Recovery Potential Metrics Summary Form

Indicator Name: CORRIDOR PERCENT FOREST

Type: Ecological Capacity

Rationale/Relevance to Recovery Potential: Broad array of influences on capacity to recover (see notes below), including intercepting and moderating the timing of runoff, buffering temperature extremes (which can also reduce certain toxicities), filtering pollutants in surface or subsurface runoff, providing woody debris to stream channels that enhances aquatic food webs, and stabilizing excessive erosion.

How Measured: Simplified calculation involves defining a standard corridor width on both sides of a watercourse (e.g. 30 meters, 90 meters) and calculating % area within the corridor. Also possible to calculate area within a variable-width corridor (e.g., an estimated flood return frequency zone).

Data Source: Land cover datasets are available through the National Land Cover Pattern Database (See: http://www.mrlc.gov/index.php). For land cover data, the National Land Cover Database (NLCD) for 2006, 2001 and 1992 is accessible at http://www.mrlc.gov/finddata.php; numerous statewide land cover mapping datasets are also available from state-specific sources. Land cover for coastal areas is available through NOAA's Coastal Change Analysis Program (See: http://www.csc.noaa.gov/digitalcoast/data/ccapregional/index.html) Forest cover can be obtained from the national forest census, compiled by the Forest Inventory and Analysis National Program (See: http://fia.fs.fed.us/tools-data/default.asp). Orthophoto maps or remote imagery can be a good source for detailed local information.

Indicator Status (check one or more)

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

Comments: widespread applicability for lotic waters in regions where riparian corridors are naturally forested.

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

- (ourso and frenzel 2002) Riparian buffer strips can successfully sustain many important aquatic habitat components (Schueler, 1995; Shaw & Bible, 1996).
- (parkyn et al 2003) Generally, streams within buffer zones showed rapid improvements in visual water clarity and channel stability, but nutrient and fecal contamination responses were variable. Significant changes in macroinvertebrate communities toward "clean water" or native forest communities did not occur at most of the study sites. Improvement in invertebrate communities appeared to be most strongly linked to decreases in water temperature, suggesting that restoration of in-stream communities would only be achieved after canopy closure, with long buffer lengths, and protection of headwater tributaries. Planting (or preserving) riparian buffer zones can be an effective means of reducing nutrient and sediment loads into streams (Barling & Moore 1994; Williamson et al. 1996; Fenney & Cronk 1997; Martin et al. 1999). However, the extent to which buffers can restore riparian ecosystems in terms of ecological function and species composition is essentially unknown (Ebersole et al. 1997; Jorgensen et al. 2000).

- (parkyn et al 2003) Across the nine sites there were few consistent improvements in water quality and habitat and few changes in the invertebrate community to suggest that the [planted] patches of buffer zone had improved stream health.
- (parkyn et al 2003) In general, there was little evidence of the channel widening phenomenon expected after riparian shading of formerly pasture stream banks (Sweeney 1993; Davies-Colley 1997). Restoration of shade is expected to result in substantial losses of stored stream bank sediment as the armoring effect of riparian grasses is reduced, at least in upland catchments (Collier et al. 2001; Parkyn et al. 2001) Loss of in-stream nutrient processing is also associated with shade.
- (parkyn et al 2003) Modeling studies in New Zealand (Rutherford et al. 1999) indicated that the deeper and wider the stream, the longer the buffer will need to be to achieve reductions in temperature, for example, 1–5 km for first-order streams versus 10–20 km for fifth-order streams, with 75% shade, to achieve a 5C reduction in temperature.
- (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. Studies have found strong positive relationships between diverse assemblages of stream benthic macroinvertebrates that are intolerant of water quality degradation and watershed-wide forested land cover (Lenat and Crawford 1994, Stewart and others 2001, Weigel and others 2003) or forested land cover within riparian zones (Basnyat and others 1999, Sponseller and others 2001, Stewart and others 2001, Weigel and others 2003). 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) Two of the three watershed land cover variables — percent agricultural and percent forested — exhibited somewhat strong relationships. The percent of agriculture land cover at the watershed scale had a positive relationship with the indices, meaning that it was negatively correlated with aquatic ecological integrity. The percent of forest was correlated with better stream conditions. In our statewide analysis, the percent of forest cover at the watershed scale and in riparian zones were highly correlated enough (0.776) that the two have similar value as predictors of macroinvertebrate tolerance for water quality degradation. 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.
- (Paul and Meyer 2001) Riparian deforestation associated with urbanization reduces food availability, affects stream temperature, and disrupts sediment, nutrient, and toxin uptake from surface runoff. Invertebrate bioassessment metrics decreased sharply in Puget Sound,Washington tributaries with increasing ISC (Horner et al. 1997). However, streams that had higher benthic index of biotic integrity scores for a given level of ISC were always associated with greater riparian forest cover in their catchment, suggesting that riparian zones in some urban catchments may buffer streams from urban impacts. Above 45% ISC, all streams were degraded, regardless of riparian status. The value of riparian forests is also reduced if the stormwater system is designed to bypass them and discharge directly into the stream (350).

- (Moore and Palmer 2005) In areas that are destined for development, the patterns we document suggest that maintenance of riparian forested buffers is vital, even in the most urbanized areas (1176).
- (Moore and Palmer 2005) The second implication arises from our finding that urban streams with the greatest amount of intact riparian forest buffer had higher levels of biodiversity than other urban streams we studied (Fig. 5). This suggests that efforts to restore or preserve riparian buffers, even when there is a substantial amount of paved surface in urban watersheds, may mitigate some of the impacts on stream biodiversity. Thus, from a biodiversity perspective, headwater streams in areas already highly urbanized should not be viewed as lost causes; a balance between conservation, restoration, and ecologically designed solutions to the problems caused by urbanization may be warranted (Palmer et al. 2005) (1174).
- (Moore and Palmer 2005) The strong positive relationships between riparian forest and taxa richness for our urban streams (Fig. 5) suggests that the presence of intact riparian zones may mitigate some of the impacts of urbanization on biodiversity loss. Urban development, with subsequent impacts on streams, is only expected to increase globally, and statistical trends in the United States show that this form of land use change is effectively irreversible (Irwin and Bockstael 2005). Thus, the maintenance of riparian buffers in urban areas may become increasingly important (1175).
- (Iwata et al. 2003) During the last two decades, an enormous research effort has illustrated that the maintenance of stream ecosystems depends on land-use regimes in the surrounding terrestrial ecosystems (Gregory et al. 1991, Naiman and De´camps 1997, Naiman et al. 2000). For example, because riparian forest plays important roles in regulating stream hydraulics, substrate characteristics, light and thermal regimes, water chemistry, and organic matter supply, riparian deforestation impacts on stream habitats profoundly, which in turn affect various ecological assemblages in the stream communities (e.g., Sweeney 1993, Harding et al. 1998). Therefore, the maintenance of riparian forest has become an integral component of management strategies for stream biodiversity conservation (Naiman and De´camps 1997, Naiman et al. 2000) (461).
- (Norton and Fisher 2000) Riparian forests positively impact water quality by: (1) acting as effective sediment traps; (2) consuming and storing nutrients by accreting biomass; (3) stimulating microbial assimilation of nutrients in forest soils and (4) providing an environment conducive to anaerobic microbial dissimilation of nitrate to nitrogen gas (denitrification) or ammonia. The ability of riparian vegetation to intercept nutrients and eroded sediment depends not only on the presence of vegetation but also on a microbial energy source, (electron donor), adequate temperatures, redox conditions, groundwater aquifer characteristics, and position of vegetation within the landscape (338).
- (Norton and Fisher 2000) A riparian forest studied by Jordan et al. (1993) effectively removed nitrate from an adjacent corn field. The groundwater contained nitrate concentrations of _8 mg l_1 at the edge of a corn field which was reduced to 0.4 mg l_1 halfway (20 m) into the forest. A riparian forest studied by Peterjohn and Correll (1984) demonstrated a 90% reduction in annual sediment load, an 80% reduction of nitrate in overland flow and an 85% reduction in groundwater nitrate originating from an adjacent agricultural field. Lowrance et al. (1984) calculated that riparian forest in Georgia's Little River Watershed retained 68% of N and 30% of P received from adjacent cropland and precipitation (338).
- (Norton and Fisher 2000) Riparian forest may have minimal impacts on reduction of N in overland runoff (Verchot et al., 1997) and long-term sediment-bound P (Whigham et al., 1988). However, restoration and conservation of coastal plain forest should be encouraged for several reasons. Forest provides valuable habitat for wildlife, stabilizes stream banks, lessens the impact of storm discharge associated with flooding and provides recreational and scenic resources. An additional benefit is that land which contains forest is not occupied by cropland, and is not being fertilized. While forest in the Chester may not have the same water quality functions as those observed in the Choptank, it does not exclude them from having value (359).

