C	0.0257907	0.0122736	0.0100602	0.0645150
27:Psi A to D	0.2073927E-08	0.0000000	0.2073927E-08	0.2073927E-08
28:Psi A to D	0.0103630	0.0074000	0.0025392	0.0412947
29:Psi A to F	0.0106147	0.0077660	0.0025119	0.0437105
30:Psi A to F	0.0066078	0.0033573	0.0024351	0.0178033
31:Psi B to A	0.0698949	0.0349518	0.0255290	0.1773319
32:Psi B to A	0.1400516	0.0299620	0.0909190	0.2096133
33:Psi C to A	0.0262931	0.0264427	0.0035538	0.1697465
34:Psi C to A	0.0255394	0.0157734	0.0075106	0.0832170
35:Psi D to A	0.2107902E-08	0.4092940E-05	-0.8020054E-05	0.8024270E-05
36:Psi D to A	0.2071271E-08	0.4760640E-05	-0.9328784E-05	0.9332926E-05
37:Psi E to F	0.2102606E-08	0.0000000	0.2102606E-08	0.2102606E-08
38:Psi E to F	0.1075214	0.0559766	0.0369821	0.2742856

39:Psi F to A	0.2076728E-08	0.3598819E-05	-0.7051608E-05	0.7055762E-05
40:Psi F to A	0.0112559	0.0043758	0.0052400	0.0240116
41:Psi F to E	0.0946374	0.0939723	0.0120336	0.4728720
42:Psi F to E	0.2067730E-08	0.8621359E-05	-0.1689580E-04	0.1689993E-04

Table 3. MARK output from the best supported model for Champion Creek (FMCS model) showing estimates for all parameters in the model. Parameter estimates are shown for both benthic fishes (first estimate in each group) and water column fishes (second estimate in each group). Parameters that could not be estimated have values listed as a number with a negative exponent or a value of 1. Parameters that could not be estimated were not included in the total number of parameters estimated for AIC calculations.

Parameter	Estimate	Standard Error	Lower	Upper
1:S A:A	0.8138596	0.0704018	0.6374772	0.9157644
2:S A:A	0.9407377	0.0316192	0.8393043	0.9796940
3:p A:A	0.2513598	0.1230255	0.0852732	0.5473624
4:p A:A	0.1853903	0.0299289	0.1336953	0.2512760
5:p B:B	0.2161100	0.0640999	0.1160782	0.3665939
6:p B:B	0.0249711	0.0127751	0.0090747	0.0668357
7:p C:C	0.2436223	0.0801245	0.1207689	0.4302884
8:p C:C	0.4412472	0.0431432	0.3591442	0.5266938
9:p D:D	0.2529477	0.0694683	0.1414258	0.4103769
10:p D:D	0.2552144	0.0435290	0.1794851	0.3492937
11:p E:E	0.2922723	0.0600715	0.1894510	0.4218544
12:p E:E	0.1721549	0.0387464	0.1087766	0.2616202
13:p F:F	0.3622229	0.1118653	0.1802259	0.5946832
14:p F:F	0.4666781	0.0377494	0.3939446	0.5408562
15:p G:G	0.3924734	0.1111495	0.2057732	0.6169778
16:p G:G	0.0573316	0.0271326	0.0222294	0.1399308
17:p H:H	0.2425609	0.0592646	0.1454264	0.3760264
18:p H:H	0.1382283	0.0547818	0.0611427	0.2831856
19:p I:I	0.3456630	0.0901367	0.1947788	0.5356733
20:p I:I	0.1254956	0.0438306	0.0615619	0.2389216
21:p J:J	0.1826089	0.0513351	0.1022141	0.3047712
22:p J:J	0.3336753	0.0475487	0.2477268	0.4323072
23:Psi A to B	0.0832689	0.0247245	0.0459321	0.1463016
24:Psi A to B	0.0918082	0.0178306	0.0623333	0.1332400
25:Psi A to C	0.0560438	0.0255486	0.0225217	0.1326884
26:Psi A to C	0.0071683	0.0051509	0.0017446	0.0289632
27:Psi A to D	0.0097301	0.0151587	0.4497966E-03	0.1766456
28:Psi A to D	0.0079362	0.0042000	0.0028039	0.0222527
29:Psi A to F	0.2061260E-08	0.1981388E-05	-0.3881459E-05	0.3885582E-05
30:Psi A to F	0.0017028	0.0012241	0.4156605E-03	0.0069480
31:Psi B to A	0.0258331	0.0119119	0.0103776	0.0628448
32:Psi B to A	0.1366529	0.0218305	0.0992088	0.1853218
33:Psi C to A	0.0063825	0.0068075	0.7829072E-03	0.0500275
34:Psi C to A	0.0150021	0.0083605	0.0049996	0.0441282
35:Psi D to A	0.2066654E-08	0.3675312E-05	-0.7201544E-05	0.7205677E-05
36:Psi D to A	0.0048079	0.0050068	0.6209574E-03	0.0362034
37:Psi E to F	0.2072020E-08	0.3636036E-05	-0.7124559E-05	0.7128703E-05
38:Psi E to F	0.0505033	0.0272554	0.0171591	0.1394497

