Recovery Potential Metrics Summary Form

Indicator Name: WATERSHED PERCENT URBAN

Type: Stressor Exposure

Rationale/Relevance to Recovery Potential: Urbanization of a watershed results in multiple stressors to a watershed, such as increased pollutant loads from stormwater runoff, altered stream flow, decreased bank stability, and increased water temperatures. The significance of this metric in reducing recovery potential is based on the multiple impacts to the watershed as well as the nearly irreversible nature of imperviousness. (See also Watershed Impervious Surface under Stressor Exposure indicators.) See literature cites below for numerous other effects.

How Measured: Measured as a percent of the area of a watershed with a land use classification of "urban" (i.e. low, medium, and high density residential; commercial; industrial; etc).

Data Source: Land cover sources include the National Land Cover Data from 1992 (See: http://www.epa.gov/mrlc/nlcd.html)and 2001 (See: http://www.epa.gov/mrlc/nlcd-2001.html) as well as various state sources. Temporal urban mapping is available from the USGS Land Cover Institute (See: http://landcover.usgs.gov/urban/umap/). Approximate watershed boundaries can be constructed by aggregating small-scale catchments from the NHDplus datasets (See: http://www.horizon-systems.com/nhdplus/).

Indicator Status (check one or more)

______ Developmental concept.
___x__ Plausible relationship to recovery.
______ Single documentation in literature or practice.
___x__ Multiple documentation in literature or practice.
______ Quantification.

Examples from Supporting Literature (abbrev. citations and points made):

- Alberti *et al.* (2007) reported that the configuration or pattern, as well as the amount, of urbanization, was significantly related to macroinvertebrate IBI scores.
- (Potter et al 2004)The resulting vulnerability models indicate that North Carolina watersheds with less forest cover are at most risk for degraded water quality and stream habitat conditions. Meanwhile, research has shown less diverse and more intolerant macrobenthic communities to be correlated with agricultural land cover (Lenat and Crawford 1994, Richards and others 1996, Weigel and others 2000, Genito and others 2002) and urban land use (Lenat and Crawford 1994, Morley and Karr 2002, Morse and others 2003, Roy and others 2003, Volstad and others 2003, Wang and Kanehl 2003).
- (Potter et al 2004) Forested land cover, at both the watershed and riparian scales, was a statistically significant predictor of benthic macroinvertebrate communities that are less tolerant of stream degradation, and that indicate a greater level of aquatic ecological integrity and better water quality. The opposite was the case for agricultural land cover at the watershed and riparian scales, and developed land cover in riparian zones.
- (wang 2001) The results shown in Table 5 indicate that the land-use components within the catchments could be major predictors for biotic integrity. The percentage of urban land was the second strongest predictor for both IBI and ICI. The negative signs of those coefficients indicate that as the intensity of human activities increase there is a tendency that the biological integrity of the rivers decreases. The percentage of wooded land was the third strongest predictor for IBI.
- (ourso and frenzel 2002) Increasing substrate embeddedness and bank erosion have been observed to increase in developing areas (Arnold et al., 1982; Furniss et al., 1991)

- (wang 2001) After statistically analyzing the spatial patterns of the water quality in receiving rivers and land uses and other point pollution sources in the watershed, the results showed that the water biotic quality did not degrade significantly below wastewater treatment plants. However, significantly lower water quality was found in areas downstream from high human impact areas where urban land was dominated or near point pollution sources.
- (Moore and Palmer 2005) Here we report on the study of 29 headwater streams showing that invertebrate diversity was extremely high in agricultural headwaters and dramatically declined as urbanization increased; however, the decline in diversity was less among urban streams if the urban streams had intact riparian forest buffers (1170).
- (Moore and Palmer 2005) There was a significant main effect of land use on the diversity and richness of macroinvertebrate communities (Table 2; *P* , 0.0001), with urban sites having lower richness and diversity than all other land use groups across both sampling years (1172).
- (Moore and Palmer 2005) There was a significant effect of land use on the richness of all functional feeding groups except shredders (Table 2). Collector, filterer, predator, and scraper richness were all significantly higher in the agricultural streams compared to the urban sites (Fig. 3) (1172-1173).
- (Kohler et al. 2004) Runoff from urban areas and golf courses is often presumed to be a significant contributor to nonpoint source (NPS) water pollution originating from the urban environment (285).
- (Kohler et al. 2004) At the same time, these developing urban areas struggle with stormwater management because urbanization decreases the amount of permeable surface available for absorption and infiltration of rainwater and snow melt. This increased runoff can potentially contain urban pollution from roofs, roads, and parking lots (Paul and Meyer, 2001) that is often carried directly to surface water. Increasing stormwater runoff and velocity magnify problems of conveyance, increase storage volume required to reduce flooding, and raises the impact of potential contaminants such as oils, sediment, and heavy metals (286).
- (Nelson and Booth 2002) However, urbanization can also indirectly increase channel erosion and downstream sedimentation by increasing the frequency and volume of channel-altering storm flows (Leopold, 1968; Hammer, 1972) (52).
- (Nelson and Booth 2002) Here and elsewhere, increased discharges resulting from urbanization cause channels to permanently enlarge to accommodate the new flow volumes (64).
- (Morgan and Cushman 2005) The ''urban stream syndrome'' prevails when human population density reaches a critical limit within a catchment (Paul and Meyer 2001, Groffman et al. 2005, Meyer et al. 2005). Such modification in stream structure and function often results in degraded physiochemical conditions and associated changes in biota (Roth et al. 1999, Paul and Meyer 2001, Gergel et al. 2002, Meyer et al. 2005, Walsh et al. 2005a) (643).
- (Morgan and Cushman 2005) Paul and Meyer (2001) noted that urbanization is second only to agriculture as an agent of stream degradation in the USA (see also USEPA 2000). Once catchments are urbanized, intermittent and perennial streams may show altered hydrologic regimes, elevated nutrient and contaminant concentrations, and degraded biota, which may be difficult to mediate or reverse (Paul and Meyer 2001, Groffman et al. 2003, Booth 2005) (643).
- (Morgan and Cushman 2005) The 6 studies cited in Paul and Meyer (2001) generally found changes in either fish diversity or indices of biotic integrity with increasing catchment imperviousness, with changes typically occurring at 10 to 12% imperviousness (e.g., Klein 1979, Steedman 1988, Wang et al. 1997, Yoder et al. 1999) (643).
- (Morgan and Cushman 2005) In particular, altered flow regimes from urbanization can affect fish assemblage structure and biodiversity (Poff and Allan 1995, Bunn and Arthington 2002, Roy et al. 2005). Flow shapes stream physical habitat, with concomitant influences on biotic composition; yet, fish populations often have evolved life histories

that reflect natural flow regimes (Bunn and Arthington 2002). Rapid alterations in flow regimes in urbanizing streams, which may be the case in Maryland streams (CWP 2003), may have occurred on too short a time scale (years to decades) to allow populations to respond, thus exacerbating the urban syndrome (Groffman et al. 2003, Booth 2005) (652).

- (Morgan and Cushman 2005) Ricciardi and Rasmussen (1999) noted that human population growth is a major factor related to fish species extinction, especially in urbanizing areas. Unfortunately, conservation practices minimizing impact of urbanization on local or regional fish assemblages, especially in the Chesapeake Bay Catchment, may be inadequate, too late, or too expensive to protect intolerant fishes because of the invasiveness and nonreversibility of urbanization. For example, it will be logistically difficult, if not politically impossible, to reverse road density and catchment imperviousness within urban Maryland and throughout the USA (Brabec at al. 2002). Wang et al. (2001) and Wang and Kanehl (2003) both suggested that minimizing effective catchment imperviousness (Walsh et al. 2005a), or restricting total imperviousness (especially to ,10–15%), may be critical to maintaining species assemblages (Gergel et al. 2002, Groffman et al. 2003); we believe this recommendation also may be useful in protecting Maryland stream fishes. Loss of fish refugia needed to maintain biodiversity within streams in urbanizing catchments is an environmental concern within Maryland (Richter et al. 1997) (653).
- (Groffman et al. 2003) Urbanization is one of the most dramatic and dynamic global human alterations of ecosystems (Grimm *et al*. 2000; Pickett *et al*. 2001). The most marked effects are on hydrology and streams, with impervious surfaces altering the rates and pathways of water movement into, through, and out of ecosystems (Arnold and Gibbons 1996; Paul and Meyer 2001) (315).
- (Groffman et al. 2003) The most obvious hydrologic changes associated with urbanization are the engineering of stream channels, in which natural features are replaced by concrete channels and streambank stabilization efforts designed to resist increased flood flows. Extensive piped storm drainage networks often completely bypass riparian zones, channeling large amounts of water from impervious surfaces directly into streams, both quickly and with increased frequency (Paul and Meyer 2001; CWP 2003). A result of this altered hydrology is that incision or “downcutting” is a common feature of urban stream channels (Wolman 1967; Henshaw and Booth 2000). Downcutting results from large volumes of water scouring out sediment that has accumulated during agricultural activity and/or residential construction in the watershed (Figure 3). Incision is especially marked in watersheds with old and/or stable urban land use, where there are few sources of sediment to replace material scoured by high flows (317).
- (Gage et al. 2004) Insect communities were more diverse in streams draining low disturbance watersheds than in streams draining highly developed watersheds. Sensitive taxa were found in streams with extensively forested watersheds, but were nonexistent in extensively developed watersheds. Disturbances occurring in streams caused declines in diversity, often eliminating sensitive taxa. Aquatic insect diversity is related to land use patterns and disturbances, and anthropogenic alteration of habitat has negative consequences to that diversity (345).
- (Gage et al. 2004) Urban runoff may cause decreased stream stability, and increased turbidity and sedimentation (Heimann and Roell 2000, Oberlin et al. 1999, Pitt et al. 1995, Winter and Duthie1998) (345-346).
- (Gage et al. 2004) Our research supports the link between urbanization, sedimentation, and declining aquatic biodiversity, and shows that development threatens small headwater streams. Streams with a high level of development and nearby construction had less diverse communities with lower abundance than streams with more heavily forested watersheds. High percentages of developed land within a watershed can help explain stream insect diversity, but direct disturbance overrides those effects (355).

- (Gage et al. 2004) Abundance was significantly higher in streams that had less than 10% developed land in their watershed than in streams that had > 25% developed land in their watershed (Fig. 3a; t-test: t = 8.5, df = 133, P < 0.0001) (351-352).
- (Gage et al. 2004) The number of families was significantly greater in streams that had < 10% developed land in their watershed than in streams that had > 25% developed land (Fig. 3b; t-test: t = 13.90, df = 133, P < 0.0001). As with abundance, despite the variability in irregular disturbance streams, the percentage of developed land in a watershed is one factor that may affect insect diversity (Fig. 3b) (353).
- (Walsh et al. 2005) In virtually all studies, sensitive species were absent or less abundant in streams draining urban areas. Globally, streams in urban areas are characterized by species-poor assemblages, consisting mostly of disturbance-tolerant taxa (712).
- (Walsh et al. 2005) Most studies have found that stream fish assemblages respond to catchment urbanization in a similar pattern to macroinvertebrates: a loss or reduced abundance of sensitive species, and a less diverse assemblage numerically dominated by disturbance-tolerant species (e.g., Roth et al. 1996, Wang and Lyons 2003). Such a trend was observed in streams of Atlanta (Roy et al. 2005), and in streams of the eastern Piedmont physiographic region of Maryland (Morgan and Cushman 2005). Similar results were reported in lower Piedmont streams of the southeast where fish health (as indicated by % of fish with eroded fins, lesions, or tumors), and proportions of sensitive breeding guilds (% lithophilic spawners) decreased with increasing urbanization (Helms et al. 2005) (712).
- (Voelz et al. 2005) Runoff from urban areas can be extensive, contain numerous chemicals and cause increased temperatures and sediment loads in receiving water bodies (Novotny and Olem, 1994; Paul and Meyer, 2001). Relative to agricultural lands, urban areas can have similar land applications of some chemicals (e.g., phosphorus and herbicides; Creason and Runge, 1992) and the runoff from urban areas can contain a much greater variety of pollutants (Novotny and Olem, 1994). Untreated stormwater runoff from urban areas can contain levels of some parameters (e.g., total solids) that exceed those found in untreatedwastewater (Pitt and Field, 1977). These pollutants entering streams as a result of urbanization can be harmful to stream organisms (e.g., Schlosser, 1982; Kemp and Spotila, 1997; Paul and Meyer, 2001) (176).
- (Paul and Meyer 2001) This extensive and ever-increasing urbanization represents a threat to stream ecosystems. Over 130,000 km of streams and rivers in the United States are impaired by urbanization (USEPA 2000). This makes urbanization second only to agriculture as the major cause of stream impairment, even though the total area covered by urban land in the United States is minor in comparison to agricultural area. Urbanization has had similarly devastating effects on stream quality in Europe (House et al. 1993) (334).
- (Paul and Meyer 2001) A further result of increased runoff is a reduction in the unitwater yield: a greater proportion of precipitation leaves urban catchments as surface runoff (Figure 1) (Espey et al. 1965, Seaburn 1969). This reduces groundwater recharge and results in a reduction of baseflow discharge in urban streams (Klein 1979, Barringer et al. 1994) (335).
- (Paul and Meyer 2001) Urbanization affects many elements of importance to stream heat budgets. Removal of riparian vegetation, decreased groundwater recharge, and the "heat island" effect associated with urbanization, covered more fully in a companion review (Pickett et al. 2001), all affect stream temperature (Pluhowski 1970), yet very little published data exists on temperature responses of streams to urbanization. In one study on Long Island urban streams had mean summer temperatures 5–8±C warmer and winter temperatures 1.5–3±C cooler than forested streams. Seasonal diurnal fluctuations were also greater in urban streams, and summertime storms resulted in increased temperature pulses 10–15±C warmer than forested streams, a result of runoff from heated impervious surface (Pluhowski 1970). Similar effects on summer temperatures and daily fluctuations have also been observed elsewhere (Table 1) (Galli 1991, Leblanc et al. 1997) (341).