- (Bernhardt and Palmer 2007) Moore & Palmer (2005) found that while the invertebrate diversity of headwater streams in suburban Maryland decreased with the proportion of impervious cover in the catchment, there was a positive effect of the extent of intact riparian vegetation on urban stream macroinvertebrate taxa richness (Fig. 4). Sudduth & Meyer (2006) found in both urban and urban restored streams that macroinvertebrate richness and biomass were strongly correlated with the per cent of streambanks covered with roots or wood, indicating that biological structures could improve habitat quality (741).
- (Bernhardt and Palmer 2007) However, the presence of riparian vegetation along urban streams is important regardless of the width of the buffer. Not only does it improve bank stability but the generally low aquatic biodiversity in urban streams may be enhanced in reaches where the riparian zone is intact (Moore & Palmer, 2005, but see Walsh, 2004, and Walsh et al., 2005a for a discussion of contrasting results) (746).
- (Bernhardt and Palmer 2007) In the Paint Branch catchment of heavily suburban Montgomery County in Maryland, progressive urban planning has led to purchase and preservation of large areas of riparian forest and used aggressive zoning laws to limit new development in the catchment. This approach has been successful at maintaining high water quality and supporting reproducing populations of trout despite very high impervious cover within the catchment (Montgomery County DEP, 2003) (747).
- (Andersen et al., 2007) Riparian forests and woodlands provide important ecosystem services through their high productivity and habitat values (Finch and Ruggiero 1993; Hughes 1994; Knopf and others 1988; Skagen and others 2005; Wright and Flecker 2004), the food and fiber they provide (Gregory and others 1991), and the organic matter they supply to aquatic ecosystems (Angradi 1994; Townsend-Small and others 2005) (453).
- (Andersen et al., 2007) A large body of research has established a clear link between the health and persistence of dryland riparian forests and natural fluvial geomorphic and hydrologic processes (Andersen 2005; Hughes 1994; Jolly and others 1993; Rood and others 2003a) (453).
- (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).
- (Poiani et al., 2000) The minimum dynamic area for the Yampa River riparian ecosystem must maintain recolonization sources for each internal patch type and provide room for the geomorphic processes that reshape the floodplain and create and destroy the complete array of patch types (143).
- (Poiani et al., 2000) Not only is riparian vegetation more vulnerable to widespread destruction by floods, but future sources of propagules for post-disturbance recovery may also be severely reduced as a result of a narrowed and fragmented riparian corridor (143).
- (Ekness and Randhir 2007) Lateral [riparian] and longitudinal [stream order] connectivity and flow regime are critical factors that influence watershed health. The latter can be impaired by land and water use practices that affect biotic diversity, water quality, esthetics and hydrology (Brooks et al., 1997) (1469).
- (Ekness and Randhir 2007) The riparian gradient is an important component of a watershed and maintenance of a ‘‘natural’’ gradient plays a vital role in protecting water quality (Corell, 1996; Novak et al., 2002), flood control, and ground-water recharge, and provides habitat for a variety of organisms (Naiman and Decamps, 1993). Riparian vegetation regulates hydrologic fluxes, light incidence, temperature, physical habitat, food and energy, and nutrient flows (Gregory et al., 1991; Sweeney 1992) into and out of the

riverine-riparian zones (Lynch et al., 2002). The riparian system contributes organic material in the form of woody debris and leaf litter to a riverine system (Gregory et al., 1991; Hubbard and Lowrance, 1997) and reduces bank erosion (Hubbard and Lowrance, 1997). This flow of energy between the terrestrial and aquatic ecosystems occurs from a more to a less productive habitat (Polis and Strong, 1996). Thus, riverine ecosystems in which the river inundates the riparian zone are more productive because they are acquiring organic maternal from the nearby terrestrial ecosystems. Riparian vegetation and litter that fall from upland environments are major sources of nutrients to streams (Oelbermann and Gordon, 2000). Minerals and oxygen levels are affected by riparian zones as well. McGlynn et al. (1999) observed that Ca concentrations increase with depth and DOC concentrations decrease with depth in riparian zones. The riparian zone was also found to effectively limit the movement of phosphorus enriched sediments (Novak et al., 2002) and assimilate NO3-N (Hubbard and Lowrance, 1997) (1469).