39:Psi F to A	0.2062455E-08	0.2226043E-05	-0.4360982E-05	0.4365107E-05
40:Psi F to A	0.2061762E-08	0.1269309E-05	-0.2485784E-05	0.2489907E-05
41:Psi F to E	0.2241116E-08	0.0000000	0.2241116E-08	0.2241116E-08
42:Psi F to E	0.2075703E-08	0.3414661E-05	-0.6690660E-05	0.6694811E-05

Table 4. MARK output from the best supported model for Nimblewill Creek (FM model) showing estimates for all parameters in the model. Parameter estimates are shown for both benthic fishes (first estimate in each group) and water column fishes (second estimate in each group). Parameters that could not be estimated have values listed as a number with a negative exponent or a value of 1. Parameters that could not be estimated were not included in the total number of parameters estimated for AIC calculations.

Parameter	Estimate	Standard Error	Lower	Upper
1:S A:A	0.6440106	0.1460302	0.3417223	0.8630973
2:S A:A	0.8259447	0.0697644	0.6470275	0.9247224
3:S B:B	0.9396056	0.1302787	0.1473954	0.9992863
4:S B:B	0.8358359	0.0686803	0.6562258	0.9314138
5:S C:C	1.0000000	0.0000000	1.0000000	1.0000000
6:S C:C	0.8006571	0.0567072	0.6668617	0.8896123
7:S D:D	0.8384082	0.1639755	0.3261134	0.9823408
8:S D:D	0.7549020	0.0787595	0.5721472	0.8764509
9:S E:E	1.0000000	0.1245700E-04	0.9999756	1.0000244
10:S E:E	0.5514874	0.0476542	0.4573710	0.6420552
11:p A:A	0.3257302	0.1214535	0.1404661	0.5881450
12:p A:A	0.2957648	0.0484604	0.2102262	0.3985446
13:p B:B	0.2276761	0.0792010	0.1086800	0.4161340
14:p B:B	0.1715781	0.0327585	0.1164793	0.2454985
15:p C:C	0.1426636	0.0355388	0.0860504	0.2272614
16:p C:C	0.2632932	0.0374337	0.1966798	0.3428393
17:p D:D	0.2009335	0.0724108	0.0941135	0.3783571
18:p D:D	0.1103615	0.0257865	0.0690209	0.1718917
19:p E:E	0.2172275	0.0390541	0.1503239	0.3032792
20:p E:E	0.2592245	0.0386335	0.1908756	0.3417117
21:p F:F	0.2410190	0.0771110	0.1220338	0.4204585
22:p F:F	0.4389034	0.0713108	0.3072165	0.5797963
23:p G:G	0.1732157	0.0693010	0.0750587	0.3510214
24:p G:G	0.3716685	0.0939491	0.2118900	0.5654815
25:p H:H	0.2819838	0.0605362	0.1793639	0.4137167
26:p H:H	0.3853499	0.0654959	0.2671979	0.5187621
27:p I:I	0.3472340	0.0855863	0.2024094	0.5271890
28:p I:I	0.4705758	0.0844275	0.3138794	0.6332950
29:p J:J	0.1574667	0.0612352	0.0703156	0.3159283
30:p J:J	0.2646498	0.0638715	0.1590631	0.4064495
31:Psi A to B	0.0972037	0.0249141	0.0581324	0.1581266
32:Psi A to B	0.0805147	0.0135002	0.0577149	0.1112578
33:Psi A to C	0.0491149	0.0205448	0.0213432	0.1089982
34:Psi A to C	0.0151519	0.0068767	0.0061963	0.0365752
35:Psi A to D	0.2073541E-08	0.3363637E-05	-0.6590655E-05	0.6594802E-05
36:Psi A to D	0.0088560	0.0053743	0.0026838	0.0288126
37:Psi A to F	0.2069004E-08	0.1617670E-05	-0.3168564E-05	0.3172702E-05
38:Psi A to F	0.2061469E-08	0.7434789E-06	-0.1455157E-05	0.1459280E-05