- (Paul and Meyer 2001) However, the Ohio Environmental Protection Agency has a very large database of land use and fish abundance from around their state and has suggested three levels of general fish response to increasing urbanization: from 0 to 5%urban land use, sensitive species are lost; from 5 to 15%, habitat degradation occurs and functional feeding groups (e.g., benthic invertivores) are lost; and above 15% urban land use, toxicity and organic enrichment result in severe degradation of the fish fauna (Table 1) (Yoder et al. 1999). This model has not been verified for other regions of the country, where studies have focused on various aspects of urbanization (351).
- (Paul and Meyer 2001) A few studies have actually examined ecological factors regulating stream fish populations and communities in urban streams. Recruitment of anadromous fish in the Hudson River Basin in New York was limited by suitable spawning habitat as a result of urbanization (Limburg & Schmidt 1990). Numbers of alewife (*Alosa pseudoharengus*) eggs and larvae in tributary streams decreased sharply between 0 and 15% urban land use. Beyond 15%, no eggs or larvae were found (353).
- (Paul and Meyer 2001) Large multi-site studies of fish responses to urban gradients also find dramatic decreases in diversity or fish multimetric indices [index of biotic integrity (IBI)] with increasing ISC or other urban land use indicators (Table 1) (Klein 1979, Steedman 1988,Wang et al. 1997, Frick et al. 1998, Yoder et al. 1999). Similar to effects observed for invertebrates, these studies also find precipitous declines in fish metrics between 0 and 15% ISC or urban land use, beyond which fish communities remain degraded (Klein 1979, Yoder et al. 1999) (352).
- (Paul and Meyer 2001) Along urban gradients within single catchments, fish diversity and abundances decline, and the relative abundance of tolerant taxa increases with increasing urbanization (Table 1) (Onorato et al. 2000, Boet et al. 1999, Gafny et al. 2000). Invasive species were also observed to increase in more urbanized reaches of the Seine River, France, and this effect extended more than 100 km below Paris (Boet et al. 1999) (351).
- (Paul and Meyer 2001) All single catchment gradient studies find a decrease in invertebrate diversity as urban land use increases, regardless of the size of the catchment (Pratt et al. 1981, Whiting & Clifford 1983, Shutes 1984, Hachmoller et al. 1991, Thorne et al. 2000). Decreases were especially evident in the sensitive orders—Ephemeroptera, Plecoptera, and Trichoptera (Pratt et al. 1981, Hachmoller et al. 1991). Most of these studies observed decreases in overall invertebrate abundance, whereas the relative abundance of Chironomidae, oligochaetes, and even tolerant gastropods increased (Pratt et al. 1981, Thorne et al. 2000). Comparative catchment studies show the same trends with increasing urbanization as those observed in single catchment studies: decreased diversity and overall abundance and increased relative abundance of tolerant Chironomidae and oligochaetes (Medeiros et al. 1983, Garie & McIntosh 1986, Pederson & Perkins 1986, Lenat & Crawford 1994) (349).
- (Paul and Meyer 2001) All aspects of aquatic invertebrate habitat are altered by urbanization (349).
- (Paul and Meyer 2001) However, general effects of urbanization on stream invertebrates have also been studied and general invertebrate responses can be summarized as follows: decreased diversity in response to toxins, temperature change, siltation, and organic nutrients; decreased abundances in response to toxins and siltation; and increased abundances in response to inorganic and organic nutrients (Resh & Grodhaus 1983, Wiederholm 1984) (349).
- (DeLuca et al., 2004) Accompanying these population increases are numerous anthropogenic landscape modifications, including expansion of urban/suburban development. The cumulative result of these pressures is that the world's coastal ecosystems, including the Chesapeake Bay estuary, have become degraded (Nordstrom and Roman 1996, Edgar et al. 2000) (837).
- (DeLuca et al. 2004) Other studies have also identified ecological thresh olds or non-linear responses to anthropogenic disturbances. Interestingly, most reported thresholds were found between 10 and 20% disturbance, similar to the results of our study. For

example, Paul and Meyer (2001) reported thresholds of stream degradation when impervious surface in stream catchments exceeded 10 to 20%. Urban disturbance has also been linked to degradation of fish communities with as little as 10 (Limburg and Schmidt 1990) and 20% (Wang et al. 1997) impervious surface area within stream catchments. These studies, in concert with our findings, provide strong evidence that the response of biological communities dependant upon various aquatic systems respond to anthropogenic disturbances in similar, nonlinear manners (i.e., threshold response) (844).

- (Bernhardt and Palmer 2007) The world's population is increasingly urban, and streams and rivers, as the low lying points of the landscape, are especially sensitive to and profoundly impacted by the changes associated with urbanization and suburbanization of catchments (738).
- (Bernhardt and Palmer 2007) Restoration in urban streams is both more expensive and more difficult than restoration in less densely populated catchments. High property values and finely subdivided land and dense human infrastructure (e.g. roads, sewer lines) limit the spatial extent of urban river restoration options, while stormwaters and the associated sediment and pollutant loads may limit the potential for restoration projects to reverse degradation (738).
- (Bernhardt and Palmer 2007) The ecological impacts of this growth and population re-distribution are profound. The loss of forests and agricultural lands to urbanization influences local climate and air quality, alters energy and nutrient flows and leads to decreased native biodiversity (Vitousek et al., 1997; Grimm et al., 2000; Alberti et al., 2003; Dudgeon et al., 2006). As running waters occupy the lowest-lying areas on the landscape, they integrate the effects of land-use change and thus are very sensitive to urbanization. As land is cleared of vegetation and replaced with a large amount of impervious surface such as asphalt, concrete and rooftops, the amount of run-off entering streams increases; the hydrology and geomorphology of receiving streams are fundamentally altered; and the consequences for ecological changes can be severe and complex (Wolman, 1967; Walsh, 2000; Paul & Meyer, 2001). Urbanization simultaneously increases the loading of water and nutrients while simplifying receiving stream channels, turning the urban river from a functioning ecosystem to an efficient gutter (738-739).
- (Bernhardt and Palmer 2007) The most obvious and immediate consequences are an increase in impervious surface area with resultant increased runoff to receiving streams, higher peak discharges, greater water export and higher sediment loads during the construction phase (Dunne & Leopold, 1978; Arnold & Gibbons, 1986; McMahon & Cuffney, 2000; Rose & Peters, 2001; Nelson & Booth, 2002; Walsh, Fletcher & Ladson, 2005a). Over time as the catchment is built out (new construction slows or ceases), the hydrologic alterations remain but sediment delivery to streams decreases dramatically (Trimble, 1997; Wheeler, Angermeier & Rosenberger, 2005), leading to channel erosion and sometimes dramatic increases in channel width and depth (incision) (Booth, 2005; Leopold, Huppman & Miller, 2005). These changes in channel morphology disconnect the stream from its floodplain, decrease sinuosity, and homogenise stream profiles (Hammer, 1972; Douglas, 1974; Roberts, 1989; Booth, 1990). Leopold, Huppman & Miller (2005) described these hydrogeomorphic changes as part of the 'urbanization cycle' in small river basins. These impacts have historically been exacerbated by sealed and piped drainage systems, as well as channelisation, which is often used for reducing lateral channel migration and managing flow to protect urban infrastructure (Dunne & Leopold, 1978) (739).
- (Bernhardt and Palmer 2007) Analogous to Leopold's 'urbanization cycle', others have referred to the predictable changes as the 'urban stream syndrome' (Meyer et al., 2005; Walsh et al., 2005b), noting that the physical effects on urban streams are often associated with reduced biotic richness (Arnold & Gibbons, 1986; Makepeace, Smith & Stanley, 1995; Paul & Meyer, 2001; Meyer et al., 2005; Walsh et al., 2005a). Thus one might use the term 'generic' to describe urban streams, making the point that despite important differences in catchment geology, climate and vegetation, the condition of

urban streams is overwhelmingly controlled by the altered timing and volume of water, sediment, nutrients and contaminants resulting from the urbanized catchment (739-740).

- (Bernhardt and Palmer 2007) An altered hydrograph with high peak flows and reduced baseflows is the most obvious and consistent effect of catchment urbanization on stream hydrology. As a result of increasing impervious cover in developing catchments, evapotranspiration and soil infiltration are reduced. The result is higher peak discharges, flashier stream flows, and reduced groundwater–surface water exchange with potentially an overall reduction in groundwater recharge and hyporheic zone size (e.g. Delleur, 2003; Groffman et al., 2003; Groffman & Crawford, 2003). Since groundwater storage is reduced, many urban streams also experience reduced baseflow. In most cities, urban stormwater drainage systems exacerbate the problem of high peak flows, with piped storm drainage networks efficiently routing stormwater directly into stream channels (Booth & Jackson, 1997; Walsh et al., 2005a). These stormwater/sewer networks effectively bypass the river floodplain (and sewage treatment plants), routing contaminants directly from roads and buildings into surface waters (Paul & Meyer, 2001; CWP, 2003) (740).
- (Bernhardt and Palmer 2007) Engineers and public works managers have historically sought to maintain channels 'unchanging in shape, dimensions and pattern' (Schumm, 1977). This desire for physical channel stability has led to highly simplified urban stream channels – in the most extreme cases urban streams are confined in concrete channels or routed through underground pipes. More commonly the banks of urban streams have been 'hardened' using over-sized boulders or rip-rap to prevent lateral channel migration and bank erosion. Often, these hardened streams are far from physically stable in the traditional sense that there is no progressive adjustment in channel form (Schumm, 1977; Henshaw & Booth, 2000), yet urban stream channels often undergo progressive enlargement and erosion (Hammer, 1972; Leopold et al., 2005). A highly impacted urban stream channel often has little variation in depth or the particle sizes of bed material. Downcutting or channel incision is a common feature of urban stream channels as a result of high volume scouring flows and lateral constraints to channel migration (Wolman, 1967; Henshaw & Booth, 2000) (740).
- (Bernhardt and Palmer 2007) Not surprisingly, given their flashy hydrographs, low habitat heterogeneity and high contaminant loads, this recent research has documented that urban fish and invertebrate assemblages are typically species poor (Wang et al., 2000; Freeman & Schorr, 2004; Miltner, White & Yoder, 2004; Walsh, 2004; Moore & Palmer, 2005; Morgan & Cushman, 2005). In Baltimore (MD, U.S.A.) and Washington (DC, U.S.A.) we have found urban streams that are not contaminated with sewage have very low levels of benthic organic matter (E. S. Bernhardt & M. A. Palmer, Fig. 2), a result that corroborates results from Atlanta streams (GA, U.S.A.) by Meyer et al. (2005). While this has been suggested as a factor that may limit community metabolism and nutrient retention in urban systems (Grimm et al., 2005; Meyer et al., 2005; M. A. Palmer, Fig. 3), recent work suggests that urban streams in older cities or near developments with septic systems have high amounts of dissolved organic matter (Kroeger, Cole & Valiela, 2006, S. Kaushal, pers. comm.). Impaired ecosystem functioning can extend out of the channel into the riparian zone, if the water table drops below the rooting zone of riparian plants because of channel incision (Groffman et al., 2003). These functionally disconnected riparian zones in urban catchments may have reduced efficiencies of nutrient removal (Groffman et al., 2002, 2003). However, uptake rates in urban streams can be quite high and variable (Grimm et al., 2005; Meyer et al., 2005, Fig. 3). This variability is due in part to large differences among urban sites with respect to geomorphology and water quality, as urban channels vary from concrete beds to earthen channels with some riparian vegetation and water quality varies from slightly to extremely polluted conditions (741).
- (Bernhardt and Palmer 2007) While bank stabilisation projects are fairly successful in rural and agricultural areas, their success rate is often lower in urban areas where they often cannot withstand storm flows, and where high flows and scarcity of transportable sediment create high erosive potential (Wolman, 1967; Ferguson, 1991; Thompson, 2002) (745).

- (Bernhardt and Palmer 2007) Spending large amounts of money on a restoration project along one kilometre of stream while the downstream kilometre remains under pavement, will not restore the ecological conditions of the stream network (Palmer et al., 2005) (746).
- (Bernhardt and Palmer 2007) While urban streamwater throughout the developed world has become progressively cleaner with effective wastewater treatment technologies, there are still significant contaminant problems in urban catchments. Sewer and stormwater pipes often run alongside streams, and many pipes leak directly into streams and riparian zones. During stormwater pulses, many cities have combined sewer stormwater overflows, such that when stormwater inputs are too high, raw sewage combined with surface runoff is allowed to overflow directly into urban streams (Chen et al., 2004). The uncontrolled connection between sewage and surface water leads to high fecal coliform concentrations and nutrient loads in many urban streams (Makepeace et al., 1995; Miltner et al., 2004). Surface runoff brings heavy metals from parking lots and roofs and carries fertilizer nutrients, herbicides and pet wastes from lawns, golf courses and parks (Makepeace et al., 1995; Yuan, Hall & Oldham, 2001; Beasley & Kneale, 2002). In addition, many pharmaceutical compounds persist in urban surface waters despite standard water treatment procedures (e.g. Stackelberg et al., 2004) (746).
- (Bernhardt and Palmer 2007) In some severely degraded catchments, restoration efforts may be doomed to failure and wise catchment management should first invest in improved water retention, detention and conveyance systems water treatment (747).
- (Bernhardt and Palmer 2007) Urbanization of catchments leads to changes of streams along three axes: (i) geomorphic simplification in that habitat heterogeneity and floodplain connectivity are reduced; (ii) diminished societal value in that stream channels become increasingly unattractive and are avoided for recreational purposes; and (iii) ecological simplification in that stream biodiversity declines and stream ecosystem functioning is impaired, resulting, for example, in a reduced capacity of streams to reduce downstream nutrient losses (747).
- (Bernhardt and Palmer 2007) Restoration of urban stream channels is highly constrained, thus it is unlikely that an urban stream will ever be restored to its preurbanization state (747).
- (Freeman et al., 2007) Urbanization imposes multiple stressors on salmon (larger and more frequent peak flows, habitat simplification, increased concentrations of toxins) and has severe deleterious effects on salmon populations. By altering flow pathways of precipitation to headwater streams (Figure 1), such as when infiltration rates are reduced by soil compaction or paving, urbanization can elevate local stormflows by 2-5 times, causing rapid channel erosion and biotic simplification (Wolman and Schick, 1967; Hollis, 1975; Booth and Jackson, 1997; Booth et al., 2002). However, the effects of urbanization on salmonids may depend on the spatial pattern of development and stream disturbance. For example, relatively healthy salmonid populations in streams in the vicinity of Seattle, WA, occur only in streams with intact headwaters (Fresh and Lucchetti, 2000). Minimizing disturbance to headwater streams may increase the ecological resilience of these stream systems (11).
- (Sondergaard and Jeppesen 2007) Lakes in highly populated or intensively cultivated areas have experienced increased nutrient loading, resulting in turbid water and loss of biodiversity (Wetzel 2001) (1090).
- (Freeman and Marcinek 2006) Population growth and urbanization are encroaching on aquatic habitats that support high levels of aquatic biodiversity and endemism, as well as supporting imperiled species (Abell and others 2000; Warren and others 2000). Threats to native biodiversity caused by altered runoff and pollution from urbanizing areas are likely to be compounded by water supply development, largely dependent on surface water in the Piedmont, unless specific management actions are taken to safeguard vulnerable streams (435).
- (Freeman and Marcinek 2006) For example, isolation by reservoirs (upstream and downstream) as well as close proximity to downstream urban areas and point-source

discharges are likely to diminish local species assemblages, whereas connections with nearby tributary systems having intact faunal communities are likely to augment local species richness, independently of flow alteration effects. The observation that water supply variables do improve predictive models for richness of fluvial-dependent species (or probability that a site scores as impaired) implies that decisions concerning how to supply water for offstream uses will have measurable consequences for biotic integrity, even though other landscape factors may add to or modify those effects (445-446).