- (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) 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).
- (Roy et al., 2007) Results from this study and other studies suggest that human alteration affects stream processes at multiple spatial extents. In addition to % land cover within catchments and riparian areas, the continuity of riparian forests (Stewart et al. 2001) and historic land use in the catchment (Harding et al. 1998) likely also influence fish assemblages (398-399).
- (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).
- (Poole and Downing 2004) Successful protection or restoration of mussels in regions that have undergone major alterations in land use over the past century must address the factors degrading stream conditions for the biota and the factors impeding recolonization. Restoration and long-term protection of mussel biodiversity should therefore address the restoration of riparian zones and the increased protection of streams from agricultural influences (124).
- (Iwata et al., 2003) Moreover, recently expanded slash-and-burn (shifting) agriculture, a major cause of the forest destruction in Borneo, produces more excessive sediment than

traditional swidden agriculture or logging operations (Douglas et al. 1993, MacKinnon et al. 1996). Therefore, riparian deforestation associated with slash-and-burn agriculture may impact on the stream biodiversity in Borneo more strongly than we expect on the basis of the empirical knowledge obtained in temperate streams (462).

- (Iwata et al., 2003) The results revealed strong ongoing effects of past riparian deforestation on the stream habitats, emphasizing the critical functions of riparian forest in habitat maintenance of the tropical streams (467).
- (Iwata et al., 2003) These results suggest that the loss of riparian forest functions, especially the role of tree root systems in stabilizing stream banks and preventing surface erosion (e.g., Chamberlin et al. 1991, Tabacchi et al. 2000), induced inputs of a large quantity of fine sediment into the secondary-forest reaches. The important role of riparian forests in stream habitat formation was also shown by the regression analysis (Fig. 4). The linear relationship between the degree of riparian forest development (represented by stand density, mean dbh, or basal area) and the gradient of depositional character (PC 1) suggests that the stream habitats once thoroughly altered by past deforestation were recovering toward a state resembling the predisturbance conditions with redevelopment of the riparian forests. However, the difference in depositional character between the primary- and secondary-forest reaches was still evident (Table 1 and Fig. 3), indicating that stream habitats had not yet fully recovered despite the elapse of 9–20 yr following the cessation of agricultural activities. The habitat alteration strongly influenced community structure (468).
- (Walsh et al., 2005) Deforestation, particularly in the riparian zone, is often identified as an important driver of urban impacts to streams (e.g., Stephens et al. 2002, Booth 2005). Urban land use and riparian degradation usually covary (e.g., Morley and Karr 2002, Burton et al. 2005, King et al. 2005), with lowland urban development often resulting in restructuring or loss of riparian vegetation (714).
- (Dodds and Oakes 2008) Riparian land use may be particularly influential and, in some cases, a better predictor of in-stream water quality than land cover in the entire catchment (Johnson and others 1997; Osborne and Wiley 1988). Intact riparian zones provide water quality benefits and help preserve the biological integrity of watersheds (Gregory and others 1991) (368).
- (Dodds and Oakes 2008) Across all studied watersheds, riparian land cover was a significant predictor of among-site variation in water chemistry concentrations at the watershed and first-order streams scales, particularly for nutrients (Table 1) (371).
- (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) In October 1996, the Executive Council of the CBP established a goal of restoring riparian forest buffers on 2010 miles of streams in the Chesapeake Bay watershed by 2010. Commitment to riparian restoration was continued in the 2000 Chesapeake Bay Agreement. These actions exemplified the widespread acceptance of riparian buffers as an important tool for the reduction of non-point-source pollution and enhancement of stream ecological condition (2).
- (Poole and Downing 2004) It seems reasonable, however, that the characteristics of whole watersheds should influence long-term resistance of mussel communities to perturbation when viewed at the small scale. Our analyses uphold this concept because watersheds with the most habitat converted to farmland had the greatest levels of decline in richness. This effect is echoed at the smallest scale by the association of deforested riparian zones in agricultural watersheds with declining richness. Also at the smallest scale, the lowest rates of declining biodiversity were associated with diversity of substrata (123).
- (Gergel et al., 2002) When examining deforestation of riparian zones, Jones et al. (1999) found decreases in fish abundance with increasing length of the nonforested riparian patch. Several species changed dramatically at particular patch lengths (Jones et al., 1999). They suggested that length and area of buffer zones should be emphasized in addition to patch width in mitigation and management (Jones et al., 1999) (120-121).

- (Norton and Fisher 2000) Riparian forest has tremendous value as ecological habitat and for recreation, but its ability to improve water quality is a function of its interaction with the hydrologic flow path (360).
- (Gage et al., 2004) Streambed characteristics and hydrology of small streams may change as development decreases riparian forests and increases sedimentation (Beschta 1996, Décamps 1993, Gore 1996, Swank et al. 1988). This may decrease flow, which can cause a drop in dissolved oxygen and an increase in temperature, eliminating taxa sensitive to those changes (Boulton and Suter 1986). Increased disturbance thus leads to decreased diversity in streams (Hogg and Norris 1991, Karr 1991, Lamberti and Berg 1995, Lemly 1982, Lenat and Crawford 1994, Schleiger 2000), and tolerant species may come to dominate the macroinvertebrates (Closs and Lake 1994) (346).
- (Gage et al., 2004) Sedimentation and turbidity also increase as headwater forests are cleared (Lamberti and Berg 1995), and this decreases species richness, the proportion of pollution-sensitive species, overall insect biomass, and abundance (Beschta 1996, Kemp and Spotila 1997, Lamberti and Berg 1995, Lemly 1982, Oberlin et al. 1999, Schleiger 2000
- (Roy et al., 2007) For example, many studies have suggested that forested riparian areas, which provide shading, organic inputs, stream-bank protection, nutrient uptake, and other essential functions for stream ecosystems, indirectly maintain fish assemblage integrity (Steedman 1988; May et al. 1997; Lee et al. 2001). Further, the longitudinal scale of riparian land uses (i.e., whether it is adjacent to the stream reach or extends upstream along the stream network) may also influence relations between land cover and in-stream communities (Roth et al. 1996; Lammert and Allan 1999; Wang et al. 2001) (386).
- (Radwell and Kwak 2005) Fish density, number of intolerant fish species, and invertebrate density were important biotic variables responsible for the rankings. Contributing physical variables included riparian forest cover, nitrate concentration, turbidity, percentage of forested watershed, percentage of private land ownership, and road density both in the watershed and in a 100-m buffer (806).
- (Ducros and Joyce 2003) The decision in this case to allocate at least 1 point to all criteria recognized the inherent environmental benefit of converting any intensively managed agricultural land into a buffer zone under the WFO scheme (255).
- (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).
- (Ducros and Joyce 2003) Trees contribute to habitat heterogeneity in buffer zones (Bren 1993, De Jalo´n 1995, Harper and others 1995) as well as assisting in subsurface water quality improvement (Osborne and Kovacic 1993). The value of buffer zones established on former arable compared to existing grassland was similar (Figure 3), indicating that both land uses are suitable for buffer zone creation. However, the slightly lower vegetation-related scores for arable conversions suggest that habitat enhancement may take longer to deliver in former arable zones (263).
- (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) Despite the lack of research in urban areas, forested riparian areas have been widely applied across the US to protect aquatic resources from all anthropogenic land uses (Lowrance 1998; Pusey and Arthington 2003). We hypothesize that streams in urban landscapes are overwhelmed by upstream disturbances, and that forested riparian patches do not influence fish assemblage integrity in urbanizing areas (386).
- (Roy et al., 2007) Reach-scale riparian forest cover was not strongly related to richness or abundances of sensitive fish species. However, high proportions of riparian forests along the lower 1-km reach, and, to a lesser extent, along the 200-m reach were negatively related to the abundance of cosmopolitan and lentic tolerant species. Cosmopolitan and lentic tolerant species include ictalurids (e.g., bullheads and catfish) and centrarchids (e.g., sunfish) that are typically tolerant of bed sedimentation, high levels of nutrients, and low dissolved oxygen typical of disturbed streams (Detenbeck et al. 1992; Jones et al. 1999). Further, these groups of species have habitat and trophic requirements conducive to disturbed conditions; they spawn in nests constructed of fine sediment, and they are primarily omnivores or trophic generalists (Etnier and Starnes 1993; Mettee et al. 1996) (395-396).
- (Roy et al., 2007) Conversely, in streams with one dominant land cover (and a very small range) we are more likely to observe shifts associated with minor changes in riparian land cover if we look at the appropriate scale. For example, Stauffer et al. (2000) and Lee et al. (2001) found that small increases in local forest cover within the riparian area resulted in shifts toward higher fish assemblage integrity in catchments that were dominated (88–100%) by agricultural land cover. Similarly, Jones et al. (1999) documented changes in fish assem blages with local riparian deforestation in primarily forested (96–100%)watersheds. In this study we found minimal evidence that reach-scale riparian forests were driving fish assemblages, possibly because the streams lie within landscapes that have multiple land uses, and because there were large differences in basin land cover across sites (394-395).
- (Roy et al., 2007) By addressing the mechanism of urban impacts on stream ecosystems, Roy et al. (2006) found that the influence of riparian forest cover was dependent on the level of instream habitat disturbance (i.e., sedimentation in stream beds) (397).
- (Roy et al., 2007) Riparian forests have been used for managing non-point source disturbances in the US since the late 1960s (Calhoun 1988; Lee et al. 2004), due to their role in ''buffering'' aquatic resources from upland disturbances (e.g., taking up nutrients and other contaminants, retaining sediment, etc.; Lowrance 1998). These regulations imply that upland disturbances can be mitigated by protecting land adjacent to streams (Allan et al. 1997; Harding et al. 1998). However, research continues to suggest not only that catchment land cover is an important driver of biotic assemblages, but also that riparian forests are not sufficient for protecting stream ecosystems in highly disturbed areas (Allan and Johnson 1997; Harding et al. 1998; Roy et al. 2006). Importantly, our results show that at low levels of urbanization (< 15%), riparian forests can moderate upland disturbances and help to maintain fish assemblage integrity. Although forested riparian buffers may not be sufficient for protecting fish assemblages in highly urbanized areas, these results do not imply that riparian forests are unimportant. In addition to buffering streams from upland disturbances, riparian forests have been recognized for their importance in providing shade, organic material, bank stabilization, and other essential functions for stream ecosystems (see reviews Gregory et al. 1991; Sweeney 1992; Naiman and Decamps 1997; Lowrance 1998; Pusey and Arthington 2003). Based on the number of potential linkages between riparian alteration and fish assemblages (Pusey and Arthington 2003), it is not surprising that reach-scale, riparian conditions have

been related to some aspects of fish assemblage integrity here and elsewhere (Meador and Goldstein 2003). Since riparian forests provide certain functions such as temperature regulation and organic matter input that are essential for maintenance of stream integrity, complete removal of riparian forests would be detrimental to stream ecosystems (398).

- (Norton and Fisher 2000) Riparian forest, forest on hydric soil, and upland forest all showed strong negative correlations with stream TN and NO3_ concentrations (Fig. 7) in the Choptank basin. Thus, stream TN and NO3_ concentrations in the Choptank basin were strongly related to both forest and cropland in the surrounding area (350).
- (Barker et al., 2006) In contrast, a set of studies by Goldstein *et al.* (1996) in Minnesota and North Dakota found fish to be more related to instream, riparian, and hydrologic conditions than to watershed agricultural land use. Lammert and Allan (1999) also found that land use immediate to the stream was more predictive of fish IBI than regional land use, but was less important than instream habitat variables.AWisconsin study by Fitzpatrick *et al.* (2000) found that several spatial scales influenced fish communities, but local riparian conditions appeared to be more important than watershed land cover (3).
- (Barker et al., 2006) There was strong agreement between the MLR and regression tree models using all sites, but regional models showed different influences between regions. Both regression tree and linear models using all sites indicated that forest buffers were influential on BIBI (the regression tree identified riparian width and MLR identified woody debris). The mean BIBI for forested buffer sites (3.3) was significantly greater than the mean BIBI for all other sites (2.8). These results were consistent with the studies of Roth *et al.* (1998), which were conducted on the entire MBSS data set (12).
- (Barker et al., 2006) Forest buffers at a site were important controls for BIBI and PHI at that site in agricultural streams in both the Coastal Plain and Piedmont physiographic regions of Maryland. Because buffers were seen to act differently on these measures of biological stream health, there is no one recommended “minimum width” of buffer. The threshold width for improved PHI indicated that installation of even very narrow forest buffers (<5 m) may directly affect instream habitat. BIBI was discernibly higher only at wide buffer sites, indicating that an investment of more than 35 m may be necessary to see ecological effects. These results are consistent with the general values reported for buffer width effectiveness in the review article on buffer size requirements by Castelle *et al.* (1994) (16).
- (Barker et al., 2006) These data showed FIBI to be a deceptive measure of buffer effectiveness. Although buffers are not a controlling factor for FIBI at a site, buffer-induced hydrologic effects may exert strong controls over FIBI that these data were unable to capture (17).
- (Barker et al., 2006) The placement and width of the buffer were shown to effectively improve BIBI and PHI (17).
- (Barker et al., 2006) Both models indicated that forest buffers had a secondary influence on PHI. PHI (Figure 1) was higher for sites with forested buffers (60) than for sites with grass buffers (53), and PHI was higher for sites with forest as the adjacent cover than sites with adjacent crops (68 vs. 44). Higher PHI values (Figure 1) reflect higher geomorphic stability (14).
- (Ekness and Randhir 2007) The riparian width that has maximum habitat gains may not always be possible in most watersheds. An effective approach is to protect riparian areas with maximum possible riparian width, to protect all four vertebrate groups. Another approach is to follow a variable width policy that allows variability in riparian protection depending on local factors like land availability, habitat needs, and other community needs. Zoning regulations (Wenger and Fowler, 2000; Grant, 2001) can be used to reduce land disturbance to riparian areas. A variable buffer zone can be identified and protected using regulations. The variable width of the riparian buffer can be determined based on tradeoffs in location-specific benefits and costs of land protection. The recommended minimum width of riparian buffers is 7.6 m. A popular recommendation is to have three zones in a riparian buffer, namely undisturbed forest, managed forest, and

the runoff control area (Welsch, 1991), that have a combined width of 30 m. In Massachusetts, a width of 7.6 m is required in urban areas 61 m in rural areas (River Protection Act). Buffer width policies could be developed based on the marginal gains identified in this study. An ideal is to have a variable width (Spackman and Hughes, 1995; Wenger and Fowler, 2000; Corlett, 2001) policy that uses optimal riparian width depending on local attributes. Subsidies and incentives that are spatially targeted can be used to encourage voluntary installation of riparian buffers (1478-1479).

- (Norton and Fisher 2000) In the Choptank, forest cover was strongly associated with low TN and NO3_ concentrations. Within first order streams, the conduits of water from terrestrial to aquatic systems, the presence of forested stream banks also had a strong relationship with low stream N. In addition, the amount of riparian wetlands and degree of 'wetness' was inversely correlated with stream N in the Choptank basin. In contrast, forest cover in the Chester basin did not have a strong impact on stream nutrients regardless of landscape position and/or flooding regime. Hydrologic characteristics, rather than land cover, had the strongest effect on predicting Chester stream nutrient concentrations (359).
- (Ekness and Randhir 2007) Spatial variations created by different riparian distance, stream order, and land use affect the type and quality of habitat potential at a particular position within a watershed. The buffer is often critical in the flow of mass and energy into and out of water bodies. The longitudinal dimension reflects upstream-downstream linkages, which is a key factor in watershed ecology. Various land uses contribute to the type and level of disturbances in a watershed. The intensity and the extent of land disturbance affect the habitat potential of a watershed (1471-1472).
- (Ekness and Randhir 2007) In general, habitat potential decreases with respect to land disturbance for most vertebrates. It is highest for birds, mammals, and amphibians in undisturbed forested, nonforested wetland, woody perennial, or open water areas. Reptiles are exposed to higher disturbance in cropland and pastures, possibly because of a higher availability of prey. There was an increase in habitat potential for birds, mammals, and amphibians between disturbance values of 1 and 2. The transition between the forested and urban areas could explain this increase. This is consistent with the intermediate disturbance hypothesis, which proposes that disturbance at intermediate levels can contribute to moderate increase in species diversity at particular levels (Dial and Roughgarden, 1998). Habitat potential for all four vertebrate species declines with increases in disturbance except for the transition between disturbance Levels 1 and 2 mentioned before (1475-1476).