39:Psi B to A	0.0163759	0.0094548	0.0052414	0.0499749
40:Psi B to A	0.0671707	0.0120784	0.0470297	0.0950768
41:Psi C to A	0.2070408E-08	0.2129918E-05	-0.4172569E-05	0.4176710E-05
42:Psi C to A	0.0193528	0.0067015	0.0097810	0.0379329
43:Psi D to A	0.2070391E-08	0.2228381E-05	-0.4365557E-05	0.4369698E-05
44:Psi D to A	0.0066561	0.0039820	0.0020539	0.0213503
45:Psi E to F	0.2111377E-08	0.3464997E-05	-0.6789284E-05	0.6793506E-05
46:Psi E to F	0.2062170E-08	0.2972927E-05	-0.5824875E-05	0.5829000E-05
47:Psi F to A	0.2064552E-08	0.1845672E-05	-0.3615452E-05	0.3619581E-05
48:Psi F to A	0.0075313	0.0033517	0.0031412	0.0179467
49:Psi F to E	0.2185146E-08	0.1362516E-04	-0.2670313E-04	0.2670750E-04
50:Psi F to E	0.2075425E-08	0.7112927E-05	-0.1393926E-04	0.1394341E-04

Table 5. MARK output from the best supported model for the sites combined (FM model) showing estimates for all parameters in the model. Parameter estimates are shown for each site in the following order: 1.Canton 2.Bluff 3.Champion 4.Nimblewill. Parameters that could not be estimated have values listed as a number with a negative exponent or a value of 1. Parameters that could not be estimated were not included in the total number of parameters estimated for AIC calculations.

