Recovery Potential Metrics
Summary Form

Indicator Name: WATERSHED PERCENT AGRICULTURE

Type: Stressor Exposure

Rationale/Relevance to Recovery Potential: Croplands and pastures have been linked to a wide variety of water quality and biotic impacts on waters. Common effects seen at moderate to high agricultural proportions of total watershed land cover include less diverse and more intolerant macrobenthic communities, increased nutrient loading resulting in turbid water, overall homogenization of the fish fauna, accelerated erosion and bank destabilization, suspended sediment particles carrying pesticides, pathogens, and heavy metals, habitat degradation and reduced biodiversity, and increases in specific conductivity, DIN, DRP, and TP concentrations. Although watershed agriculture is commonly linked to degraded aquatic conditions that may be difficult to reverse and quite persistent over time, it is important to note that some degree of recovery is rarely considered impossible.

How Measured: Calculated as % by area within watershed of the impaired water segment being assessed. Simple GIS operation aggregating and measuring all agricultural classes from land cover data within each watershed.

Data Source: Land cover sources include the National Land Cover Data from 1992 (See: http://www.epa.gov/mrlc/nlcd.html), 2001 (See: http://www.epa.gov/mrlc/nlcd-2001.html), and 2006 (http://www.mrlc.gov/nlcd06_data.php) as well as various state sources. The USGS lists cropland by county since 1850 (See: http://landcover.usgs.gov/cropland/index.php). Approximate watershed boundaries can be constructed by aggregating small-scale catchments from the NHDplus datasets (See: http://www.horizon-systems.com/nhdplus/). If the user chooses to use this indicator for a specific crop relevant to the study area, USDA has developed a national GIS crop dataset that can be downloaded from Geospatial Data Gateway (See: http://datagateway.nrcs.usda.gov/GDGHome.aspx). In addition, where applicable, BLM data set on range allotments and pastures can be used (See: http://www.geocommunicator.gov/GeoComm/).

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.

Status/Comments: Operational, widely applicable although specific research implications cited below may vary in regional relevance.

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

- (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.
- (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).
- (Dils and Heathwaite 1999) In addition to nutrient transport, the loss of suspended sediment in agricultural runoff is a major threat to water quality as particles may carry pesticides, pathogens, heavy metals and other pollutants. These particles may reduce the amount of sunlight available to aquatic biota, block fish gills and cause changes in biodiversity and habitat (59).
- (Allmendinger et al., 2007) Following European settlement and the development of agriculture, excessive erosion occurred on upland surfaces of the small watersheds of the region [Maryland Piedmont] (Costa, 1975). This created sediment yields four times higher than they were prior to settlement by Europeans (Costa, 1975) (1485).
- (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) 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).
- (Ekness and Randhir 2007) Haag and Kaupenjohann (2001) identify agro-ecosystems as leaky systems that emit nutrients, such as nitrates, which affect watersheds. Livestock grazing practices can cause erosion and bank destabilization, and degrade salmonid habitats (Borman et al., 1999;. Haag and Kaupenjohann (2001) use a budget approach to account for the role of corridors, consider the scale and scope of nitrogen emissions, and observe that landscape metabolism and self-purification may postpone problems of nitrogen overloads in a watershed (1471).

- (Sondergaard and Jeppesen 2007) Many streams and lakes have disappeared or have been heavily modified to reclaim land for agriculture, habitats have deteriorated following canalization, water levels have been controlled resulting in less contact with wetlands and reduced pulses, shorelines have been used for agriculture or dwelling purposes, thereby decimating or eliminating the natural vegetated buffers, and reservoirs have been established (1090).
- (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).
- (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).
- (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) While many agricultural preservation programs tout their conservation value, ecological research has historically shown that farming practices are detrimental to stream health (e.g., Rothrock et al. 1998, Genito et al. 2002). Only recently have researchers suggested that agricultural lands may support diverse and compositionally different aquatic invertebrate communities when compared to nearby urbanized areas (Lenat and Crawford 1994, Wang et al. 2000, Stepenuck et al. 2002) (1170).
- (Moore and Palmer 2005) First, the fact that invertebrate biodiversity was extremely high in the agricultural headwater streams and progressively declined along a land use gradient toward urbanization suggests that agricultural preservation programs in our region may be important to conservation of freshwater biodiversity. Invertebrate diversity and richness at our agricultural sites was almost twice that of the urban sites, with mixed land use sites falling consistently between these groups (Fig. 3) (1173).
- (Moore and Palmer 2005) We suggest below that the very high levels of invertebrate diversity we found in these Piedmont agricultural headwaters compared to other agricultural streams worldwide may be related to the management strategies of agricultural operations surrounding these streams (1173).
- (Pringle 2001) Reserves located in lower watersheds within arid regions of the world are particularly vulnerable to alterations in hydrologic connectivity from irrigation projects. Water extracted from surface and groundwater for irrigation often reduces surface water flow; this problem is compounded by irrigation return flow, which can be toxic to wildlife (e.g., Presser 1994, Presser et al. 1994). Intensive irrigation in arid regions often leads to leaching of soil minerals (e.g., selenium, arsenic, boron, lithium, and molybdenum) that then become mobilized and enter the food chain. In arid regions of the United States, irrigation is considered to be the most widespread and biologically important source of contaminants to surface water (Lemly et al. 1993) (984).
- (Nelson and Booth 2002) Construction, agriculture, mining, and timber harvesting accelerate natural erosion rates, increasing the supply of sediment to surface water (51).

- (Poole and Downing 2004) Conversion of natural landscapes to agricultural lands is driving a great deal of habitat degradation and reduced biodiversity (115).
- (Poole and Downing 2004) This result may indicate that the intensification of agricultural production over the last century in this region has removed a considerable number of habitat patches that were suitable for mussels in the past, or that many populations were on their way toward extinction triggered by massive land use change early in the 20th century (118).
- (Poole and Downing 2004) Multiple regression analysis, done on the watershed-averaged changes in richness showed that changes in richness were most closely associated with agricultural land use, presence of alluvial deposits, and prevalence of the Mississippian geologic formation (Table 3). Intensive agriculture can adversely influence water quality, so the negative partial effect on change in species richness is not surprising. A bivariate plot of this partial effect (Fig. 4B) shows that species richness increased or was unchanged in watersheds where agricultural practices accounted for 25% of land use. Both alluvial deposits and the Mississippian formation enhance groundwater quantity and quality (Anderson 1998) and were associated with lowest rates of decline probably because they stabilize the hydrologic regime. The alteration of drainage in this agricultural area through channelization and subsurface drain tiling that accompanied wetland drainage has led to flashy hydrology that can decimate the stream biota (121).
- (Poole and Downing 2004) The combination of intense agricultural land use and low potential for water recharge (inferred by low fractions of Mississippian formations) results in suboptimal water quality and hydrologic conditions for mussels. The results of the watershed analysis underscored the positive influence of site-specific, nonagricultural, geomorphic features like wooded riparian zones (Table 3). Both results are consistent with other recent analyses (Hoggarth et al. 1995) in indicating that agricultural practices degrade mussel richness (122).
- (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).
- (Lewis et al., 2007) Clearly identifying the effects of urban land cover on streams can be difficult in the presence of other anthropogenic and certain natural influences. In particular, agricultural land use around the headwaters of a river can significantly modify stream chemistry upstream of an urban area, as is the case in the Connecticut (Douglas et al., 2002), Great Ouse (Neal et al., 2000), Seine (Roy et al., 1999), upper Rhine (Flintrop et al., 1996), and Pearl (Zhu et al., 2002) Rivers (304).
- (Lewis et al., 2007Cattle may be important sources of fecal bacteria to rural streams and rivers (Fernández-Alvarez, Carballo-Cuervo, de la Rosa-Jorge, & Rodríguez-de Lecea, 1991) (318).
- (Iwata et al., 2003) In contrast to a scarcity of ecological studies, deforestation impacts on stream hydrology have been well investigated in tropical rain forests of Borneo (Douglas et al. 1992, Greer et al. 1995, Malmer 1996, Chappell et al. 1999, Fletcher and Muda 1999), as well as in other Southeast Asian regions (see reviews by Douglas et al. 1993, Douglas 1999). These studies have revealed that deforestation associated with logging operations or agricultural development greatly increases rates of soil erosion and sediment supply to streams (462).
- (Iwata et al., 2003) Ecological impacts of such sustained anthropogenic disturbance [deforestation/slash and burn agriculture] on stream communities can be more severe than our findings (Ryan 1991, Waters 1995, Harding et al. 1998), and a full recovery of the communities may require several decades (471).
- (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).

- (Radwell and Kwak 2005) Reduced biotic integrity was found in other studies of midwestern United States lotic systems, with 36–84% of their watersheds in agricultural use (Roth and others 1996; Wang and others 1997). The watersheds in our study were much less disturbed, with forest cover ranging from 84% to 98% (808).
- (Radwell and Kwak 2005) High fish and invertebrate productivity, changes in fish assemblage structure, and lower invertebrate taxa richness in War Eagle Creek and the White River might reflect the influence of nutrient enrichment associated with land conversion from forest to pasture in these watersheds (808).
- (Gergel et al., 2002) Many studies have demonstrated that upland land use can influence riverine ecosystems. For example, nutrient losses from many agricultural catchments in the United States are consistently higher than from forested or grassland basins (e.g., Omernik et al., 1981; Johnson et al., 1997). In an upland catchment of the Calado floodplain along the Amazon, Williams et al. (1997) and Williams and Melack (1997) found large increases in solute mobilization from the upper soil horizons after cutting and burning in the catchment. Nutrient ratios in streams were altered from an N to P ratio of 120 :1 before deforestation to a ratio of 33:1 after deforestation (120).
- (Niyogi et al., 2007) Agricultural development alters streams in a variety of ways. Comparisons of streams in agricultural catchments with those in undeveloped forest catchments (reviewed in Allan, 2004; Quinn, 2000) have revealed a suite of changes, including alterations to incident light, stream temperature, and inputs of organic matter, nutrients, and sediment, as well as hydrology. These physicochemical changes, in turn, produce far-reaching effects on stream biota and ecosystem processes (e.g., see Quinn and others 1997; Richards and others 1996; Sponseller and others 2001; Townsend and others 1997; Wang and others 1997), but unraveling the differential responses to multiple stressors is a daunting task when so many physicochemical factors are involved (213).
- (Niyogi et al., 2007) Specific conductivity ranged from 14 to 175 lS/cm (Table 1) and was highly correlated ($r = 0.73$, $P < 0.01$, $n = 21$) with pastoral land cover in the catchments. The DIN ranged from 5 lg/L in tussock streams to 1797 lg/L in agricultural streams. Almost all DIN was nitrate-N; ammonium-N reached a concentration of 100 lg/L in the Wairuna catchment, but this was less than 15% of the total DIN. DRP ranged from 2 to 101 lg/L, and TP ranged from 2 to 242 lg/L. DIN, DRP, and TP concentrations in streams were positively related to the percentage of pastoral land use in the catchments (Figure 2) (217-219).
- (Niyogi et al., 2007) Physical measures of water and stream substrate were also related to land use. Cover of fine sediment on the streambed, TSS, and SIS were all positively related to the amount of pasture development in the catchments (Figure 2), as were turbidity and the degree of embeddedness (not shown) (219).
- (Niyogi et al., 2007) In multiple regressions (Table 2), the percentage of pastoral land cover in the catchment and stock access (as a categorical variable) explained most of the variation in TP concentrations (72%), DRP concentrations (80%), and fine-sediment cover (65%); riparian land cover did not explain significant variation in these variables after accounting for catchment land-use effects (219).
- (Niyogi et al., 2007) Epilithic chlorophyll-a on cobbles increased linearly with pastoral development in the catchment (Figure 3). Epilithic chlorophyll-a also increased with both the nutrient index and fine-sediment cover (219).
- (Niyogi et al., 2007) Density and biomass of benthic macroinvertebrates were significantly related, in a nonlinear manner, to pastoral land cover in the catchments (Figure 3) (219).
- (Niyogi et al., 2007) The EPT density had significant nonlinear relationships with pastoral land cover and the nutrient index, but no relationship with fine-sediment cover (Figure 4) (219).
- (Niyogi et al., 2007) As expected, both nutrient concentrations and fine sediment cover were related to pastoral land cover in our catchments and were correlated with each other (221).

- (Niyogi et al., 2007) The proportion of pastoral land use in catchments was related to most measures of stream health, as other studies have found (Quinn, 2000; Townsend and others 1997). As expected, streams in undeveloped catchments of tussock grasses had very low concentrations of nutrients and little fine sediment or embeddedness of substrates on the streambed. However, at the higher end of pasture development (>75% pasture), streams had a wide range of water quality and physical condition. Some of this additional variation was explained by stock access to streams (Table 2). Other factors related to agricultural intensity (fertilizer use, stocking intensity) probably accounted for additional variation in abiotic variables, and other studies on smaller catchments in the Otago region have used these data (Buck and others 2004) (221).
- (Niyogi et al., 2007) In particular, agricultural development can provide both subsidies and stresses to biodiversity, primary production, invertebrate and fish production, and other ecosystem processes (Niyogi and others 2003; Quinn, 2000; Riley and others 2003) (223).
- (Niyogi et al., 2007) We cannot propose a specific cutoff for agricultural development because many responses tended to be catchment-specific and would certainly be region-specific because of varying stressors such as temperature, erosion tendencies, and agricultural practices. For example, even within our study region, catchments with over 90% conversion to pasture could have a wide range of nutrient and sediment inputs to the stream. We can suggest that the prevention of fine-sediment inputs into streams, whether from pasture management, streamside fencing, or riparian restoration, has good potential to keep streams in this area in the positive area of the subsidy stress response curves (223).
- (Barker et al., 2006) Several studies have demonstrated strong associations between agricultural land use and alterations in stream habitats that caused compositional changes to stream communities (Lenat, 1984; Corkum, 1989; Quinn and Hickey, 1990; Roth *et al.*, 1998; Allan *et al.*, 1997), and by the late 1990s, the importance of large-scale land use and catchment characteristics as determinants of stream assemblages was recognized (2).
- (Barker et al., 2006) Regional models also identified land use as an important influence on fish IBI, but focused on specific types of agricultural land use (pasture in the Coastal Plain and % row crops in the Piedmont) (11-12).
- (Pringle 2001) Massive fish and bird mortality near the mouth of the Carson River in 1986 and 1987 (Rowe and Hoffman 1987) was attributed to bioaccumulation of elements within irrigation drainage (e.g., Dwyer et al. 1992, Lemly et al. 1993) (984).
- (Pringle 2001) By the early 1980s, scientists and agencies reported deformities in birds and massive die-offs of waterfowl and fishes at Kesterson that were linked to application of irrigation water to soils naturally rich in elements such as selenium (e.g., Ohlendorf et al. 1986) (984).
- (Pringle 2001) The potential for bioaccumulation of toxic elements in other arid regions is very high (McCully 1996) and may emerge as a threat to reserves in developing countries with expanding irrigation development (984).
- (Poole and Downing 2004) The significant relationships between changes in mussel biodiversity and residual streamside woodlands, and between changes in mean species richness of mussels and most intensive agricultural land use indicate that the extinction debt is most severe where habitat destruction has been the most complete. This finding echoes the results of terrestrial studies of biodiversity in both long- (Cowlishaw 1999) and short-lived (Brooks et al. 1997) organisms. Land use has been fairly static in this region for several decades (Fig. 1) and we observed the lowest rates of species loss in areas most similar to historic conditions. It thus appears that biodiversity may decline for decades following habitat alteration (122).
- (Poole and Downing 2004) Changes in large-scale watershed characteristics can affect community composition and environmental conditions (Frissell et al. 1986, Davies et al. 2000). Likewise, habitat degradation and fragmentation often drive declines in local and regional biodiversity (114).

- (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) 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).
- (Grau et al., 2003) Suspended sediments, an index of low-quality stream habitat, are significantly higher in first-order streams surrounded by pastures than in those surrounded by forests (Heartsill- Scalley and Aide 2003) (1165).
- (Grau et al., 2003) A detailed study showed that a watershed dominated by agriculture (the Río Grande de Loiza) yielded about 50% more landslides and suspended sediments than a forested watershed located in the Luquillo Experimental Forest (Larsen 1997) (1165-1166).
- (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) 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).
- (Grau et al., 2003) Relatively small changes in the type of land cover could have major effects on rates of soil erosion. For example, if only the 5% of the watershed with the highest erosion rates (bare soil, agriculture on steep slopes) is transformed into closed canopy forests, erosion in the watershed will decrease by 20%. If open woodlands are transformed through succession into closed-canopy forests, erosion will decrease by 7%. If instead the landscape is transformed into a mixture of pasture and agriculture, as it was during the first half of the 20th century, total basinwide erosion will increase between 33% (all pasture) and 103% (all agriculture) (1166).
- (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).

- (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).
- (Norton and Fisher 2000) In both basins, no significant relationships were identified between stream TP concentrations and land cover variables. However, the models for predicting stream N concentrations based on land cover for the Chester and Choptank basins exhibited very different results (Table 5). In the Choptank, cropland was highly correlated with annual average stream TN and NO3- (*r*2_0.69*** and 0.70***, respectively) (349-350).
- (Ducros and Joyce 2003) