Responses of Benthic Macroinvertebrates to Urbanization in Nine Metropolitan Areas of the Conterminous United States

T.F. Cuffney, G. McMahon, R. Kashuba, J.T. May, I.R. Waite

Abstract

The effects of urbanization on benthic macroinvertebrates were investigated in nine metropolitan areas (Boston, MA; Raleigh, NC; Atlanta, GA; Birmingham, AL; Milwaukee–Green Bay, WI; Denver, CO; Dallas–Fort Worth, TX; Salt Lake City, UT; and Portland, OR) as a part of the U.S. Geological Survey National Water Quality Assessment Program. Several invertebrate metrics showed strong, linear responses to urbanization when forest or shrublands were developed. Responses were difficult to discern in areas where urbanization was occurring on agricultural lands because invertebrate assemblages were already severely degraded. There was no evidence that assemblages showed any initial resistance to urbanization. Ordination scores, EPT taxa richness, and the average tolerance of organisms were the best indicators of changes in assemblage condition at a site. Richness metrics were better indicators than abundance metrics, and qualitative samples were as good as quantitative samples. A common set of landscape variables (population density, housing density, developed landcover, impervious surface, and roads) were strongly correlated with urbanization and invertebrate responses in all non-agricultural areas. The instream environmental variables (hydrology, water chemistry, habitat, and temperature) that were strongly correlated with urbanization and invertebrate responses were influenced by environmental setting (e.g., dominant ecoregion) and varied widely among metropolitan areas. Multilevel hierarchical regression models were developed that predicted invertebrate responses using only two landcover variables—basin-scale landcover (percentage of basin area in developed land) and regional-scale landcover (antecedent agricultural land).

Cuffney, McMahon, May, and Waite are research scientists, all with the U.S. Geological Survey, Water Resources Discipline. Kashuba is with Duke University, Durham, NC. T.F. Cuffney: 3916 Sunset Ridge Road, Raleigh, NC 27607. Email: tcuffney@usgs.gov.

Keywords: benthic macroinvertebrates, disturbance, landcover, water chemistry, urbanization, water quality

Introduction

Stream ecosystems are increasingly affected by urban development associated with human population growth (Booth and Jackson 1997, Paul and Meyer 2001, Walsh et al. 2001). Changes in landcover, hydrology, and impervious surfaces associated with urbanization alter the physical and chemical environment of streams and degrade invertebrate assemblages (Kennen 1999, Yoder et al. 1999, Huryn et al. 2002, Kennen and Ayers 2002, Morley and Karr 2002, Ourso and Frenzel 2003, Morse et al. 2003, Roy et al. 2003, Brown et al. 2005).

In 1999, the U.S. Geological Survey (USGS) initiated a series of urban stream studies as part of the National Water-Quality Assessment (NAWQA) Program. These studies are based on a common study design (McMahon and Cuffney 2000, Coles et al. 2004, Cuffney et al. 2005, Tate et al. 2005), consistent measures of urban intensity (Cuffney and Falcone 2008), and standard sample-collection and processing methods (Fitzpatrik et al. 1998, Moulton et al. 2002). Nine major metropolitan areas—Boston, MA (BOS); Raleigh, NC (RAL); Atlanta, GA (ATL); Birmingham, AL (BIR); Milwaukee–Green Bay, WI (MGB); Denver, CO (DEN); Dallas–Fort Worth, TX (DFW); Salt Lake City, UT (SLC); and Portland, OR (POR) (Figure 1)—were chosen to represent the effects of urbanization in regions of the country that differ in potential natural vegetation, temperature, precipitation, basin relief, elevation, and basin slope. The objectives of these studies are to (1) determine the response of benthic macroinvertebrates to urbanization, (2) identify the landscape (census, landcover, and infrastructure) and instream (water chemistry, hydrology, habitat, and temperature) environmental variables that are strongly associated with urbanization and invertebrate responses, and (3) compare the similarities and differences in the process of urbanization and its effects among these nine metropolitan areas.

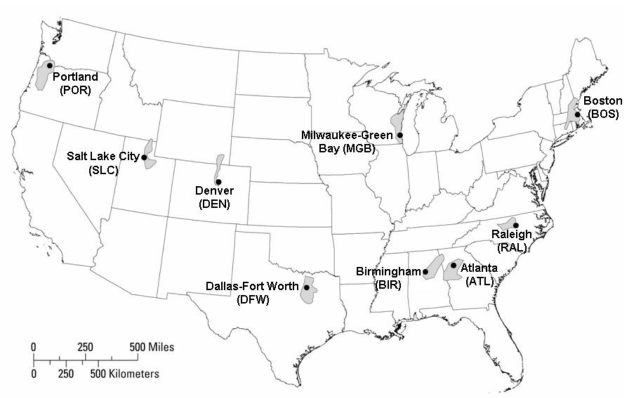

Figure 1. Locations of the nine metropolitan areas in which urban studies were conducted (shaded areas).

Methods

A population of candidate basins (typically basins drained by 2nd- to 3rd-order streams) was delineated within each of the nine metropolitan areas based on 30-m digital elevation models (U.S. Geological Survey 2003). Landcover, census, and infrastructure variables were summarized using nationally available databases in a geographic information system (GIS) (Falcone et al. 2007).

Urban intensity was defined by combining housing density, percentage of basin area in developed landcover, and road density into an index (metropolitan area national urban-intensity index, MA-NUII) scaled to range from 0 (little or no urban) to 100 (maximum urban) within each metropolitan area (Cuffney and Falcone 2008). Once groups of basins with relatively homogeneous environmental features were defined, 28–30 basins were selected to represent the gradient of urbanization in each metropolitan area. This spatially distributed sampling network represents changes in urbanization through time (i.e., substitute space for time).

The BOS, BIR, and SLC metropolitan areas were studied during 1999–2000; ATL, DEN, and RAL were studied during 2002–2003 and DFW, MGB, and POR in 2003–2004. Details of the study designs can be found in Coles et al. (2004), Tate et al. (2005), and Cuffney et al. (2005).

The NAWQA Program sampling protocols were used to collect benthic macroinvertebrates over a 1- to 4-week period during summer low base flows (Cuffney et al. 1993, Moulton et al. 2002). Quantitative (RTH) and qualitative multihabitat (QMH) samples were collected within each sampling reach. Samples were preserved in 10-percent buffered formalin and sent to the USGS National Water Quality Laboratory in Denver, CO, for taxa identification and enumeration (Moulton et al. 2000). The USGS Invertebrate Data Analysis System (IDAS; Cuffney 2003) was used to resolve taxonomic ambiguities and calculate assemblage metrics and diversity measures (Cuffney et al. 2007).

Water temperature, stream stage, water chemistry (nutrients, major ions, pesticides), dissolved oxygen, pH, and specific conductance were collected for about 1 year prior to the collection of biological samples (Cuffney and Brightbill 2008, Giddings et al. 2009). Water-column chemistry data were collected twice—once during high base flow (typically spring) and once during low base flow (typically summer) periods—using NAWQA Program sampling protocols (Shelton 1994; U.S. Geological Survey, variously dated). Samples were sent to the USGS National Water Quality Laboratory in Denver, CO, for analysis (Fishman and Friedman 1989, Brenton and Arnett 1993, Fishman 1993, Zaugg et al. 1995, U.S. Environmental Protection Agency 1997). Pesticide concentrations were weighted by toxicity to form an aggregate pesticide toxicity index (PTI; Munn and Gilliom 2001). Details on water-chemistry sampling can be found in Sprague et al. (2007) and Giddings et al. (2009). Water column chemistry measurements were supplemented with data from semipermeable membrane devices (SPMDs) that were used to collect hydrophobic organic compounds from water during a 4- to 6-week period in early to midsummer in RAL, ATL, MGB, DEN, DFW, and POR (Bryant et al. 2007).

Physical habitat structure was characterized by using NAWQA Program protocols (Fitzpatrick et al. 1998), generally after invertebrate sampling was completed.

Details on the processing and derivation of habitat metrics can be found in Giddings et al. (2009).

Invertebrate responses to urbanization were evaluated by relating assemblage structure and assemblage metrics to urban intensity. Nonmetric multidimensional scaling (NMDS) was used to derive the ordination axis sample scores after applying a 4th root transformation to the density data and using Bray-Curtis similarity for RTH samples and Jaccard similarity for QMH samples (Clarke and Gorley 2006). NMDS plots were examined for outliers, which were removed prior to analysis. Ordination sample scores were rescaled (Cuffney et al. 2005) to convert all ordination scores to positive values that were consistent with the response of the EPT richness metric to urbanization (i.e., decrease in value as urbanization increases) (Paul and Meyer 2001, Morse et al. 2003). Linear regression was used to determine the relation (slope) between assemblages (ordination sample scores) and urban intensity.

Assemblage metrics were correlated (Spearman rank correlation; SPSS 2007) with urban intensity to determine how strongly metrics were associated with urban intensity. Correlation analyses emphasized similarities among metropolitan areas by focusing on correlations that were statistically significant and indicative of strong correlation ($|\rho| \geq 0.65$) in at least three of the nine metropolitan areas.

Infrastructure, hydrology, water chemistry, habitat, and water temperature were correlated (Spearman rank correlation; SPSS 2007) with urban intensity (MA-NUII) and invertebrate responses (rescaled NMDS axis-1 sample scores) to identify environmental factors that may be important in characterizing and managing urbanization locally and at the national scale. Strong correlations were identified using the same criteria that identified strong correlations between NMDS sample scores and urban intensity ($|\rho| \geq 0.65$ in three or more metropolitan areas).

Results

The rates at which indicators of urbanization (landcover, census, and infrastructure) change in response to changes in population density differed among metropolitan areas (e.g. developed land, Figure 2). Consequently, the MA-NUII index was modified to form the national urban intensity index (NUII) that accounted for these regional differences in response rates (Cuffney and Falcone 2008). The MA-NUII index is scaled to range from 0 to 100 in each metropolitan area (Figure 3A), whereas the NUII is scaled to range from 0 to 100 over all nine metropolitan areas (Figure 3B). Rescaling to account for regional differences among metropolitan areas showed that the maximum level of urbanization that occurred in the eastern United States was substantially less than in the central and western United States (Figure 3). This difference is thought to be associated with stream burial in the inner cities of the east (Elmore and Kaushal 2008) that effectively eliminated these streams from the studies.

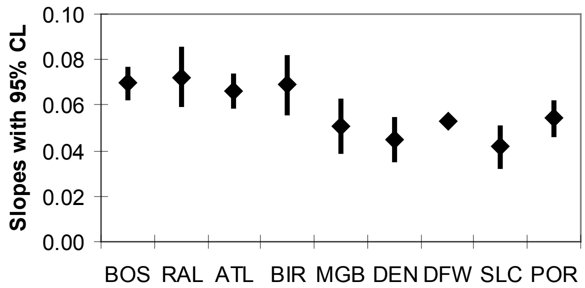

Figure 2. The rates at which the percentage of basin area in developed land change as population density changes differ among the nine metropolitan areas. [CL, confidence limit]

The biological responses to urbanization showed statistically significant responses between ordination site scores (NMDS axis 1) and MA-NUII in 6 of the 9 metropolitan areas for RTH and 8 of 9 for QMH samples. The metropolitan areas that did not show responses to urbanization for invertebrates (MGB, DEN, DFW) were those where urbanization progressed by the conversion of agricultural lands (row crop, pasture, and grazing lands). In these areas the effects of urbanization were masked by the effects of agriculture on invertebrate assemblages.

Invertebrate responses to urbanization were generally linear with no evidence for an initial threshold. That is, degradation of the invertebrate assemblages began as soon as the background landcover was disturbed (Figure 4). Proposed criteria for biological protection based on 5 and 10 percent impervious surface were not protective. For example, 10 percent impervious surface in Boston corresponded to an MA-NUII score of 33 and represented one-third of the change that occurred over the entire urban gradient (Figure 4).

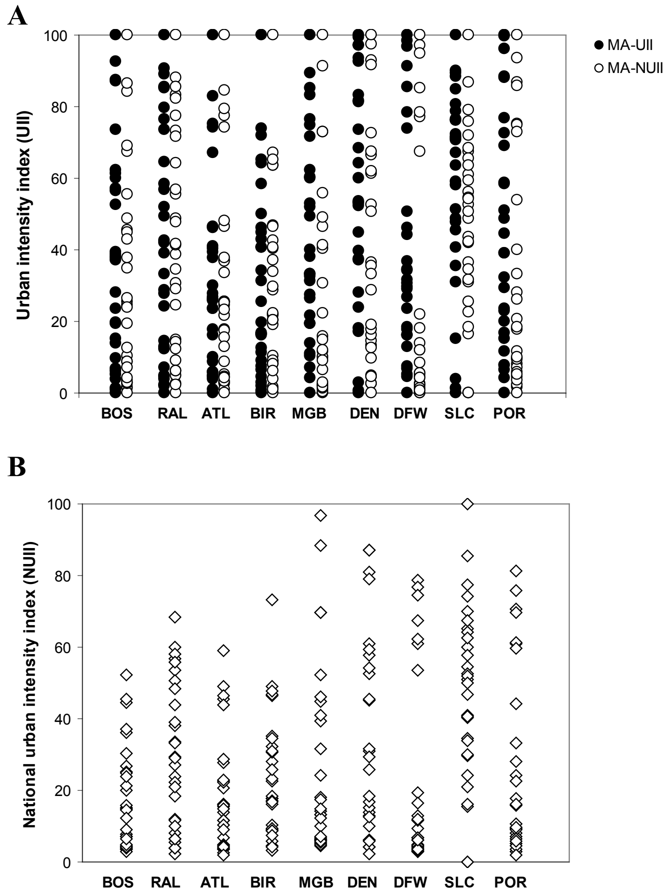

Figure 3. Distribution of sites in each metropolitan area based on the (A) MA-UII and MA-NUII indices or the (B) NUII index.

The environmental variables that were most strongly correlated with invertebrate responses (NMDS axis 1 site scores) showed considerable variation among metropolitan areas (Table 1). Census, landcover, and infrastructure variables were associated with changes in invertebrate assemblages in 5 of the 9 metropolitan areas. Chemistry, hydrology, water temperature, and habitat were less frequently associated with changes in invertebrates. In part, this reflects the difficulty in measuring these highly variable parameters in an ecologically meaningful way.

The relations between invertebrate responses and environmental variables were more fully developed using multilevel hierarchical linear regression models (Gelman and Hill 2007). This type of regression model can incorporate predictor variables at multiple scales. We used the percentage of basin area in developed land as the site level predictor and antecedent agricultural landcover (mean row crop, pasture, and grazing land at sites with MA-NUII ≤ 10) and mean annual air temperature for each metropolitan area as regional predictors. The multilevel hierarchical linear regression model predicts the intercept (a) and slope (b) of the regression relating invertebrate responses to percentage of developed land ($y_{ij} = a_j + b_j X_{ij}$) on the basis of regional variables ($a_j = \alpha_{0j} + \beta_{0j} R_j$ and $b_j = \alpha_{1j} + \beta_{1j} R_j$). The mean tolerance value for invertebrates at a site (Cuffney 2003) was used as the invertebrate response variable, y, that varied by site (i) and region (j).

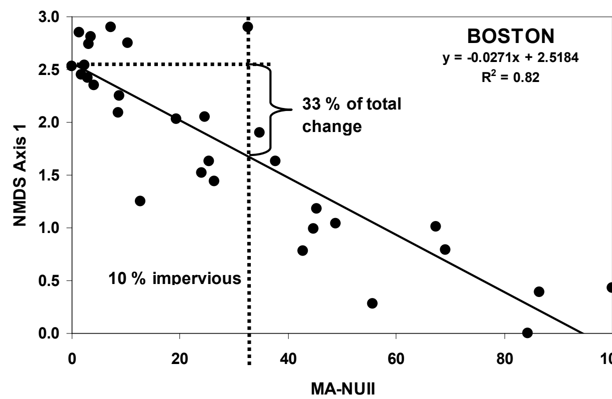

Figure 4. A criterion of 10 % impervious surface (equivalent to a MA-NUII of 32.6) corresponds to a 33 % change in the invertebrate assemblage and is not protective.

Table 1. Environmental variables that are strongly correlated with changes in invertebrate assemblages. [Number of metropolitan areas with strong correlations are listed in parentheses. Pop., population; TEQ, toxicity equivalents]

Environmental variable	**Environmental variable**
Census	Chemistry
Pop. density (5)	Conductivity (3)
Housing density (5)	Sulfate (3)
% pop. in urban (5)	Pesticides detected (4)
Landcover	Toxicity: TEQ (4)
% developed (5)	Hydrology
% impervious (5)	Flashiness: rise/fall (2)
Infrastructure	Water temperature
Roads (5)	Mean summer (3)
	Habitat (1)

Multilevel regression showed that antecedent agriculture had a strong effect on the value of the slope and intercept that relates invertebrate tolerance to percentage developed land (Figure 5). The higher average tolerance value of the intercept in metropolitan areas with high antecedent agriculture shows that high agriculture results in tolerant assemblages even in the absence of urbanization. The higher slopes associated with metropolitan areas with low antecedent agriculture indicates that these areas undergo more degradation as a result of urbanization than those areas that are already heavily affected by agriculture. These results indicate that the efforts to mitigate effects of urbanization in areas with high levels of antecedent agriculture must take into account the negative effects of the agriculture when determining the levels of recovery that can be achieved. These results also establish why statistically significant responses were not observed for MGB, DEN, and DFW, which have high levels of antecedent agriculture.

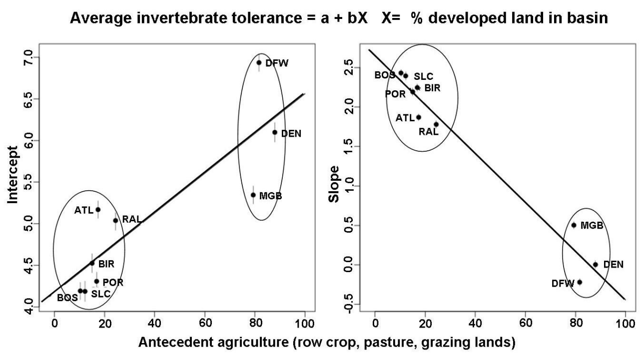

Figure 5. Multilevel hierarchical regression relating average invertebrate tolerance to percentage developed land using antecedent agriculture as a regional variable.

A cursory examination of Figure 5 shows that the 9 metropolitan areas group into high (≥70) and low (<30) classes based on antecedent agriculture. Consequently, the effects of antecedent agriculture can be expressed as a categorical variable (high and low), and regional affects can be examined within each of these categories (Figure 6). This combined analysis shows both the effects of antecedent agriculture and average annual temperature on the response of invertebrates to changes in landcover.

The intercepts in Figure 6 indicate that in the absence of urbanization the average tolerance of invertebrates are higher (conditions are worse) in regions with high levels of antecedent agriculture than in regions with low levels. In addition, this figure shows that average tolerance is affected by temperature regardless of the degree of antecedent agriculture. This has significance for climate change models because it shows that rising temperatures will result in degradation of invertebrate assemblages in both agricultural and non-agricultural regions as tolerant taxa are lost due to rising temperatures.

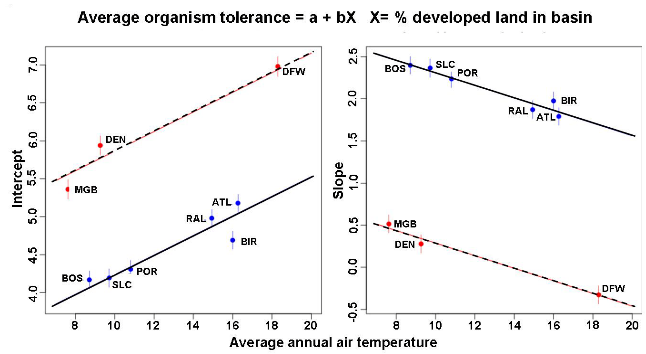

Figure 6. Multilevel hierarchical regression relating average invertebrate tolerance to percentage developed land using antecedent agriculture as a categorical variable regional variable and average annual air temperature as a continuous regional variable.

The slopes shown in Figure 6 indicate that the rates at which average invertebrate tolerance changes are affected by temperature in both agricultural and non-agricultural regions, though the rates of change are higher in the non-agricultural regions. Consequently, climate changes can be expected to affect non-agricultural areas to a greater extent than agricultural areas that are already degraded.

The distribution of regional antecedent agriculture shown in Figure 5 has implications for an understanding of national patterns in urban development. That is, is the gap in regional antecedent agriculture between 30 and 70 percent of basin area real or is it an artifact of the 9 metropolitan areas chosen for study? The NAWQA Program has data from 3 other metropolitan areas (Anchorage, AK; Chicago, IL; and Seattle, WA) that will be used to assess model performance; however, each of these areas falls into the existing categories of high (Chicago) and low (Anchorage and Seattle) agriculture. We are currently working toward compiling statistics on antecedent agriculture for the Nation to determine the representativeness of the 9 metropolitan areas.

Conclusions

The rates at which census, landcover, and infrastructure characteristics change relative to population density are not constant across the country, and the characterization of urbanization should take this regional variability into account. Invertebrate responses are generally linear and did not display any threshold responses. The environmental variables associated with invertebrate responses varied regionally. Multilevel hierarchical regression models revealed that antecedent agriculture had a strong influence on invertebrate responses because of the degradation associated with agriculture. Multilevel modeling also showed a strong influence of temperature on the initial condition of invertebrate assemblages and the rates at which they respond to urbanization. This result has important implications for assessing the effects of climate change on invertebrate assemblages and water quality.

Acknowledgments

We gratefully acknowledge the many colleagues who have worked tirelessly to collect samples, analyze data, and write the reports that support this summary paper. We also thank the many property owners and municipalities for granting us permission to access and sample these streams. Without their generous support, none of this work would have been possible. We also thank Larry Brown and Jim Coles of the USGS for providing critical comments on previous versions of this paper.

References

Booth, D.B., and C.J. Jackson. 1997. Urbanizaiton of aquatic systems: Degradation thresholds, stormwater detention and the limits of mitigations. Water Resources Bulletin 33:1,077–1,090.

Brenton, R.W., and T.L. Arnett. 1993. Methods of analysis by the U.S. Geological Survey National Water Quality Laboratory: Determination of dissolved organic carbon by UV-promoted persulfate oxidation and infrared spectrometry. U.S. Geological Survey Open-File Report 92-480.

Brown, L.R., R.H. Gray, R.M. Hughes, and M.R. Meador. 2005. Introduction to Effects of Urbanization on Stream Ecosystems. In L.R. Brown, R.M. Hughes, R. Gray, and M.R. Meador, eds., Effects of Urbanization on Stream Ecosystems, pp. 1–8. American Fisheries Society, Symposium 47, Bethesda, MD.

Bryant, W.B., S.L. Goodbred, T.L. Leiker, L. Inouye, and B.T. Thomas. 2007. Use of chemical analysis and assays of semipermeable membrane devices extracts to assess the response of bioavailable organic pollutants in streams to urbanization in six metropolitan areas of the United States. U.S. Geological Survey, Scientific Investigations Report 2007-5113.

Clarke, K.R., and R.N. Gorley. 2006. PRIMER (Plymouth Routines in Multivariate Ecological Research), ver. 6: User manual/tutorial. PRIMER-E, Ltd., Plymouth, UK.

Coles, J.F., T.F. Cuffney, and G. McMahon. 2004. The effects of urbanization on the biological, physical, and chemical characteristics of coastal New England streams. U.S. Geological Survey, Professional Paper 1695.

Cuffney, T.F. 2003. User's manual for the National Water-Quality Assessment Program Invertebrate Data Analysis System (IDAS) software, ver. 3. U.S. Geological Survey, Open-File Report 03-172.

Cuffney, T.F., M.D. Bilger, and A.M. Haigler. 2007. Ambiguous taxa: Effects on the characterization and interpretation of invertebrate assemblages. Journal of the North American Benthological Society 26:286–307.

Cuffney, T.F., and R.A. Brightbill. 2008. Methods for processing and summarizing time-series temperature data collected as a part of the National Water-Quality Assessment Program studies on the effects of urbanization on stream ecosystems. U.S. Geological Survey, Data Series 330.

Cuffney, T.F., and J.F. Falcone. 2008. Derivation of nationally consistent indices representing urban intensity within and across nine metropolitan areas of the conterminous United States. U.S. Geological Survey, Scientific Investigations Report 2008–5095.

Cuffney, T.F., M.E. Gurtz, and M.R. Meador. 1993. Methods for collecting benthic invertebrate samples as part of the National Water-Quality Assessment Program. U.S. Geological Survey, Open-File Report 93-406.

Cuffney, T.F., H. Zappia, E.M.P. Giddings, and J.F. Coles. 2005. Effects of urbanization on benthic macroinvertebrate assemblages in contrasting environmental settings: Boston, Massachusetts; Birmingham, Alabama; and Salt Lake City, Utah. In

L.R. Brown, R.M. Hughes, R. Gray, and M.R. Meador, eds., Effects of Urbanization on Stream Ecosystems, pp. 361–407. American Fisheries Society, Symposium 47, Bethesda, MD.

Elmore, A.J., and S.S. Kaushal. 2008. Disappearing headwaters: Patterns of stream burial due to urbanization. Frontiers in Ecology and the Environment 6(6):308–312.

Falcone, J.F., J. Stewart, S. Sobieszcyk, J. Dupree, G. McMahon, and G. Buell. 2007. A comparison of natural and urban characteristics and the development of urban intensity indices across six geographic settings. U.S. Geological Survey, Scientific Investigations Report 2007-5123.

Fishman, M.J. 1993. Methods of analysis by the U.S. Geological Survey National Water-Quality Laboratory: Determination of inorganic and organic constituents in water and fluvial sediment. U.S. Geological Survey, Open File Report 93-125.

Fishman, M.J., and L.C. Friedman. 1989. Methods for determination of inorganic substances in water and fluvial sediments. U.S. Geological Survey, Techniques of Water-Resources Investigations, Book 5, Chapter A1.

Fitzpatrick, F.A., I.R. Waite, P.J. D'Arconte, M.R. Meador, M.A. Maupin, and M.E. Gurtz. 1998. Revised methods for characterizing stream habitat in the National Water-Quality Assessment Program. U.S. Geological Survey, Water-Resources Investigations Report 98-4052.

Gelman, A., and J. Hill. 2007. Data Analysis Using Regression and Multilevel/Hierarchical Models. Cambridge University Press, New York.

Giddings, E.M.P., A.H. Bell, K.M. Beaulieu, T.F. Cuffney, J.F. Coles, L.R. Brown, F.A. Fitzpatrick, J. Falcone, L.A. Sprague, W.L. Bryant, M.C. Peepler, C. Stephens, and G. McMahon. 2009. Selected physical, chemical, and biological data used to study urbanizing streams in nine metropolitan areas of the United States, 1999–2004. U.S. Geological Survey Data Series 423.

Huryn, A.D., V.M.B. Huryn, C.J. Arbuckle, and L. Tsomides. 2002. Catchment land-use, macroinvertebrates and detritus processing in headwater streams: Taxonomic richness versus function. Freshwater Biology 47:401–415.

Kennen, J.G. 1999. Relation of macroinvertebrate community impairment to catchment characteristics in New Jersey streams. Journal of the American Water Resources Association 36:939–955.

Kennen, J.G., and M.A. Ayers. 2002. Relation of environmental characteristics to the composition of aquatic assemblages along a gradient of urban land use in New Jersey, 1996–98. U.S. Geological Survey, Water-Resources Investigations Report 02-4069.

McMahon, G., and T.F. Cuffney. 2000. Quantifying urban intensity in drainage basins for assessing stream ecological conditions. Journal of the American Water Resources Association 36:1,247–1,261.

Morley, S.A., and J.R. Karr. 2002. Assessing and restoring the health of urban streams in the Puget Sound basin. Conservation Biology 16:1,448–1,509.

Morse, C.C., A.D. Huryn, and C. Cronan. 2003. Impervious surface area as a predictor of the effects of urbanization on stream insect communities in Maine, U.S.A. Environmental Monitoring and Assessment 89:95–127.

Moulton, S.R., J.L. Carter, S.A. Grotheer, T.F. Cuffney, and T.M. Short. 2000. Methods for analysis by the U.S. Geological Survey National Water Quality Laboratory: Processing, taxonomy, and quality control of benthic macroinvertebrate samples. U.S. Geological Survey, Open-File Report 00-212.

Moulton, S.R., II, J.G. Kennen, R.M. Goldstein, and J.A. Hambrook. 2002. Revised protocols for sampling algal, invertebrate and fish communities as part of the National Water-Quality Assessment Program. U.S. Geological Survey, Open-File Report 02-150.

Munn, M.D., and R.J. Gilliom. 2001. Pesticide toxicity index for freshwater aquatic organisms. U.S. Geological Survey, Water-Resources Investigation Report 01-4077.

Ourso, R.T., and S.A. Frenzel. 2003. Identification of linear and threshold reponses in streams along a gradient of urbanization in Anchorage, Alaska. Hydrobiologia 501:117–131.

Paul, M.J., and J.L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32:333–365.

Roy, A.H., A.D. Rosemond, M.J. Paul, D.S. Leigh, and J.B. Wallace. 2003. Stream macroinvertebrate response to catchment urbanization (Georgia, U.S.A). Freshwater Biology 48:329–346.

Shelton, L.R.. 1994. Field guide for collecting and processing stream-water samples for the National Water-Quality Assessment Program. U.S. Geological Survey, Open-File Report 94-455.

Sprague, L.A., D.A. Harned, D.W. Hall, L.H. Nowell, N.J. Bauch, and K.D. Richards. 2007. Response of stream chemistry during base flow to gradients of urbanization in selected locations across the conterminous United States, 2002–04. U.S. Geological Survey, Scientific Investigations Report 2007–5083.

SPSS. 2007. SYSTAT 12 Statistics I & II. SPSS, Inc., Chicago, IL.

Tate, C.M., T.F. Cuffney, G. McMahon, E.M.P. Giddings, J.F. Coles, and H. Zappia. 2005. Use of an urban intensity index to assess urban effects on streams in three contrasting environmental settings. In L.R. Brown, R.M. Hughes, R. Gray, and M.R. Meador, eds., Effects of Urbanization on Stream Ecosystems, pp. 291–315. American Fisheries Society, Symposium 47, Bethesda, MD.

U.S. Environmental Protection Agency. 1997. Determination of carbon and nitrogen in sediments and particulates of estuarine/coastal waters using elemental analysis, rev. 1.4. U.S. Environmental Protection Agency, Office of Research and Development, National Exposure Research Laboratory, Washington DC.

U.S. Geological Survey. Variously dated. National field manual for the collection of water-quality data. U.S. Geological Survey, Techniques of Water-Resources Investigations, Book 9, Chapters A1–A9, ver. 2. [online] URL: http://pubs.water.usgs.gov/twri9a. Accessed 18 June 2009.

U.S. Geological Survey. 2003. National elevation data set. U.S. Geological Survey, Earth Resources Observation and Science Data Center, Sioux Falls, SD. [online] URL: http://ned.usgs.gov/.

Walsh, C.J., A.K. Sharpe, P.F. Breen, and J.A. Sonneman. 2001. Effects of urbanization on streams of the Melbourne region, Victoria, Australia: I. Benthic macroinvertebrate communities: Freshwater Biology 46:535–551.

Yoder, C.O., R.J. Miltner, and D.White. 1999. Assessing the status of aquatic life designated uses in urban and suburban watershed. In National Conference on Retrofit Opportunities for Water Resource Protection in Urban Environments, Proceedings, Chicago, IL, 9–12 February 1998, pp. 16–28.. U.S. Environmental Protection Agency, Office of Research and Development, EPA/625/R-99/002.

Zaugg, S.D., M.W. Sandstrom, S.C. Smith, and K.M. Fehlberg. 1995. Methods of analysis by the U.S. Geolgocial Survey National Water-Quality Laboratory: Determination of pesticides in water by C-18 solid-phase extraction and capillary-column gas chromatography/mass spectrometry with selected-ion monitoring. U.S. Geological Survey, Open-File Report 95-181.