Parameter	Estimate	Standard Error	Lower	Upper
1:S A:A	0.7920816	0.1035061	0.5263916	0.9288643
2:S A:A	0.7485759	0.1223751	0.4542855	0.9141532
3:S A:A	0.8830482	0.0633270	0.6941849	0.9617083
4:S A:A	0.7870185	0.0650549	0.6332892	0.8877278
5:S B:B	0.8176657	0.1195866	0.4821228	0.9557552
6:S B:B	0.8605083	0.0938032	0.5714647	0.9661443
7:S B:B	0.9708332	0.0987241	0.0346088	0.9999676
8:S B:B	0.8602800	0.0601991	0.6976195	0.9426351
9:S C:C	0.7891770	0.1069645	0.5149648	0.9295676
10:S C:C	0.6313955	0.0900950	0.4450888	0.7853213
11:S C:C	0.8715832	0.0497177	0.7396971	0.9418963
12:S C:C	0.8564886	0.0547122	0.7138167	0.9345549
13:S D:D	0.5541094	0.0888486	0.3804734	0.7154731
14:S D:D	0.8847073	0.1090350	0.4856557	0.9842177
15:S D:D	1.0000000	0.2750398E-05	0.9999946	1.0000054
16:S D:D	0.7700010	0.0722931	0.6006640	0.8816764
17:S E:E	0.7039432	0.0932549	0.4972787	0.8510902
18:S E:E	0.8459522	0.0759090	0.6368018	0.9450539
19:S E:E	0.8553192	0.0413976	0.7542232	0.9192815
20:S E:E	0.6019395	0.0469141	0.5074481	0.6893990
21:p A:A	0.0659155	0.0403045	0.0191857	0.2029161
22:p A:A	0.1717938	0.0501691	0.0941422	0.2927928
23:p A:A	0.2046709	0.0372966	0.1410700	0.2873531
24:p A:A	0.3018435	0.0454620	0.2207384	0.3975461
25:p B:B	0.2224273	0.0658348	0.1194497	0.3762485
26:p B:B	0.2478037	0.0500020	0.1629909	0.3578795
27:p B:B	0.0825958	0.0236801	0.0465256	0.1424523
28:p B:B	0.1798716	0.0301350	0.1281286	0.2466001
29:p C:C	0.0369035	0.0268914	0.0086217	0.1444419
30:p C:C	0.3106780	0.0762286	0.1832312	0.4751965
31:p C:C	0.4114032	0.0447907	0.3272368	0.5010935
32:p C:C	0.2279117	0.0301211	0.1742729	0.2922173
33:p D:D	0.3623561	0.0910949	0.2078559	0.5517126
34:p D:D	0.1745809	0.0505663	0.0961045	0.2961439
35:p D:D	0.2275154	0.0290594	0.1755972	0.2893954
36:p D:D	0.1255187	0.0251576	0.0839074	0.1836293
37:p E:E	0.3161431	0.0771510	0.1867921	0.4819787
38:p E:E	0.3311233	0.0723783	0.2069149	0.4843564

39:p E:E	0.2377562	0.0360541	0.1743726	0.3153785
40:p E:E	0.2455310	0.0334384	0.1859737	0.3167400
41:p F:F	0.2344569	0.0954345	0.0974873	0.4647668
42:p F:F	0.3008228	0.0562154	0.2030660	0.4207914
43:p F:F	0.4959271	0.0408351	0.4166696	0.5753898
44:p F:F	0.3929886	0.0591786	0.2847542	0.5128636
45:p G:G	0.4295831	0.1190681	0.2251442	0.6612414
46:p G:G	0.1668612	0.0470483	0.0935225	0.2799497
47:p G:G	0.1593437	0.0376141	0.0985459	0.2473582
48:p G:G	0.2755656	0.0581721	0.1768682	0.4024134
49:p H:H	0.2512591	0.0656216	0.1448494	0.3993358
50:p H:H	0.3677284	0.0804349	0.2279299	0.5339697
51:p H:H	0.1935903	0.0398129	0.1271164	0.2835345
52:p H:H	0.3406054	0.0489185	0.2520886	0.4418416
53:p I:I	0.2047580	0.0586481	0.1127693	0.3427924
54:p I:I	0.2027103	0.0554499	0.1148736	0.3324816
55:p I:I	0.2111906	0.0495883	0.1299735	0.3242444
56:p I:I	0.4293197	0.0600775	0.3175100	0.5488398
57:p J:J	0.2222595	0.0587062	0.1280629	0.3573467
58:p J:J	0.0583080	0.0283858	0.0219843	0.1457066
59:p J:J	0.2761998	0.0433520	0.1996594	0.3685698
60:p J:J	0.2067640	0.0416312	0.1368093	0.3000546
61:Psi A to B	0.1534680	0.0317392	0.1009697	0.2263889
62:Psi A to B	0.1079712	0.0209202	0.0732799	0.1563155
63:Psi A to B	0.0712586	0.0118168	0.0512962	0.0981856
64:Psi A to B	0.0865959	0.0120973	0.0656391	0.1134315
65:Psi A to C	0.0124489	0.0116228	0.0019723	0.0744265
66:Psi A to C	0.0243234	0.0108757	0.0100517	0.0576780
67:Psi A to C	0.0155097	0.0060689	0.0071769	0.0331938
68:Psi A to C	0.0250145	0.0078668	0.0134507	0.0460556
69:Psi A to D	0.2062671E-08	0.0000000	0.2062671E-08	0.2062671E-08
70:Psi A to D	0.0085171	0.0060993	0.0020811	0.0341758
71:Psi A to D	0.0079697	0.0038895	0.0030537	0.0206358
72:Psi A to D	0.0064187	0.0043747	0.0016812	0.0241831
73:Psi A to F	0.2061636E-08	0.1328269E-05	-0.2601346E-05	0.2605470E-05
74:Psi A to F	0.0071677	0.0030612	0.0030973	0.0164986
75:Psi A to F	0.0011533	0.9293069E-03	0.2374959E-03	0.0055808
76:Psi A to F	0.2061178E-08	0.7120615E-06	-0.1393579E-05	0.1397702E-05
77:Psi B to A	0.1113469	0.0258079	0.0698975	0.1728090
78:Psi B to A	0.1168002	0.0233814	0.0781802	0.1709598
79:Psi B to A	0.0788324	0.0129162	0.0569507	0.1081573
80:Psi B to A	0.0529237	0.0090922	0.0376854	0.0738508
81:Psi C to A	0.0153868	0.0118588	0.0033584	0.0675760
82:Psi C to A	0.0263878	0.0136305	0.0094900	0.0712106
83:Psi C to A	0.0111881	0.0055620	0.0042059	0.0294192
84:Psi C to A	0.0149958	0.0051169	0.0076615	0.0291451

85:Psi D to A	0.0045114	0.0046044	0.6071745E-03	0.0326991
86:Psi D to A	0.2062156E-08	0.1994882E-05	-0.3907907E-05	0.3912032E-05
87:Psi D to A	0.0032687	0.0029820	0.5450632E-03	0.0193389
88:Psi D to A	0.0050745	0.0029685	0.0016085	0.0158901
89:Psi E to F	0.0271133	0.0285726	0.0033239	0.1888974
90:Psi E to F	0.0908736	0.0479186	0.0310730	0.2375466
91:Psi E to F	0.0309789	0.0171955	0.0102955	0.0894590
92:Psi E to F	0.2063816E-08	0.2199193E-05	-0.4308355E-05	0.4312482E-05
93:Psi F to A	0.2061581E-08	0.1522858E-05	-0.2982741E-05	0.2986864E-05
94:Psi F to A	0.0092623	0.0035812	0.0043319	0.0196934
95:Psi F to A	0.2061241E-08	0.1147207E-05	-0.2246464E-05	0.2250586E-05
96:Psi F to A	0.0043118	0.0019272	0.0017933	0.0103306
97:Psi F to E	0.2127577E-08	0.1008767E-04	-0.1976971E-04	0.1977397E-04
98:Psi F to E	0.0113950	0.0136504	0.0010710	0.1102566
99:Psi F to E	0.2063269E-08	0.1834916E-05	-0.3594373E-05	0.3598500E-05
100:Psi F to E	0.2074021E-08	0.3446712E-05	-0.6753483E-05	0.6757631E-05

Table 6. Species list for Canton Creek showing total caught (n) and 10th, 25th, 75th and 90th percentiles for standard length (mm).

Genus species	**n**	**SL 25th percentile**	**SL 75th percentile**	**SL 10th percentile**	**SL 90th percentile**
Benthic fishes					
Amieurus melas	1	98	98	98	98
Amieurus sp.	1	93	93	93	93
Campostoma oligolepis	76	47	84	42.5	93
Etheostoma scotti	51	45	49.5	43	56
Hypentelium etowanum	13	111	121	94	125.4
Percina kathae	40	65.75	75	56.6	76.1
Percina nigrofasciata	31	54	62	52	70
Water column fishes					
Gambusia holbrooki	3	38	40	37.4	40.6
Lepomis auritus	296	52	74	36	87
Lepomis gulosus	4	83	90.75	77.6	93.9
Lepomis macrochirus	366	38.25	64	33	73
Lepomis microlophus	4	60.75	67.5	60.3	70.2
Micropterus coosae	2	79.75	85.25	78.1	86.9
Micropterus salmoides	5	47	68	46.4	79.4
Notropis xaenocephalus	8	64.5	67.5	61.2	70.2
Semotilus atromachlatus	2	62.25	92.75	53.1	101.9

Table 7. Species list for Bluff Creek showing total caught (n) and 10th, 25th, 75th and 90th percentiles for standard length (mm).

Genus species	**n**	**SL 25th percentile**	**SL 75th percentile**	**SL 10th percentile**	**SL 90th percentile**
Benthic fishes					
Campostoma oligolepis	81	37	67	35	76
Cottus sp. cf. carolinae	53	55	63	53	71.4
Etheostoma scotti	23	35.5	50.5	32.2	54
Etheostoma stigmaeum	1	58	58	58	58
Hypentelium etowanum	315	41	67	37	98
Noturus leptacanthus	24	54.5	69.25	42	72
Percina kathae	7	92.5	106.5	90.8	110
Percina nigrofasciata	89	53	70	43	78
Water column fishes					
Cyprinella callistia	131	34	68	32	82
Fundulus stellifer	2	63	67	61.8	68.2
Lepomis auritus	9	33	68	32.8	104.2
Lepomis cyanellus	1	85	85	85	85
Lepomis macrochirus	46	67.25	88.75	53.5	103
Micropterus coosae	12	38	47	37.1	81.2
Micropterus punctulatus	1	36	36	36	36
Micropterus salmoides	3	55	72	53.8	81
Moxostoma sp.	25	52	57	51	61.6
Nocomis leptocephalus	39	35.5	52.5	32.8	87.8
Notropis lutipinnis	172	34	50	32	57
Notropis xaenocephalus	83	35	58	34	63
Semotilus atromaculatus	99	37	42.5	34	44

Table 8. Species list for Champion Creek showing total caught (n) and 10th, 25th, 75th and 90th percentiles for standard length (mm).

Genus species	**n**	**SL 25th percentile**	**SL 75th percentile**	**SL 10th percentile**	**SL 90th percentile**
Benthic fishes					
Campostoma oligolepis	213	47	59	42	84.8
Cottus sp. cf. carolinae	224	50	60.25	47	67.7
Etheostoma scotti	92	35	48	33	52
Hypentelium etowanum	163	41	59.5	37	92.8
Noturus leptacanthus	41	57	64	54	67
Percina sp. cf. macrocephala	1	48	48	48	48
Percina nigrofasciata	27	49	56.5	47	62.4
Percina palmaris	36	49.5	62.25	40.5	67
Water column fishes					
Cyprinella callistia	257	47	57	43	72
Lepomis auritus	3	98	116	91.4	120.2
Lepomis cyanellus	4	109	117.5	109	123.8
Lepomis macrochirus	7	73	94.5	71.4	104.4
Lepomis punctatus	1	106	106	106	106
Micropterus coosae	19	38.8	58.5	38.8	91
Nocomis leptocephalus	1	46	46	46	46
Notropis xaenocephalus	494	38	49	38	53
Semotilus atromaculatus	13	33.4	52	33.4	82.4

Table 9. Species list for Nimblewill Creek showing total caught (n) and 10th, 25th, 75th and 90th percentiles for standard length (mm).

Genus species	**n**	**SL 25th percentile**	**SL 75th percentile**	**SL 10th percentile**	**SL 90th percentile**
Benthic fishes					
Campostoma oligolepis	16	67.75	93.25	44.5	100.5
Cottus sp. cf. carolinae	469	45	57	33	63
Etheostoma sp. cf. brevirostrum	1	52	52	52	52
Hypentelium etowanum	130	48	96.75	41.9	118.2
Percina palmaris	11	58	66.5	58	68
Water column fishes					
Nocomis leptocephalus	177	50	76	39	97.4
Notropis lutipinnis	1268	43	52	40.7	57
Semotilus atromaculatus	18	42.25	73	40.7	96