- (Moore and Palmer 2005) Two factors, the amount of impervious surface and of riparian forest cover, are often the focal point of discussions on the link between land use change and stream ecosystem health (e.g., Schueler 1994, Weigel et al. 1999, Stewart et al. 2001). These two variables influence stream hydrology and water quality (Brabec et al. 2002). Furthermore, impervious cover has been shown to be correlated with the diversity of macroinvertebrates (Schueler 1994), and the removal or clearcutting of riparian trees in forested watersheds has been shown to have a strong influence on entire stream invertebrate communities (Wallace et al. 1997) (1170).
- (Moore and Palmer 2005) Agricultural practices such as livestock grazing and tilling on land adjacent to streams can lead to soil erosion and subsequent runoff of fine sediments, nutrients, and pesticides (e.g., Schulz and Liess 1999, Cuffney et al. 2000, Kang et al. 2001). Urbanization leads to enhanced runoff, channel erosion, and reduced water quality due to inputs of metals, oils, and road salts (Hammer 1972, Booth and Jackson 1997, Paul and Meyer 2001) (1169).
- (Brett et al., 2005) Urban land use, through alterations of physical and hydrologic features of watersheds and the production of additional anthropogenic nutrient sources (i.e., lawn fertilizers, pet waste, septic tank effluent, accelerated erosion) are thought to be an important cause of lake and stream water quality degradation (US EPA 1990; Carpenter and others 1998; Tufford and others 1998) (330).
- (Brett et al., 2005) The loss of forested cover with urbanization in humid areas minimizes recycling and uptake of nutrients due to reduced microbial and vegetative processes that immobilize nutrients in the forest canopy, forest litter, forest soils, and organic material (Wahl and others 1997; Abelho 2001). Moreover, replacement of forest areas with impervious surfaces is associated with long-term changes in the quantity and composition of suspended solids inputs, which, in turn, act as an important transport mechanism for nutrients in streams through flocculation, adsorption, and colloidal action (Stone and Droppo 1994; Mulliss and others 1996) (331).
- (Brett et al., 2005) Urban streams typically have larger peak flows during precipitation events and lower base flows during seasonal dry periods (Booth 1991; Paul and Meyer 2001), both of which can impact sediment and associated nutrient (especially phosphorus) transport (331).
- (Voelz et al., 2005) In conclusion, data on macroinvertebrates and water chemistry showed that both rivers are negatively impacted by urban land use (198).
- (Voelz et al., 2005) Once these streams enter urban areas, they do not recover to reference condition as they move downstream (198).
- (Allmendinger et al., 2007) Urbanization causes profound changes in patterns of erosion and sedimentation in watersheds. Impervious surfaces and compacted soils increase runoff (Leopold and Skibitzke, 1967; Hollis, 1975; Sauer et al., 1983), leading to bank erosion, channel enlargement, and channel incision (Hammer, 1972; Morisawa and LaFlure, 1979; Arnold et al., 1982; Peck, 1986; Neller, 1988). Upland sediment production may dramatically increase during construction, but after construction has ceased, buildings, lawns, and roadways are widely believed to produce relatively little sediment (Dawdy, 1967; Wolman, 1967; Wolman and Schick, 1967) (1483).
- (Allmendinger et al., 2007) Changes in flow regime associated with urban development can have dramatic effects on the structure of ecological communities and the rates of biological processes (Poff and Nelson-Baker, 1997; Palmer et al., 2002; Nilsson et al., 2003). Higher percentages of impervious surfaces in a watershed, for example, may be associated with decreases in invertebrate species richness (Moore and Palmer, 2005),

and increased suspended sediment concentrations can have numerous important ecological effects (Waters, 1995). In downstream areas, changes in nutrient and sediment loading can adversely affect receiving waterways; this is a significant concern in the watersheds that drain to the Chesapeake Bay (Brush, 1989; Cronin and Vann, 2003; Kemp et al., 2005) (1484).

- (Allmendinger et al., 2007) In the 1970s, Montgomery County was one of the most rapidly urbanizing counties in the nation, and during periods of active construction in the Little Patuxent watershed, suspended sediment concentrations increased significantly (Roberts and Pierce, 1974) (1485).
- (Allmendinger et al., 2007) Urbanization causes extensive changes in hydrologic processes, including increased peak flows and decreased low flows (Leopold and Skibitzke, 1967; Hollis, 1975; Sauer et al., 1983; Moglen et al., 2004) (1486).
- (Allmendinger et al., 2007) In a study of Watts Branch, an urbanized watershed in Montgomery County, Maryland, Beighley and Moglen (2002) presented hydrological modeling results suggesting that the two-year peak discharge increased by factors from 1.3 to 3.0 from 1951-2007 for second-order subwatersheds similar in size to the Good Hope Tributary. A similar analysis was presented by Palmer et al. (2002) for the NW Branch watershed from 1951 to 1997. The NW Branch watershed is immediately adjacent to the Good Hope tributary. Palmer et al. (2002) reported that peak discharge in second-order subwatersheds of the NW Branch increased by factors from 1.3 to 7.7. The variability in these predictions is largely controlled by the extent of impervious cover associated with urbanization in each subwatershed (1486).
- (Allmendinger et al., 2007) The width of the water surface at Howden's (1949) three sites (Figure 1) increased significantly during the last 54 years (Figure 6, Table 2). The range of enlargement ratios is quite large, ranging from 1.06 along the Little Paint Branch to 3.56 along the Northeast Branch (the enlargement ratio as used here is defined as the width after urbanization divided by the width before urbanization) (1490).
- (Light and Marchetti 2007) Hydrologic modification (impoundments and diversions), invasions, and proportion of developed land were all predictive of the number of extinct and at-risk native fishes in California watersheds in the AIC analysis (434).
- (Light and Marchetti 2007) In California watersheds both the richness of nonindigenous fishes (Marchetti et al. 2004) and overall homogenization of the fish fauna (Marchetti et al. 2001, 2006) are positively associated with a variety of measures of habitat alteration, including urbanization, agriculture, and hydrologic modification (impoundments and diversions) (435).
- (Light and Marchetti 2007) Fish species richness and species composition in California watersheds have been markedly altered over the last 150 years by both invasions and extinctions, and these alterations were associated with many forms of watershed alteration, including development, agriculture, and hydrologic alteration (Table 2) (439).
- (Ekness and Randhir 2007) Species richness has been shown to increase as vegetative density increases and with distance from developed areas (1470).
- (Ekness and Randhir 2007) The urban environment is also characterized by the displacement of native by nonnative species (Dickman, 1987).

  Disturbance caused by urbanization affects water flow and water velocity in stream systems, and can increase sediment load (Hession et al., 2000). Increased storm discharge results in higher erosion and subsequent scouring of the stream channel, reduces the pool-to-riffle ratio, and changes the aquatic habitat that supports the fish and invertebrate insects (Pizzuto et al., 2000). Stream channels subjected to increased urban discharges often become wider (Hammer, 1972; Leopold, 1973; Hession et al., 2000; Pizzuto et al., 2000) and have higher water temperatures both of which affect stream habitats. The increase in impervious cover that comes with urban development decreases infiltration into ground water and increases runoff. All of these are consequences of urban land use being detrimental to watershed ecosystems (1470-1471).

- (Ekness and Randhir 2007) Industrial, residential, and agricultural sources of disturbance can increase loading of total nitrogen and phosphorus in running waters and impact the ecology of a river system (Pieterse et al., 2003) (1471).
- (Walsh et al., 2005) Increased concentrations and loads of several chemical pollutants in stream water appear universal in urban streams, often occurring even at low levels of catchment urbanization (Hatt et al. 2004). Even in regions where the ecological importance of stormwater-derived pollution is minor (Booth et al. 2004), positive correlations have been observed between catchment urbanization and concentrations of some streamwater pollutants (Horner et al. 1997). Urban catchments in the southwestern US, however, may show high variation in streamwater nutrient concentrations, and may even exhibit transient nutrient limitation (Grimm et al. 2005). Obviously, the problems of urban-induced water-quality impairment will be much greater in areas where sewage and industrial effluents are poorly managed (e.g., Schoonover et al. 2005), although controlling such impairment without addressing stormwater impacts is unlikely to ameliorate all water-quality problems (Hatt et al. 2004).

  Variation in water chemistry changes within and among urban areas with increasing urban land use can result from several causes: natural climatic or geological differences (e.g., urbanization increased conductivity in streams of eastern Melbourne, Australia, but diluted the more saline streams to the northwest of Melbourne, Walsh et al. 2001); from historical differences in land use that predate urbanization (Frost 1993, Iwata et al. 2003); or from differences in the age of urban land use (e.g., sediment loads may decline in streams draining older urban areas, Finkenbine et al. 2000) (710).
- (Walsh et al., 2005) Because of this hydrologic effect, or because direct engineering intervention often straightens channels or lines them with impermeable surfaces, reduction in channel complexity, and thus instream habitat, appears an almost universal symptom of the urban stream syndrome. In turn, channel incision and simplification, including reduction in hyporheic flow (Grimm et al. 2005) and hydrologic isolation from riparian vegetation (Groffman et al. 2003), often have important effects on several instream ecological processes (Fig. 1) (711).
- (Walsh et al., 2005) Nutrient uptake was reduced in more urbanized streams of both Georgia (Meyer et al. 2005) and desert streams of Arizona and New Mexico (Grimm et al. 2005). In Atlanta streams, reduced uptake likely occurred because of reduced abundance of fine benthic organic matter, which decreased as catchment urbanization increased (Meyer et al. 2005). In desert streams, reduced uptake rates in urban streams were attributable to reduced channel complexity (hence reduced transient zone storage), and possibly reduced primary productivity, with the latter likely occurring from direct application of algicides into streams (Grimm et al. 2005) (712).
- (Walsh et al., 2005) Other anthropogenic impacts that may or may not be associated with urban land use may obscure the relationship between stream condition and imperviousness. For instance, Miltner et al. (2004) found that effects of combined or sanitary sewer overflows, wastewater treatment plant effluents, and legacy pollutants occurred independently of the urban density gradient in Ohio streams (713).
- (Walsh et al., 2005) King et al. (2005) demonstrated that urban land cover was a better predictor of macroinvertebrate composition if it was inversely weighted by the distance from the sampling site (i.e., modelling a larger effect for closer urban land use than for more distant urban land use) (714).
- (Walsh et al., 2005) Symptoms of the urban stream syndrome that appear to occur consistently across regions are predominantly driven by urban stormwater runoff, which, in almost all urban areas of the world, has traditionally been managed for flood control by direct piped connection between impervious surfaces and streams (Fig. 1). Therefore, it is likely that stormwater impacts are the primary driver behind the often-reported correlations between stream condition and catchment imperviousness (713).
- (Walsh et al., 2005) In a study of paired reaches with and without riparian forest along an urban gradient in Pennsylvania, Hession et al. (2003a, b) found that the presence of riparian forest affected geomorphology, concentrations of bioavailable nutrients, and algal

biomass independently of urban effects. In contrast, assemblage composition of diatoms, macroinvertebrates, and fishes were associated with the urban density gradient, but were less strongly affected by the presence of riparian forest (Hession et al. 2003a).

Riparian forests certainly have important ecological links to stream ecosystems through their influence on water chemistry, organic matter input, and shading (e.g., Pusey and Arthington 2003). It is conceivable, therefore, that loss of riparian forest may severely limit the potential for recovery of streams impacted by urban land use (Fig. 1). However, even in catchments with intact riparian forests, channel incision and increases in impervious surfaces and piped drainage can interact to significantly lower riparian water tables and, thus, potentially reduce the interaction between the riparian zone and pollutants moving in shallow groundwater flow from uplands (Groffman et al. 2002) (714).

- (Roy et al., 2007) Landscapes are being developed and managed to meet human needs, subsequently altering stream hydrology, geomorphology, and water quality (Allan 2004). These changes, in turn, affect stream biotic assemblages by reducing fish richness, diversity, and density, particularly of endemic and pollution-sensitive species. Such associations between catchment-scale land cover disturbances and stream fish communities have been documented for agricultural (Roth et al. 1996; Lammert and Allan 1999), silvicultural (Davies and Nelson 1994; Stevens and Cummins 1999), and urban (Wang et al. 1997, 2001; Walters et al. 2003) land uses (386).
- (Roy et al., 2007) Fish assemblages were correlated with urban, forest, and agriculture land cover variables, with the greatest number of strong relations with % forest and % urban in the catchment (eight strong models), and % forest and % agriculture in the 1-km riparian network (four strong models; Table 4). Cosmopolitan and lentic tolerant species were the only groups correlated with agriculture, with increased richness and abundance associated with agriculture at some spatial extents. For all except cosmopolitan species, the strongest relationships were with the largest spatial extents of land cover (catchment), followed by riparian land cover in the 1-km and 200-m reach, respectively. Endemic richness, endemic:cosmopolitan richness and abundance, insectivorous cyprinid richness and abundance, and fluvial specialist richness were all negatively correlated with % urban cover and positively correlated with % forest cover in the catchment (Table 4) (391-392).
- (Gregory et al., 2002) Projections of geomorphic and hydrologic changes are not simple and will vary greatly based on local landscapes and climate. Ecological interactions are complex because of the interactions between adjacent terrestrial and aquatic ecosystems, predator–prey interactions, competition, succession, and dispersal of aquatic and terrestrial organisms. Even more complex is the array of social actions in river systems that dictate ecological responses, such as hydrologic alteration, water diversion, bank hardening, land use conversion, exotic species introductions, and water quality impairment (721).
- (Nelson and Booth 2002) Construction, agriculture, mining, and timber harvesting accelerate natural erosion rates, increasing the supply of sediment to surface water (51).
- (Nelson and Booth 2002) Wolman and Schick (1967) found that construction activity in once-forested watersheds can increase sediment yield up to several orders of magnitude. Trimble (1997) examined the role of channel-bank erosion in sediment yield from the 228-km2 San Diego Creek watershed, an urbanizing watershed in southern California. In that study, sediment production from channel enlargement accounted for approximately two-thirds of the measured suspended sediment yield and downstream sediment accumulation (54).
- (Nelson and Booth 2002) Although landslides and forest processes contribute most of the sediment in this budget, construction and other land-clearing activities yield the most sediment on a unit-area basis (Fig. 4) (62).
- (Nelson and Booth 2002) The overall estimated current sediment production in the watershed is 44 tonnes km22 yr21, compared to a pre-development sediment production of 24 tonnes km22 yr21. Consequently, urbanization has increased watershed-wide sediment production, primarily through channel erosion resulting from increased

discharges, and this process now accounts for approximately 20% of the total watershed sediment budget. Other urban elements, including construction, roadsurface erosion, and sediment production from residential and commercial areas, contribute an additional 12% to the total sediment production. Construction practices have been documented to be a major sediment contributor in urbanizing watersheds (Wolman and Schick, 1967); however, there is very little land being developed (0.3% yr21) relative to the size of the watershed, and so construction contributes relatively little (,1%) at present. Although the urban areas of the watershed directly generate relatively little sediment, much of the sediment produced by channel-bank erosion can be attributed to urbanization (63-64).

- (Nelson and Booth 2002) In either case our overall conclusion remains unchanged—this source is a very significant component of the total yield, and it may account for as much as 1/3 of the total under existing urbanizing conditions (64-65).
- (Nelson and Booth 2002) As with our study, most sediment budgets conducted in forested areas have also found landslides to be the greatest contributor of sediment to overall watershed sediment production (Dietrich and Dunne, 1978; Slaymaker, 1993; Paulson, 1997). Gravel roadsurface erosion, however, can also be an important sediment contributor (Reid, 1981; Madej, 1982; Paulson, 1997). Trimble's (1997) study of the San Diego Creek watershed was the only study reviewed here that quantified channel-bank erosion resulting from urbanization. The San Diego Creek watershed is rapidly urbanizing and was approximately 50% urban at the time of Trimble's study. Other land uses in that watershed were agriculture and undeveloped property. Trimble found that approximately 67% of total sediment production from the San Diego watershed came from channel-bank erosion, consistent with the 20% contribution found in this study for a watershed that is 19% urbanized (66).
- (Nelson and Booth 2002) In contrast, roadsurface erosion in forested areas can be directly attributed to forest practices (Reid and Dunne, 1984). Maintenance of active roads, and closure and revegetation of roads that are no longer used, would likely have a significant impact on road sediment production and at least a modest effect on overall watershed sediment yield. Probably the greatest opportunity to limit sediment production in the Issaquah Creek watershed, and in other urbanizing basins as well, is by reducing channel-bank erosion resulting from increased stormwater discharges. As urbanization continues, better efforts should be made to minimize increases in discharge to the channel network (66).
- (Kohler et al., 2004) Urban input was the main source of N-NO3/NO2 and N-NH3 (Table 2) into created wetland. Concentrations of N-NO3/NO2 and N-NH3 at UI were higher during storm events than nonstorm events, similar to the findings of Kao and Wu (2001) (Tables 2 and 3) (290).
- (Lewis et al., 2007) Concentrations of most major anions and cations (especially nitrate, sulfate, chloride, sodium, potassium, and calcium) were highest in the urban headwaters and decreased downstream. Generally, the highest concentrations of suspended coliform bacteria occurred in the urban headwaters. In contrast, stream habitat quality and the abundance, species richness, and species diversity of fishes did not differ significantly between urban and rural sites (303).
- (Lewis et al., 2007) The expansion of urban land areas affects the chemistry and biology of streams in important ways (Paul & Meyer, 2001; Walsh et al., 2005). Many of the deleterious effects of urban growth on streams stem from the alteration of land cover and runoff patterns caused by urbanization within the stream's drainage basin. For example, streams are sensitive to increased impervious surface cover (ISC) in their drainage basins (e.g., parking lots, roads, and buildings). As ISC increases in urban areas, the infiltration of precipitation into soils decreases, resulting in peak stream discharges that are higher, occur earlier, and have shorter durations than in areas with lower ISC (Paul & Meyer, 2001; Walsh et al., 2005). Marked changes in stream solute concentrations and bacterial abundance may also be associated with urban storm runoff (e.g., McConnell, 1980). Effects of urban land cover on stream chemistry and biology are evident under baseflow conditions, as well.

Compared to streams in rural areas, urban streams often have elevated concentrations of solutes, such as nitrate, sulfate, phosphate, chloride, and base cations, even under baseflow conditions (Hoare, 1984; Smart, Jones, & Sebaugh, 1985; Wahl, McKellar, & Williams, 1997; Wernick, Cook, & Schreier, 1998; Williams, Hopkinson, Rastetter, Vallino, & Claessens, 2005). There are many causes of these elevated concentrations. For example, industrial and wastewater treatment plant (WWTP) point sources add significant amounts of solutes, especially nitrate, sulfate, phosphate, chloride, and sodium, to streams (e.g., Andersen, Lewis, & Sargent, 2004; Cameron, Hall, Veizer, & Krouse, 1995; Ceasar et al., 1976; Flintrop et al., 1996; House & Denison, 1997; Jarvie, Whitton, & Neal, 1998; Roy, Gaillardet, & Allegre, 1999). In some urban residential areas, septic tanks and lawn fertilizers may increase phosphate or nitrate concentrations in streams (e.g., La Valle, 1975; Hoare, 1984; Wernick et al., 1998). Road salt runoff may contribute sodium and chloride to urban streams in colder climates (e.g., Ceasar et al., 1976; Douglas, Chamberlain, & Blum, 2002; Williams et al., 2005). Elevated dry deposition (including dust, nitrogen oxides, and sulfur oxides) in urban areas may enhance elemental fluxes to soils both directly and indirectly (Chiwa, Kim, & Sakugawa, 2003; Lovett et al., 2000). Finally, the lower biomass of vegetation in urban areas may reduce nutrient retention in urban watersheds and reduce the supply of dissolved organic carbon to urban streams (Wahl et al., 1997).

The physical and chemical changes associated with urban areas in turn can have important effects on stream organisms and communities. Negative effects of urbanization include increased sedimentation and habitat instability, reduced habitat complexity, inputs of various toxins (e.g., oils, heavy metals, pesticides), and increased fluctuations in water temperature associated with removal of riparian vegetation (Paul & Meyer, 2001). As a result of these changes, diversity of sensitive stream organisms, such as many invertebrates and fish, declines in urban streams (Paul & Meyer, 2001). Increased nutrient loading to streams could stimulate algal (and therefore secondary) productivity, although this response depends in part on other factors such as shading by riparian vegetation, turbidity, stability of streambed sediments, and pollutant levels (Paul & Meyer, 2001). Finally, bacterial abundances, especially of coliform bacteria, can be high in urban streams, possibly due to leaking or overflowing sanitary sewers (Duda, Lenat, & Penrose, 1982), runoff from impervious surfaces and lawns (Young & Thackston, 1999), or discharges from storm sewer drains (Frenzel & Couvillion, 2002). Although often peaking during storm events, bacterial abundances also may be high under baseflow conditions (Duda et al., 1982; Young & Thackston, 1999) (303-304).

- (Lewis et al., 2007) Conductivity and concentrations of Na^+, Cl^-, TDN, NO_3^-, K^+, SO_4^{2-}, Ca^{2+}, and Mg^{2+} were highest in the urban headwaters and decreased downstream (Figs. 2 and 4). With the exception of Mg^{2+}, mean values were significantly higher at urban than rural sites on at least three sample dates (Figs. 2 and 4; Table 1). Nitrate concentrations differed the most between urban and rural sites. The mean NO_3^- concentration for urban sites was as high as 2.6 times the mean concentration for rural sites (Table 1). Mean concentrations of dissolved O_2, Si^{4+}, Mg^{2+}, and HCO_3^- tended to be higher at urban than rural sites, but the differences were significant on only one or two dates (Figs. 2, 5; Table 1). Mean DOC concentrations and turbidity tended to be lower at urban than rural sites, but the differences were significant on only one or two dates (Fig. 5; Table 1). For $H_2PO_4^-$, pH, DON, PCO_2 saturation, and TDC, there were no significant differences between means for urban and rural sites on any date (Figs. 2, 4, and 5) (310).
- (Lewis et al., 2007) Mean concentrations of total coliforms were up to five times higher and mean concentrations of E. coli were up to six times higher at urban than rural sites (Table 3). Although this trend was consistent over time, there was considerable temporal variation in concentrations at some sites (Fig. 7), and concentrations at urban and rural sites did not differ significantly in three of the eight comparisons (Table 3). Mean concentrations of total heterotrophic bacteria did not differ significantly between urban and rural sites on any date ($p>0.20$). There also were no significant differences in bacterial concentrations upstream and downstream of the WWTP ($p>0.19$) (312).

- (Lewis et al., 2007) Fish abundance did not differ significantly (p=0.98) between rural and urban sites (rural mean=48.7, SE=16.5; urban mean=49.7, SE=21.2). Compared to urban sites, rural sites had slightly higher species richness (rural mean=7.5, SE=2.1; urban mean=5.7, SE=1.3), Simpson's diversity (rural mean=4.0, SE=1.1; urban mean=2.8, SE=0.5), and Shannon–Weiner diversity (rural mean=0.65, SE=0.14; urban mean=0.52, SE=0.09), although these differences were not statistically significant (p>0.34). A total of 14 species were found among all rural sites, whereas nine species were found among the urban sites (313).
- (Lewis et al., 2007) For example, in some urban areas, septic systems may contribute to elevated stream phosphate (La Valle, 1975) or nitrate (Hoare, 1984; Wernick et al., 1998) concentrations (314).
- (Lewis et al., 2007) Other studies have proposed that leaking sewer lines affect stream chemistry (Duda et al., 1982; Paul & Meyer, 2001) (314).
- (Lewis et al., 2007) Lawn fertilizers are another possible urban source of stream solutes (La Valle, 1975; Paul & Meyer, 2001) (315).
- (Lewis et al., 2007) The major sources of solutes to the urban headwaters in our study are unclear. The bedrock and saprolite in this region are unlikely to supply appreciable quantities of Cl^-, SO_{2_4} , or $NO_{\_3}$. One important source of solutes could be atmospheric dry deposition. Recent studies have demonstrated that dry deposition of $NO_{\_3}$, $NH_{þ4}$, SO_{2_4} , Cl^-, Ca^{2+}, Mg^{2+}, and Na^+ is higher in and near cities than in rural areas (Chiwa et al., 2003; Lovett et al., 2000). This suite of solutes closely matches the suite that is elevated in the urban headwaters of Big Brushy Creek. In addition, $NH_{þ4}$ from dry deposition could be nitrified in soils and thus contribute to the elevated $NO_{\_3}$ in urban streams. Many of the same solutes ($NO_{\_3}$, SO_{2_4} , Cl^-, K^+, Na^+, Ca^{2+}, and Mg^{2+}) were elevated in urban streams unaffected by agriculture or point sources of wastewater on the Missouri Ozark Plateau (Smart et al., 1985).

  Although stream solute concentrations differed significantly between urban and rural sites, most of the decline in solute concentrations that occurred downstream of the uppermost sampling sites in Big Brushy Creek (Figs. 2, 4, and 5) occurred within areas of urban land cover (315).
- (Lewis et al., 2007) In both our study and in the study by Smart et al. (1985), $NO_{\_3}$ concentrations differed more than did other solutes between urban and rural land cover. In our study, mean $NO_{\_3}$ concentrations at urban sites were up to 2.6 times higher than mean concentrations at rural sites. Furthermore, mean $NO_{\_3}$ concentrations at the upper urban headwater sites were five to six times higher than mean concentrations for some rural sites downstream (excluding those downstream of the WWTP). This pattern may reflect larger supplies of nitrogen than other elements in dry deposition. For example, Lovett et al. (2000) note that urban dry deposition was more enriched in $NO_{\_3}$ than in Ca^{2+}, suggesting that both dust and nitrous oxide gases were sources of $NO_{\_3}$. In addition, incision of urban stream channels may reduce the capacity for $NO_{\_3}$ removal or retention in riparian zones (Groffman et al., 2002) (316).
- (Lewis et al., 2007) Similarly, concentrations of Na^+, K^+, Ca^{2+}, Cl^-, and SO_{2_4} in the urban headwaters (sites 18, 19, 20, 21) were up to three to four times higher than concentrations in the two rural tributaries. Although there were no completely forested subbasins in the Big Brushy Creek watershed, $NO_{\_3}$ and SO_{2_4} concentrations (<0.45 and 1–2 mg/l, respectively) in mostly forested watersheds further upstream in the Saluda River basin and in the nearby Enoree River basin (Lewis, Garrett, Andersen, & Sargent, 2003) were well below concentrations in the urban headwaters of Big Brushy Creek (Figs. 2 and 4) (316).
- (Lewis et al., 2007) Also, during all four sampling intervals in June and July, NO_3–N accounted for a significantly greater percentage of TDN at urban than rural sites (Table 4). This observation supports the hypothesis that $NO_{\_3}$ becomes an increasingly important form of dissolved N in streams and rivers as human activity increases in watersheds (e.g., Hedin, Armesto, & Johnson, 1995) (316).

- (Lewis et al., 2007) Concentrations of coliform bacteria (including E. coli) tended to be higher in urban than rural portions of the Big Brushy Creek watershed. Similar results have been found by studies in other regions. For example, Frenzel and Couvillion (2002) found that concentrations of E. coli, fecal coliforms, and enterococci were higher in stream sub-basins with higher human population densities in and near Anchorage, Alaska. Similarly, Mallin, Williams, Esham, and Lowe (2000) found that fecal coliform abundance correlated positively with human population density, percentage of developed land, and percentage of impervious surface in five tidal creek basins in North Carolina (318).
- (Lewis et al., 2007) Specifically, mean concentrations of total coliforms were significantly greater at urban than rural sites for all sampling periods (318).
- (Lewis et al., 2007) However, even if we restrict our comparison of the urban headwaters to only the two rural tributaries (sites 11, 12A), differences between urban and rural areas still are evident: concentrations of E. coli and, with the exception of one sample from urban site 19, total coliforms were in all cases higher in the urban headwaters than in either rural tributary (318).
- (Lewis et al., 2007) Nonetheless, the highest concentrations of coliforms measured in our study (>60,000 cells/100 ml) occurred in the urban headwaters. A number of studies have provided evidence that pets (dogs and cats) are the primary sources of enteric bacteria in urban runoff and streams (Kelsey, Porter, Scott, Neet, & White, 2004; Mallin et al., 2000; Young & Thackston, 1999). The urban headwaters of Big Brushy Creek flow through residential areas where pet feces could be sources of coliform bacteria. In fact, dogs were observed near houses in those areas (318).
- (Lewis et al., 2007) In urban areas, the combination of extensive impervious surface cover and high density of storm drains promote the rapid flushing of bacteria from lawns, roads, and other surfaces into streams (Frenzel & Couvillion, 2002; Mallin et al., 2000). However, we have documented high concentrations of E. coli and total coliforms under baseflow, rather than storm flow, conditions. Therefore, there appear to be sources of bacteria to Big Brushy Creek besides storm runoff (318-319).
- (Lewis et al., 2007) We found no statistically significant differences in fish abundance, species richness, or diversity between urban and rural sites within the Big Brushy Creek watershed. This contrasts with other studies in which urbanization appears to negatively affect fish abundance, richness, and/or diversity in the southeastern United States (Lenat & Crawford, 1994; Long & Schorr, 2005; Weaver & Garman, 1994). Given our small sample sizes, it is possible that we failed to detect real effects of urban land cover on fish. However, mean species richness and diversity both were slightly higher at rural sites. Also, more species were found only at rural (seven species) than at urban sites (two species). In addition to effects of urban land cover, smaller stream size also might account for lower abundance and diversity of fishes in the urban headwaters. Headwater streams naturally tend to have lower fish abundance and diversity than larger rivers, presumably because of factors such as lower habitat diversity and availability, as well as more variable flow and habitat conditions in the headwaters (Horwitz, 1978; Schleiger, 2000) (319).
- (Lewis et al., 2007) The other six species found only at rural sites can inhabit smaller headwater streams of suitable habitat and water quality (e.g., Jenkins & Burkhead, 1994; Rohde, Arndt, Lindquist, & Parnell, 1994). The absence of those species from the urban sites is thus consistent with the generalization that a shift from sensitive to more tolerant species occurs during urbanization (Walsh et al. 2005). Further, one of the two species that was found only at an urban site, S. atromaculatus, is considered to be a pollution-tolerant species (Schleiger, 2000) (319).
- (Lewis et al., 2007) Urban land cover may affect fish in ways other than reducing abundance or diversity. For example, in watersheds of the lower piedmont in Georgia, fish species richness and diversity did not correlate significantly with percent urban land cover (Helms, Feminella, & Pan, 2005). However, with increasing percent urban land cover, herbivorous species became relatively more abundant and lithophilic spawners

became relatively less abundant. Also, tolerant centrarchids tended to be more common in urban watersheds. In addition, fish health tended to be lower in more urbanized watersheds (320).

- (Lewis et al., 2007) Our study of a stream drainage with urban headwaters receiving no agricultural inputs demonstrates that urban land cover influences stream chemistry and microbiology. Concentrations of major ions and suspended coliform bacteria generally were higher in the urban headwaters than in downstream rural areas (320).
- (DeLuca et al., 2004) Small and Hunter (1988) found that roads, power lines, and edges, all characteristics of developed areas, provide pathways for potential predators to enter undisturbed habitat and depredate bird nests. Pathways or corridors such as these may act in similar ways near marshes to increase nest predation and lower reproductive success. Another explanation for the reduction in marsh bird community integrity may be the transfer of pollutants from adjacent land. Chemical pollutants and nutrients transferred from developed areas through point sources may reduce the food resources of marsh birds (Poulin et al. 2002). For example, aquatic macroinvertebrates might be impacted from such point source pollution. Interestingly, most secretive marsh birds, marsh foraging specialists, feed primarily on aquatic macroinvertebrates, and only three of the 45 lowest scoring wetland sites of our study had secretive marsh birds present (844).
- (Pringle 2001) However, their lower watersheds have largely been cleared for agriculture or urbanization. Deforestation has resulted in greater runoff, decreased infiltration rate and aquifer recharge, and increased erosion and sedimentation in rivers (Pringle and Scatena 1999) (992-993).
- (Nelson and Booth 2002) The sediment production from upland sources and in-stream erosion in the Issaquah Creek watershed is approximately 6400 tonnes yr21 (Table 4). Since the watershed is 73% forested, forest processes (landslides, soil creep and forest road erosion) understandably contribute the greatest volume of sediment, with the bulk coming from landslides (3264 tonnes). The next most voluminous sediment sources are channel-bank erosion (1299 tonnes), urban land uses (382 tonnes), and urban road-surface erosion (268 tonnes) (Table 4). Normalized by land area, the total watershed yield is 44 tonnes km22 yr21, compared to an estimated predevelopment rate of 24 tonnes km22 yr21 (61).
- (Groffman et al., 2003) However, we suggest that, over time, urban watersheds move towards stable land use, with large amounts of impervious cover and low sediment production leading to stream incision in most locations (317).
- (Groffman et al., 2003) Stream incision, in combination with reduced infiltration in impervious urban uplands, can reduce riparian groundwater levels (Figures 3 and 4), which can have dramatic effects on soil, plants, and microbial processes. As discussed below, water table level is critical in the control of riparian ecosystem structure and function. It influences soil type, for example the presence of wetland or hydric soils (ie wet, with high levels of organic matter), plant communities (wetland and upland/wetland transition plants) (Gold *et al*. 2001), and the unique fauna (eg salamanders) that depend on the presence of specific soils and plants (Bodie 2001; Groom and Grubb 2002) (317).
- (Gergel et al., 2002) Landscape indicators that quantify the amount of and distance to land converted to human uses often explain variability in water chemistry parameters among catchments. For example, the amount of urban land cover and its distance from the stream were the most important variables in predicting N and P concentrations in stream water (Osborne and Wiley, 1988) (120).
- (Gage et al., 2004) This analysis indicates that the percentage of developed land in the watershed is one factor that affects macroinvertebrate abundance. The trend of decreasing abundance of insects in irregular disturbance streams over time (Figure 2a) indicates that other factors also affect abundance. Low disturbance streams always had significantly more EPT individuals than both irregular and high disturbance streams (Fig. 2b) (352).

- (Ekness and Randhir 2007) Species richness has been shown to increase as vegetative density increases and with distance from developed areas (1470).
- (Barker et al., 2006) Lammert and Allan (1999) found that fish IBI showed a strong relationship to flow variability and immediate land use, while the benthic IBI correlated most strongly with dominant substrate in an agricultural watershed in Michigan. A local Chesapeake Bay watershed found similar results. This study assessed stream ecological conditions using Maryland Department of Natural Resources Biological Stream Survey (MBSS) data from the Patapsco river basin, which is a largely urban area. Fish Index of Biotic Integrity (IBI) was found to be significantly related to urban land use, with more variation explained as the scale of focus was increased from local to riparian to catchment. In contrast, benthic IBI was found to be strongly affected by local conditions, including riparian buffer width (Southerland *et al.*, 2002) (3).
- (Barker et al., 2006) The regression model using all sites identified the variable "percent high urban land use" as a primary explanatory variable, which was consistent with other published studies that used the entire MBSS data set (Roth *et al.*, 1998; Boward *et al.*, 1999) (9-11).
- (Gergel et al., 2002) For example, in a study of fish in Wisconsin streams, the health of fish communities was negatively correlated with the amount of upstream urban development (Wang et al., 1997). The health of fish communities was also positively correlated with amount of upstream forest and negatively correlated with amount of agriculture. This relationship exhibited a nonlinear, threshold response; declines in condition of the fish fauna occurred only after ~20% of the catchment was urbanized, and no impacts were attributed to agriculture until it occupied ~50% of the catchment (Wang et al., 1997) (120-121).
- (Gergel et al., 2002) Freshwaters are degraded by increasing inputs of silt, nutrients and pollutants from agriculture, forest harvest, and urban areas (Carpenter et al., 1998) (125).
- (Voelz et al., 2005) The analyses of the macroinvertebrate metrics indicate that urban impacts occur in both the Big Thompson and Cache la Poudre Rivers (195).
- (Voelz et al., 2005) Our data support significant effects of the urban areas on macroinvertebrates with either spring or autumn data for the Cache la Poudre River (195).
- (Voelz et al., 2005) The overall message from our analysis of the macroinvertebrate metrics is that differences exist between the reference condition and the sites in and below the respective urban areas. In all cases these differences indicate worse conditions at the downstream sites. However, analysis of longer-term data suggests no worsening trends or somewhat better conditions within sites at least in the Cache la Poudre River, even with increased urban development (also see Shieh *et al.*, 1999) (196).
- (Voelz et al., 2005) Using the surrogate variables of potential urban impact (population and housing units), and the environmental gradient represented primarily by chemical factors, it is evident that there is an effect of urban land use that is reflected in the macroinvertebrate assemblages in both the Big Thompson and Cache la Poudre Rivers. The results from the environmental analyses suggest some reasons for the observed impacts on the macroinvertebrate assemblages. Although fewor no violations ofwater quality standards were observed in either river, increased concentrations of many chemical variables, especially nutrients, with distance downstream from the urban areas may more indirectly affect the organisms (197).
- (Voelz et al., 2005) For example, sediments that enter rivers from unstable stream banks or construction can directly affect the biota through abrasion, and indirectly by smothering habitat, and carry bound contaminants downstream. Sediment is the main pollutant in U.S. rivers (US EPA, 1998) and a major component of urban runoff (Novotny and Olem, 1994) (197).
- (Voelz et al., 2005) Increased levels of plant nutrients are also common in urban runoff and can cause changes in the periphyton assemblage structure, even at low concentrations (Peterson *et al.*, 1985; Hart and Robinson, 1990). This change in

periphyton may result in effects on macroinvertebrates in at least two ways (also see Dodds and Welch, 2000). First, shifts in algal resources can negatively affect the food base for some organisms such as scrapers. Second, some benthic invertebrates are influenced by changes in algal assemblages because a change in habitat structure occurs (see Dudley *et al.*, 1986). As for sediment, shifts in algal resources and/or structure may differentially affect individual taxa (198).

- (Voelz et al., 2005) The increased level of plant nutrients (NH4–N, NO3–N and PO4–P) found in this study, as the rivers flowed through and exited the urban areas, has been observed in other research (e.g., Osborne andWiley, 1988; Tate and Heiny, 1995) (198).
- (Rhodes et al., 2001) Throughout the watershed, average concentrations of NO3- and $SO4^{2-}$ correlate with percent catchment area classified for human use (agricultural, residential, commercial, industrial, urban open, and transportation areas; (Figure 3). A linear, best-fit line demonstrates a positive correlation between NO3- (and $SO4^{2-}$) and percent human land use at *R* 2) 0.68 (and 0.69). Standard deviation from average values also increases with greater percent human land-use area. Similar toNO3- and $SO4^{2-}$, Cl- concentrations increase with greater road density with a positive linear correlation of *R* 2) 0.87 and higher standard deviation from mean values with higher road density. These results are similar to those reported for first-order catchments of the Ipswich River, Massachusetts (*26*), where a positive relationship exists between $SO4^{2-}$, NO3-, Cl-, and Na+ with percent urban area. Gubrek (*7*) also devised an algorithm to predict NO3- from percent forest and agricultural land uses in a 730 ha watershed. Similarly in this study, the positive correlation in Figure 3 can predict how NO3-, $SO4^{2-}$, and Cl- concentrations will increase with future development within the subcatchments of MRW.
- (Paul and Meyer 2001) The major impact of urbanization on basin morphometry is an alteration of drainage density, which is a measure of stream length per catchment area (km/km2). Natural channel densities decrease dramatically in urban catchments as small streams are filled in, paved over, or placed in culverts (Dunne & Leopold 1978, Hirsch et al. 1990, Meyer & Wallace 2001). However, artificial channels (including road culverts) may actually increase overall drainage densities, leading to greater internal links or nodes that contribute to increased flood velocity (Graf 1977, Meyer & Wallace 2001) (338).
- (Paul and Meyer 2001) Urbanization affects both sediment supply and bankfull discharge. During the construction phase erosion of exposed soils increases catchment sediment yields by 102–104 over forested catchments and can be more exaggerated in steeply sloped catchments (Wolman 1967, Leopold 1968, Fusillo et al. 1977). Most of this export occurs during a few large, episodic floods (Wolman 1967). This increased sediment supply leads to an aggradation phase as sediments fill urban channels (Figure 2). During this phase stream depths may decrease as sediment fills the channel, and the decreased channel capacity leads to greater flooding and overbank sediment deposition, raising bank heights (Wolman 1967). Therefore, overall channel cross-sections stay the same or even decrease slightly (Robinson 1976) (338-339).
- (Paul and Meyer 2001) High ISC associated with urbanization increases the frequency of bankfull floods, frequently by an order of magnitude or, conversely, increases the volume of the bankfull flood (Leopold 1973, Dunne & Leopold 1978, Arnold et al. 1982, Booth & Jackson 1997). As a result, increased flows begin eroding the channel and a general deepening and widening of the channel (channel incision) occurs to accommodate the increased bankfull discharge (Hammer 1972, Douglas 1974, Roberts 1989, Booth 1990) (339-340).
- (Paul and Meyer 2001) However, erosion more commonly occurs disproportionately to discharge changes, often leading to bank failure and catastrophic erosion in urban streams (Neller 1988, Booth 1990). In developed urban catchments, as a result of this erosional readjustment phase, the majority of sediment leaving the catchment comes from within channel erosion as opposed to hillslope erosion (Trimble 1997). The magnitude of this generalized geomorphic response will vary longitudinally along a stream network as well as with the age of development, catchment slope, geology, sediment characteristics, type of urbanization, and land use history (Gregory et al. 1992) (340).

- (Paul and Meyer 2001) Changes in sediment supply may also alter channel pattern. Increased sediment supply during construction has converted some meandering streams to braided patterns or to straighter, more channelized patterns (Arnold et al. 1982). In the latter case, channelizing leads to increased slope and therefore higher in-stream velocities, especially where artificial channel alteration is carried out to increase the efficiency of the channel in transporting flows (Pizzuto et al. 2000).
  Urbanization can also alter sediment texture. Less fine sediment, increased coarse sand fractions, and decreased gravel classes have been observed in urban channels as a result of alteration of sediment supply and altered velocities (Finkenbine et al. 2000, Pizzuto et al. 2000). In addition to sediment changes, large woody debris is also reduced in urban channels. Catchments in Vancouver, British Columbia with greater than 20% ISC generally have very little large woody debris, a structural element important in both the geomorphology and ecology of Pacific Northwest stream ecosystems (Finkenbine et al. 2000).
  Other geomorphic changes of note in urban channels include erosion around bridges, which are generally more abundant as a result of increased road densities in urban channels (Douglas 1974). Bridges have both upstream and downstream effects, including plunge pools created belowbridge culverts that may serve as barriers to fish movement. Knickpoints are another common feature of urban channels. These readily erodeable points of sudden change in depth are created by channel erosion, dredging, or bridge construction and are transmitted throughout the catchment, causing channel destabilization (Neller 1988). Other features include increased tree collapse, hanging tributary junctions as a result of variable incision rates, and erosion around artificial structures (e.g., utility support pilings) (Roberts 1989).
  Changes in the hydrology and geomorphology of streams likely affect the hydraulic environment of streams, altering, among other things, the velocity profiles and hyporheic/parafluvial dynamics of channels. Such changes would affect many ecological processes, from filter-feeding organisms (Hart&Finelli 1999) to carbon processing and nutrient cycling (Jones & Mulholland 2000) (340-341).
- (Paul and Meyer 2001) Chemical effects of urbanization are far more variable than hydrologic or geomorphic effects and depend on the extent and type of urbanization (residential versus commercial/industrial), presence of wastewater treatment plant (WWTP) effluent and/or combined sewer overflows (CSOs), and the extent of stormwater drainage (341).
- (Paul and Meyer 2001) In general, there is an increase in almost all constituents, but consistently in oxygen demand, conductivity, suspended solids, ammonium, hydrocarbons, and metals, in urban streams (Porcella & Sorenson 1980, Lenat & Crawford 1994, Latimer & Quinn 1998, USGS 1999) (341).
- (Paul and Meyer 2001) In some cases increases in phosphorus [in urban catchments] can even rival those seen in agricultural catchments both in terms of concentration and yield (Omernik 1976). Even an attempt to understand the agricultural contribution to catchment phosphorus dynamics in a midwestern catchment discovered that urbanization was a dominant factor (Osborne & Wiley 1988). Even though urban areas constituted only 5% of the catchment area and contributed only a small part to the total annual yield of dissolved phosphorus, urban land use controlled dissolved phosphorus concentration throughout the year (342).
- (Paul and Meyer 2001) As with phosphorus, nitrogen concentrations in streams draining agricultural catchments are usually much higher (USGS 1999), but some have noticed similar or even greater levels of nitrogen loading from urbanization (Omernik 1976, Nagumo & Hatano 2000) (343).
- (Paul and Meyer 2001) Another common feature of urban streams is elevated water column and sediment metal concentrations (Bryan 1974, Wilber & Hunter 1977, Neal et al. 1997, Horowitz et al. 1999, Neal & Robson 2000) (343).

- (Paul and Meyer 2001) Pesticide detection frequency is high in urban streams and at concentrations frequently exceeding guidelines for the protection of aquatic biota (USGS 1999, Hoffman et al. 2000) (344).
- (Paul and Meyer 2001) Most surprising is that many organochlorine pesticide concentrations in urban sediments and biota frequently exceed those observed in intensive agricultural areas in the United States (USGS 1999), a phenomenon observed in France as well (Chevreuiel et al. 1999). Additionally, it is estimated that the mass of insecticides contributed by urban areas is similar to that from agricultural areas in the United States (Hoffman et al. 2000) (344-345).
- (Paul and Meyer 2001) As with metals, the main vector of transport of pesticides into urban streams appears to be through NPS runoff rather than WWTP effluent (Foster et al. 2000). A strong correlation between particle concentration and pesticide concentration was found in the Anacostia River basin in Maryland and the San Joaquin River in California, suggesting NPS inputs are most important (Pereira et al. 1996, Foster et al. 2000) (345).
- (Paul and Meyer 2001) Bacterial densities are usually higher in urban streams, especially after storms (Porcella & Sorenson 1980, Duda et al. 1982). Much of this is attributable to increased coliform bacteria, especially in catchments with wastewater treatment plant (WWTP) and combined sewer overflow (CSO) effluent (Gibson et al. 1998, Young & Thackston 1999). In Saw Mill Run, an urban stream near Pittsburgh, Pennsylvania, fecal coliform colony–forming units (CFU) increased from 170–13,300 CFU/100 ml during dry weather to 6,100–127,000 CFU/100 ml during wet weather (Gibson et al. 1998). CSOs contributed 3,000–85,000 CFU/100 ml during wet weather. These data indicate that non–point sources (NPSs) as well as point sources contribute to fecal coliform loads in urban streams. High values during dry weather are not uncommon in urban streams and may indicate chronic sewer leakage or illicit discharges. Storm sewers were also a significant source of coliform bacteria in Vancouver, British Columbia; stormwater there contained both human and nonhuman fecal coliform bacteria (Nix et al. 1994). Other pathogens, including *Cryptosporidum* and *Giardia*, have also been associated with CSOs (Gibson et al. 1998) (347).
- (Paul and Meyer 2001) Extirpation of fish species is not uncommon in urban river systems (Ragan & Dietmann 1976, Weaver & Garman 1994, Wolter et al. 2000). Comparative catchment studies also find dramatic declines in fish diversity and abundances in urban catchments compared with forested references (Scott et al. 1986, Weaver & Garman 1994, Lenat & Crawford 1994) (351-352).
- (Paul and Meyer 2001) Benthic feeders quickly reappeared as sedimentation rates declined after construction. Flow modification associated with urbanization also affects stream fish. In the Seine, modification of flow for flood protection and water availability has affected pike (*Esox lucius*) by reducing the number of flows providing suitable spawning habitat. With urbanization, the river contains enough suitable spawning habitat in only 1 out of 5 years as opposed to 1 out of every 2 years historically (Boet et al. 1999) (352-353).
- (Paul and Meyer 2001) Introduced fish species are also a common feature of urban streams. As a result of channelization, other river transportation modifications, and voluntary fisheries efforts in the Seine around Paris, 19 exotic species have been introduced, while 7 of 27 native species have been extirpated (Boet et al. 1999). The red shiner (*Cyprinella lutrensis*), a Mississippi drainage species commonly used as a bait fish, has invaded urban tributaries of the Chattahoochee River in Atlanta, Georgia where it has displaced native species and now comprises up to 90% of the fish community (DeVivo 1995) (353).
- (Paul and Meyer 2001) Uptake lengths in these rivers are much longer than in nonurban rivers of similar size, suggesting that not only is nutrient loading elevated in urban streams, but also nutrient removal efficiency is greatly reduced. The net result of these alterations in urban streams is increased nutrient loading to downstream lakes, reservoirs, and estuaries (355).

- (Gage et al., 2004) Increasing development north of Charlotte, NC, threatens aquatic life in streams by reducing riparian zones and increasing runoff (345).
- (Gage et al., 2004) Changing land use patterns may affect several water chemistry parameters (Prowse 1987), and changes in water chemistry may negatively affect stream insect biota, resulting in decreased stream biodiversity (Lillie and Isenring 1996, Winter and Duthie 1998). Our results indicate that alkalinity and dissolved oxygen change in streams subject to increased development or disturbance (353).
- (Russell et al., 1997) Areas with medium or high wetness indices that also had bare/herbaceous, scrub, or agricultural cover classes were regarded as potential sites for riparian wetland restoration. The urban class was regarded as ineligible for restoration, because of the probable high costs associated with altering this land use (64).
- (Paul and Meyer 2001) The Acheres (Seine Aval) treatment plant, which serves 8.1 million people, discharges 75 km west of Paris and releases 25,000 liters/s during low flow periods (Horowitz et al. 1999), increasing baseflow discharge in the Seine by up to 40% during low flow periods. More strikingly, wastewater effluent constitutes 69% annually and at times 100% of discharge in the South Platte River below Denver, Colorado (Dennehy et al. 1998). In our experience, high percentage contributions of wastewater discharge to urban rivers are not uncommon (336).
- (Paul and Meyer 2001) Some exceptions to these observations have been noticed, largely depending on the location of urbanization within a catchment. If the ISC occurs lower in a catchment, flooding from that portion can drain faster than stormflow from forested areas higher in the catchment, leading to lower overall peak flood discharge and increased flood duration (Hirsch et al. 1990). In addition, blocked culverts and drains, swales, etc. may also detain water and lower peak flood discharges (Hirsch et al. 1990) (335).
- (Paul and Meyer 2001) Lastly, sedimentation associated with urban streams reduces available refugial space, and invertebrates are more susceptible to drift when refugial space is limited during the frequent floods characteristic of urban environments (Borchardt & Statzner 1990) (350).
- (Roy et al., 2007) Endemic richness, endemic:cosmopolitan abundance, insectivorous cyprinid richness (after accounting for drainage area) and abundance, and fluvial specialist richness were all best predicted by % urban land cover (Table 5). In all cases, these single variable models were weighted at least twice as strong as competing models. Although these models explained a small percent variance in fish assemblages, they indicated that % forest in the 1-km or 200-m riparian area was not important in explaining these aspects of fish assemblages, regardless of the level of urbanization. Plots revealed wedge-shaped relationships, whereby the maximum richness or abundance of these groups of sensitive species declined with increased urbanization (Fig. 3) (392-393).
- (Roy et al., 2007) Urbanization and the concomitant declines in forest land cover throughout catchments result in hydrologic alteration, increased bank erosion and sedimentation, altered in-stream habitat, and increased delivery of pollutants to streams, among other impacts (Paul and Meyer 2001). These changes, in turn, alter biotic assemblages, resulting in the observed linkages between catchment land cover and fish assemblages in this and other studies (e.g., Wang et al. 1997, 2001; Scott and Helfman 2001; Walters et al. 2003). Studies that incorporate a range of catchment land cover often demonstrate significant relationships between land cover and stream quality (see Table 1). This study had the greatest differences in %forest and%urban land cover (vs smaller ranges in % agriculture) across sites, and these variables were most important in predicting aspects of fish assemblage integrity (394).
- (Allmendinger et al., 2007) When compared with watersheds in other parts of the U.S., the Good Hope Tributary watershed has produced a relatively large amount of sediment. The data in Table 4 suggest that sediment yield from the Good Hope Tributary over the last 45 years is almost eight times greater than the average for streams along the Atlantic coast. The data also suggest that the sediment yield from the Good Hope Tributary is

greater than mid-Atlantic Piedmont watersheds with forested land use and about equal to that of watersheds with rural land use. Suburban watersheds that are being actively developed without stormwater management yield much more sediment than the Good Hope Tributary. Wolman (1967) presented a frequently cited curve of annual sediment yield for mid-Atlantic watersheds that spans several periods of changing land use from the 1700s through the mid-20th Century. Wolman (1967) suggested that sediment yields should increase dramatically during periods of construction, reaching values exceeding 200 tons ⁄km2 ⁄ year in the absence of sediment management practices. Following development, sediment yields should become significantly lower, leveling off to slightly greater than 10 tons ⁄km2 ⁄ year. Allmendinger (1999, 2004) noted that the 45-year average sediment yield of 135 tons ⁄km2 ⁄ year obtained during this study represents an intermediate value between these two extremes. This is consistent with Wolman (1967) results, because the period between 1951 and 1996 included brief periods of extensive construction as well somewhat longer periods when relatively few areas were under construction (1495-1496).

- (Allmendinger et al., 2007) During this period, the percentage of the watershed covered with impervious surfaces increased from 1.7 to 7.5% and the channel area of the Good Hope Tributary increased by factor of 1.7. A sediment budget indicates that upland erosion and channel enlargement were significant sources of sediment in the watershed, each producing an amount of sediment equivalent to 70 and 80% of the total sediment yield. Floodplain sediment storage accounted for 50% of the total sediment yield, demonstrating that floodplains are an important component of the sediment budget of the study area during recent urban development. Solving a simple mass balance equation suggests that the sediment yield of the Good Hope Tributary is equivalent to 135.0 tons ⁄km2 ⁄ year (1496-1497).
- (Dodds and Oakes 2008) Our results were consistent with previous studies (Johnson and others 1997; Jones and others 2001; Osborne and Wiley 1988; Sliva and Williams 2001), suggesting that agricultural and/or urban lands were the most important predictors of water quality variability.

  Maintaining buffers or other passive land uses in headwater streams may effectively reduce diffuse pollution downstream. The importance of these streams and their riparian zones is due in part to their sheer numbers; small streams often comprise the majority of stream miles within a drainage network (Horton 1945; Leopold and others 1964), and in this study the smallest (first-order) streams on average comprised more than 60% of the stream miles in the study watersheds. Riparian land cover near the firstorder streams of watersheds explained greater variance in TN, NO3-, and TP concentrations than did riparian land cover immediately upstream from sampling sites. Firstorder riparian land cover was statistically related to most water quality measures, even when all potential correlation related to watershed land cover was controlled for. Our results suggest that headwater riparian areas could have an important impact on downstream water quality (375).
- (Han et al., 2008) Other non-native fish species commonly observed in Hokkaido belong mostly to cyprinids. Their occurrence was not associated with dams or reservoirs but with development and ⁄ or agriculture at relatively lower elevations in the Ishikari river, the largest river basin of Hokkaido. These non-native fish species (excluding common carp) have been introduced through unintentional or unauthorised stocking (e.g., accidental introduction of bitterling and gudgeon associated with intentional stocking of common carp) which prevents a complete invasion history (Hokkaido Government; http://bluelist.hokkaido-ies.go.jp). The Ishikari river is most densely populated and has the longest history of reclamation in Hokkaido (Ishigaki & Fukuda 1994). The proportion of riverbanks that are adjacent to an urban area is highest in this river among all rivers in Hokkaido (http://www.biodic.go.jp/reports2/4th/kasen.html). The distributions of nonnative fish species have been positively correlated with human population density and the proportion of developed area (Shea & Chesson 2002; Meador et al. 2003; McKinney 2006) (7).

- (Radwell and Kwak 2005) Our research revealed several insightful findings applicable to river ecology and management. First, we found that physical characteristics were more influential in ranking rivers in terms of ecological integrity, relative to biotic attributes. Among physical attributes, those at the watershed level, including land use, ownership, and road density, were the most influential components, playing a major role in discriminating among rivers. However, fish density, biomass, and occurrence of intolerant fishes were influential biotic factors, as well as invertebrate density and taxa richness (806).
- (Morgan and Cushman 2005) For both CP [Coastal Plain] (Table 3) and EP [Eastern Piedmont] (Table 4), fish richness and abundance in sites at the lowest urbanization level increased with increasing stream order. Richness in EP sites also decreased as catchment urbanization increased within each order (Table 4), whereas richness in CP sites did not (Table 3). Similar to richness, fish abundance increased at the lowest urbanization level as stream order increased in both ecoregions (Tables 3, 4); however, there was a general decline in abundance in EP sites within each order as catchment urbanization increased (Table 4) (647).
- (Morgan and Cushman 2005) [In Coastal Plain 1st-order streams] Mean fish species richness ranged from; 4 to 6 per site across all urbanization categories (Table 3). There were no significant differences in fish abundance or richness across all urbanization levels (Tables 3, 5). Abundance in highly urbanized sites was only slightly lower than the least-urbanized sites. Slightly higher fish abundances in 0–10% and (647) 10–25% than .50% urbanized sites resulted from increased presence of tolerant fish species and an overall reduction of species in other tolerance categories (Table 2).
- (Morgan and Cushman 2005) [In Coastal Plain 2nd-order streams]. Mean richness ranged from 11 to 12 species per site across all urbanization categories (Table 3). Abundance and richness did not significantly differ among urbanization levels (Tables 3, 5); however, high abundances of fish per site at the 2 highest urbanization levels (.330 fish per site; Table 3) was possibly associated with replacement of intolerant with tolerant species (generalists) as catchment urbanization increased.
- (Morgan and Cushman 2005) [In Coastal Plain 3rd-order streams] Mean richness ranged from 15 to 16 species in both urbanization categories (Table 3). Mean richness and abundance was higher in 3rd-order sites than in 1st- and 2nd-order sites (Table 3). Neither fish abundance nor rich (648) ness differed between the 2 urbanization levels for 3rd-order sites (Tables 3, 5).
- (Morgan and Cushman 2005) [In Eastern Piedmont 1st-order streams] Mean richness ranged from; 3 to 7 species per site across all urbanization categories (Table 4), and richness significantly differed among urbanization categories (Tables 4, 5). Richness was generally low in sites with .25% urbanization, rarely exceeding 5 species per site, whereas richness in .50% urbanized catchments was ,3 species per site. Fish abundance in sites from .25% urbanized catchments was significantly lower than in less-urbanized sites (Tables 4, 5). Mean abundance in the 0–25% urbanized sites was ;2.53 higher than .50% urbanized sites (Table 4).
- (Morgan and Cushman 2005) [In Eastern Piedmont 2nd-order streams] Mean richness ranged from ;5 to 12 species across all urbanization categories, with a progressive decrease from the least- (0–25%) to the most-urbanized (.50%) catchments (Table 4). Richness significantly differed between the least- and most-urbanized catchments (0–25% vs .50%, respectively; Tables 4, 5). Abundance did not differ among urbanization levels (Tables 4, 5); however, high abundance of fish in sites with .50% urbanization resulted from high numbers of tolerant blacknose dace (*Rhinichthys atratulus*). More than 1400 *R. atratulus* (98% of fish collected) were collected in 1 highly urbanized catchment (.75% urbanization), and .200 *R. atratulus* per site were found in 3 other catchments with .75% urbanization.
- (Morgan and Cushman 2005) [In Eastern Piedmont 3rd-order streams] Mean richness ranged from; 5 to 17 species across all urbanization categories (Table 4). Richness values were significantly different among urbanization categories (Tables 4, 5), and rarely

exceeded 5 species per site in .50% urbanized catchments. Richness differed significantly among urbanization categories, where 0–25% urbanized sites displayed fish richness .33 higher than in the .50% urbanized sites and 2.53 higher in the 25–50% urbanized sites (Tables 4, 5). Abundance also differed with degree of urbanization, being; 1.83 and .33 lower in the 25–50 and .50% urbanized sites, respectively, than in the 0–25% urbanized sites (Tables 4, 5).

- (Morgan and Cushman 2005) In CP [Coastal Plain] sites, FIBI was inversely correlated with catchment urbanization (*p* , 0.05; Fig. 2A), although fit to the regression line (not shown in figure) was extremely low (*r*2 5 0.035) because of high intersite variation. In contrast, EP [Eastern Piedmont] sites displayed a strong inverse relationship between FIBI and % catchment urbanization (*r*2 5 0.49, *p* , 0.0001, Fig. 2B). Using the breakpoint of 3.0 that separated ''poor'' from ''fair'' FIBI scores (Roth et al. 1999), we estimated that .20 and .29% catchment urbanization within CP and EP sites, respectively, could result in either a ''poor'' or ''very poor'' FIBI rating (649).
- (Morgan and Cushman 2005) For CP sites, there were significant differences between expected and observed species assemblages at all levels of urbanization (649).
- (Morgan and Cushman 2005) Interestingly, comparison of observed and expected richness values for CP [Coastal Plain] sites indicated that assemblages differed in richness and in composition (Tables 2, 3). Specifically, 2nd-order sites showed higher observed richness than expected (i.e., 11.5 vs 10 species per site, respectively; Tables 3, 6); however, species composition differed from the expected assemblage at these sites. These results contrasted with 1st- and 3rd-order sites, which showed lower or similar observed than expected richness, respectively (Tables 3, 6). For EP [Eastern Piedmont] sites, there were significant differences between expected and observed assemblages at .50% urbanization for 1st- and 3rd-order sites, and at $25–50% urbanization for 2nd-order sites (Table 7). Differences between expected and observed assemblages did not become nonsignificant for all urbanization levels until expected richness was artificially lowered to 3 for 1st-order sites (75% of the expected species assemblage), 5 for 2nd-order sites (50%), and 5 for 3rdorder sites (63%) (Table 7). Differences in fish assemblages for EP sites (Table 7) were usually observed at higher % catchment urbanization ($50%) than CP sites (Table 6) (649-651).
- (Morgan and Cushman 2005) Using the MBSS data set, we found that Maryland stream fish assemblages were associated with urban land use, with major assemblage differences generally occurring at .25% catchment urbanization. Yet, our analyses showed strikingly different patterns in the 2 ecoregions. Neither abundance nor species richness differed between streams in low- vs highly urbanized catchments in CP [Coastal Plain], whereas in EP [Eastern Piedmont] streams abundance, richness, and FIBI all decreased with increasing urbanization. Moreover, richness and abundance decreased in 1st-, 2nd-, and 3rd-order EP sites as catchment urbanization increased, except for elevated abundance of tolerant species in 2nd-order EP sites. We found no evidence for a similar trend in CP sites, where fish assemblage composition apparently shifted from the complex expected to one that was unresponsive to urbanization. The probable assemblage was derived from fish occurrences across the entire CP instead of just the western shore of the CP, but fish richness and abundance did not change as urbanization intensity increased. Furthermore, the expected assemblage in CP sites was dominated by more tolerant species than sites in the EP, even at low catchment urbanization.

  The significant negative relationship in the FIBI for EP sites but not CP sites with increasing urbanization was interesting because the FIBI was developed specifically for each ecoregion (Roth et al. 1999). However, our results suggest that components of the FIBI are useful in understanding potential fish response to urbanization in the EP but have limited application in the CP (651-652).
- (Morgan and Cushman 2005) Richness, abundance, and FIBI provided limited information about fish assemblage–urbanization relationships in CP [Coastal Plain] sites, but we found significant differences between observed and expected species assemblages in this ecoregion at all urbanization levels and across all stream orders. EP [Eastern Piedmont] assemblages showed less congruence among stream orders across

urbanization levels. Urbanization was apparently more intense in 2nd-order sites than 1st- or 3rd-order sites; effects were potentially enhanced by the greater expected species richness (10) than in 1st- or 3rdorder sites (4 and 8, respectively; Table 7). The 1st- and 3rd-order sites ostensibly lost 1 and 3 species, respectively, of the expected assemblage at the .75% level of urbanization, whereas 2ndorder sites lost 5 species with this level of urbanization (652).

- (Morgan and Cushman 2005) Loss of fish refugia needed to maintain biodiversity within streams in urbanizing catchments is an environmental concern within Maryland (Richter et al. 1997). Maintenance of source populations and dispersal should be key considerations in urban planning efforts (Lowe 2002). Connectivity within catchments is being destroyed by urbanization, along with daily destruction of small perennial and intermittent streams (CWP 2003) (653).
- (Norton and Fisher 2000) Stream nutrient concentrations in the Salt Fork basin, a central Illinois watershed with about 90% of its land in agriculture, was found to be more influenced by proximity of urban areas than by extent or position of cropland (Osborne and Wiley, 1988). However, with only 10% forest in the basin, most of which was in the lower portion, it is not surprising that positional effects were small in this highly disturbed basin (339).
- (Brett et al., 2005) These calculations showed the most urban streams had, on average, 122% higher SRP concentrations, 95% higher TP concentrations, and 44% higher TN concentrations during baseline conditions. Turbidity was 71% higher in the most urban catchments; however, this result is equivocal because it was obtained by excluding data from Tibbets Creek, which was both heavily forested and had the highest turbidity (Figure 2). NO3, NH4, and TSS concentrations were not statistically higher in the most urban catchments (Table 4) (336).
- (Brett et al., 2005) The results showed that stream water nutrient (and especially phosphorus) concentrations were higher in the streams draining urban catchments. In addition, we developed regression equations showing that a conversion of 10% of catchment area from forest to urban land cover will result in an increase in average stream water concentrations of 7 lg/L for TP, 4 lg/L for SRP, 73 lg/L for TN, 57 lg/L for NO3, 1 lg/L for NH4, 0.2 NTU for turbidity, and 0.021 mg/L for TSS (337).
- (Brett et al., 2005) The TP concentrations in Seattle area urban streams were, on average, 48% less than those observed in Omernik's (1976) ''agricultural streams''; similarly, the SRP, TN, and DIN concentrations in Seattle urban streams were 39%, 66%, and 65%, respectively, lower than those observed in Omernik's agricultural streams. Overall, Omernik's agricultural streams had about twice as much phosphorus and three times as much nitrogen as observed in Seattle area urban streams, however, Seattle area forest streams had substantially higher nitrate concentrations than did Omernik's (339).
- (Brett et al., 2005) Given these caveats, this alternative means of estimating extreme urban enrichment values suggests that completely urban catchments would have 291% higher TP, 763% higher SRP, and 85% higher TN concentrations, as well as 94% higher turbidity compared to ''pristine'' streams. These results also predict 99% higher nitrate and 48% higher ammonium concentrations in fully built-out streams; however, these results were not statistically significantly (340).
- (Brett et al., 2005) These results show that the phosphorus content of wadable streams in the Seattle, Washington, area during baseline conditions was moderately strongly correlated with urban land cover, whereas stream water nitrogen concentrations were only weakly correlated with land cover. TSS concentrations were not correlated with land cover and turbidity was only weakly correlated (340).
- (Brett et al., 2005) Phosphorus and nitrogen concentrations in Seattle area urban streams were, on average, only 50% and 30%, respectively, as high as what has previously been reported by agricultural streams nation in the United States (340).
- (Roy et al., 2007) However, endemic:cosmopolitan richness, cosmopolitan abundance, and lentic tolerant abundance were related to % forest cover in the 1-km stream reach,

but only in streams that had < 15% catchment urban land cover. In these cases, catchment urbanization overwhelmed the potential mitigating effects of riparian forests on stream fishes. Together, these results suggest that catchment land cover is an important driver of fish assemblages in urbanizing catchments, and riparian forests are important but not sufficient for protecting stream ecosystems from the impacts of high levels of urbanization (385).

- (Dodds and Oakes 2008) In areas such as the Midwestern United States largescale land use conversion has resulted in some of the worst water pollution in the United States (U.S. EPA 2000) and imperilment of many native aquatic species (Fausch and Bestgen 1997) (368).
- (Pringle 2001) Emissions from fossil fuel-burning power plants and other industrial facilities have caused serious pollution problems (including acid deposition) in biological reserves throughout the world, from South Africa's Kruger National Park (Scholes 1995, Fig. 3B), to Poland's Tatra National Park (Kot 1992), to Acadia National Park in Maine, USA (NPCA 1993). For example, Great Smoky Mountains National Park (2107 km2) in North Carolina and Tennessee, USA, is downwind of many urban and industrial areas that generate millions of tons of air pollution annually (Shaver et al. 1994). Its regional nitrate deposition is the highest of any monitored site in North America. The park includes the largest remaining area of old-growth red spruce (*Pices rubens*) and Fraser fir (*Abies fraseri*) in the world. There is evidence in the park that acid deposition and associated pollutants are altering forest resistance to winter injury (Barnard and Lucier 1990), and are contributing to forest declines in the northern Appalachians (992).
- (Gage et al., 2004) We found a significant disturbance effect on family richness (Table 2). On average, there were more insect families in low disturbance streams than in irregular disturbance streams (353).
- (Gage et al., 2004) However, the total number of insects, the number of insect families, and the abundance of pollution-sensitive EPT taxa were all lower in high disturbance streams than in low disturbance streams, and we found that abundance and diversity declined with an increase in the percentage of developed land in a watershed. Plecopterans in general have low tolerance values (Lenat 1993), and were rare in irregular disturbance streams and nonexistent in high disturbance streams. Several other families were found only in low disturbance streams or in low abundance in irregularly disturbed streams (Table 3) (354).
- (Gage et al., 2004) Disturbances that introduce sediment result in the death or drift of aquatic insects (Hochmoeller et al. 1991, Lenat and Crawford 1994) (354).
- (Gage et al., 2004) Irregular disturbances that directly affected streams, such as drought or near-stream construction, caused declines in insect diversity and abundance, and dissolved oxygen (354).
- (Grau et al., 2003) Widespread effects of LUCC include habitat loss and fragmentation, soil degradation, species introductions, and changes in vegetation. Indirectly, LUCC affects the interactions between the biosphere and the atmosphere (through alterations of biogeochemical cycles) and between ecosystems and cultural systems (Turner et al. 1990) (1159).
- (Novotny et al., 2005) Instead of or in addition to an irreversible dominant surrogate stressor expressed, e.g., by percent imperviousness or percent urbanization, other stressors may be significant and more manageable. Obviously, for nonurban streams landscape features such as percent forested or agricultural area of the watershed (Wang et al., 2000; Van Sickle, 2003), riparian zone conditions and buffers, geology of the watershed and morphology of the stream, ecoregional attributes (Omernik, 1987; Omernik and Gallant, 1989) or hydrologic stressors such as flow variability (Poff and Ward, 1989) are important. The other surrogates of stresses such as agricultural or forest land become important as the dominating effect of urbanization diminishes at low percentages of imperviousness but may have the same drawbacks as using percent imperviousness (189).

- (Voelz et al., 2005) Agricultural land use occurred along both rivers between sites 3 and 4, however our results indicated that the urban influences had already impacted the macroinvertebrate assemblages especially for the Cache la Poudre River (196).
- (Voelz et al., 2005) In general, most of the water chemistry parameters for the Big Thompson River exhibited increased levels downstream from site 1 [located above all known impacts], with large increases at site 3 [located 1.4km below wastewater treatment plant discharge] (NH_4–N and NO_3–N) or site 4 [located below all potential impacts from the City of Loveland] (PO_4–P) (193).
- (Voelz et al., 2005) The overall longitudinal chemical gradient in the Cache la Poudre River was similar to that observed in the Big Thompson, with large increases occurring at sites 2 [located 800m upstream of wastewater treatment plant discharge] and 4 [located below all potential impacts from the City of Loveland] (193).
- (Andersen et al., 2007) Recruitment can be spatially restricted by flow alteration as noted earlier, but also by agricultural, urban, or industrial developments or management practices that reduce or eliminate seed bed generation or seed production or that destroy seedlings or young trees (464).
- (Roy et al., 2007) For endemic:cosmopolitan richness and lentic tolerant abundance, the best supported model was % forest in the 1-km riparian area plus an interaction between % riparian forest and % urban in the catchment (Table 5). Cosmopolitan abundance was best supported by a model with% forest in the 1-km riparian area alone; however, the model that also included an interaction term with % urban explained the most variation in abundance of cosmopolitans across sites. The model with the interaction term suggests that these fish assemblage variables are related to % forest in the riparian area, but the slopes of the models are different for different levels of urban land cover. Thus, we plotted the relationships with % forest in riparian areas, and regressed fish variables against riparian forest according to categories of % urban land cover in the catchment (Fig. 4). Interestingly, only sites with <15% urban land cover were related to % forest in the riparian area. In other words, sites with >15% urban land cover have consistently low endemic:cosmopolitan richness and high cosmopolitan and lentic tolerant abundance, regardless of % forest in the riparian area (Fig. 4) (393-394).
- (Roy et al., 2007) We hypothesized that streams in urban settings would not respond to differences in riparian forest cover because catchment-level processes would overwhelm reach-scale land cover and reduce assemblage integrity. Richness of endemic:cosmopolitan species was positively associated with % forest in the 1-km riparian area only in sites with <15% urban cover in the catchment, thus broadly supporting this hypothesis. Similarly, abundances of cosmopolitan and lentic tolerant species declined with increased % riparian forest cover, particularly in sites with low levels of catchment urbanization. This suggests that a forest cover in both catchments and riparian areas is important for moderating effects of relatively low levels of urbanization, but at high levels of catchment urbanization the potential benefits of riparian forests are overwhelmed (396-397).
- (Pringle 2001) Abandoned, existing, and future mines are also serious threats to Yellowstone Park. Toxic mine drainage can be transmitted throughout the park landscape via hydrologic connectivity. For example, large flood events on the park's eastern boundary could wash massive quantities of toxic mine tailings downstream into Soda Butte Creek (Fig. 5B; NPCA 1993) (992).
- (Pringle 2001) Even though many claims do not lie directly within the park's watershed, pollutants from mining downstream of the park could also harm park resources (e.g., migrating trout) (992).
- (Palmer et al., 2005) A major problem in urban streams is an increase in peak flows because of runoff from impervious surfaces in the watershed (212).
- (Lewis et al., 2007) The chemical and physical properties of the WWTP effluent differed from the composition of stream water upstream of the WWTP. Compared to stream water immediately upstream (site 14), the effluent had higher concentrations of solutes except Si4+, HCO_3 , and DOC (Table 2). Magnesium and NO_2 concentrations in the effluent

were only marginally greater than concentrations in stream water. Proportionately, the effluent had higher proportions of Na+ + K+ and SO2_ 4 + Cl− (Fig. 3). In addition, the effluent was less turbid than stream water (Table 2).

Stream chemistry changed significantly downstream of the WWTP on the eastern branch of the river. Among all sampling sites in the watershed, the highest conductivity and mean concentrations of Na+, K+, Cl−, H2PO_4 , NO_3 , SO_4 , TDN, and DON occurred at the two sites downstream of the WWTP (Figs. 2 and 4). Conductivity, pH, and concentrations of Na+, K+, Cl−, H2PO_4 , NO_3 , SO_4 , TDN, DON, and DOC were significantly higher downstream than upstream of the WWTP, though in the case of pH and DOC these differences were small (Figs. 2, 4, and 5). In addition, temporal variations in conductivity and concentrations of Na+, Cl−, TDN, NO_3 , H2PO_4 , K+, and SO_4 were greater at the site (13) immediately downstream of the WWTP than at most other sites in the watershed (Figs. 2 and 4). This was especially evident in H2PO_4 concentrations. Carbon dioxide saturation was significantly lower downstream than upstream of the WWTP (Fig. 5). However, the differences in concentrations of Ca2+, Mg2+, Si4+, HCO_3 , and TDC (Figs. 2 and 5), as well as differences in water temperature, dissolved O2, and turbidity (data not shown), were not significant (310).

- (Lewis et al., 2007) In addition, temporal variations in conductivity and concentrations of Na+, Cl−, TDN, NO_3 , H2PO_4 , K+, and SO_4 were greater at the site (13) immediately downstream of the WWTP than at most other sites in the watershed (Figs. 2 and 4). This was especially evident in H2PO_4 concentrations. Carbon dioxide saturation was significantly lower downstream than upstream of the WWTP (Fig. 5). However, the differences in concentrations of Ca2+, Mg2+, Si4+, HCO_3 , and TDC (Figs. 2 and 5), as well as differences in water temperature, dissolved O2, and turbidity (data not shown), were not significant (310).
- (Lewis et al., 2007) The release of effluent from the WWTP markedly increased discharge of the eastern branch, at least in mid-July. On 16 July 2003, discharge at site 14 was 0.19 m3/s. Effluent discharge on the same day was 0.08 m3/s. Therefore, effluent would have increased river flow by ~40%, accounting for the change in chemical composition (311).
- (Lewis et al., 2007) Mean algal chlorophyll a at site 13 downstream of the WWTP (0.25 µg/cm2; SE=0.0009) was approximately 12 times higher than the mean chlorophyll upstream at site 14 (0.02µg/cm2; SE=0.006). However, algal chlorophyll was highly variable at site 15 further upstream from the WWTP (mean=0.10 µg/cm2; SE=0.09). As a result, chlorophyll a did not differ significantly among the three sites (Kruskal-Wallis test, p=0.51) (313).
- (Lewis et al., 2007) Effluent from the WWTP strongly affected river conductivity and the concentrations of nine solutes (Figs. 2 and 4). These effects are consistent with effects of WWTP effluent on river chemistry in the lower piedmont of South Carolina (Andersen et al., 2004) (317).
- (Lewis et al., 2007) At site 13, mats of algae several millimeters thick covered much of the bedrock streambed, and gas bubbles formed on the mats on sunny days, indicating high rates of photosynthesis. Therefore, we believe that nutrients from the WWTP effluent stimulated primary productivity in the eastern branch of the river (317).
- (Bernhardt and Palmer 2007) A recent synthesis of river restoration project information for the United States, the National River Restoration Science Synthesis (NRRSS). (Bernhardt et al., 2005), suggests that urban streams receive a disproportionately large share of river restoration monies and effort (Fig. 5). In Maryland, for example, 30% of all river restoration projects over the last decade and about 50% of all reported river restoration funds were spent in the four (of 23) most densely populated counties (Hassett et al., 2005). In part, this concentration of river restoration effort in urban areas may be a response to the more intense degradation in these systems. However, much of the restoration may be motivated by needs to protect streamside infrastructure or by requirements to spend mitigation monies within the same political boundaries as new development. It may also be argued that a large portion of taxpayer money devoted to

restoration should be spent to improve the immediate environment of cities, where the majority of people live (742).

- (Palmer et al., 2005) Different restoration activities should be selected based on the extent and type of damage, land-use attributes of the catchment, the size and position of the river within the catchment, and stakeholder needs and goals. Even when constraints are significant, there are almost always choices that are more or less ecologically sound, as illustrated by the following four examples (212).
- (Ekness and Randhir 2007) Focusing on the headwaters and limiting the number and types of land uses with high disturbance values could be beneficial to the whole drainage system. Some longitudinal policies could improve regional connectivity in open space and low disturbance areas. Longitudinal restoration can be increased by using greenways to establish regional connectivity in watersheds (Wenger and Fowler, 2000) (1480).
- (Ekness and Randhir 2007) The statistical analysis shows that potential habitat of vertebrate species declined with an increase in land use disturbance (1478).
- (Ekness and Randhir 2007) Land use disturbances decreased habitat potential, with maximum habitat potential in areas with no disturbance (1480).
- (Gage et al., 2004) Although one irregular disturbance stream had the highest percentage of developed area (Reeds: 48% developed area, Table 1), early on it had high insect diversity (Figure 3b), indicating that the percentage of developed land is not the sole or best predictor of benthic insect diversity. A major disturbance occurring in or next to the stream had an overriding effect on diversity and abundance of insects (354-355).
- (Gage et al., 2004) The suspended sediment concentrations in the streams increased dramatically due to the direct pumping of muddy water into the streams as ditches were dug nearby. Not only was there no attempt to control erosion, we observed workers pumping mud from ditch digging directly into the stream. The effect of this in Reeds Creek was to reduce the total number of insects from an average of 314 individuals/sample in May and June to 54 individuals/sample in October (355).
- (Thompson et al., 2005) It is unlikely that dam removal alone will cause a substantial improvement in downstream ecological integrity because the Manatawny dam had minimal downstream effects on water quality (Bushaw-Newton et al. 2002), and urban impacts continue Unabated (201).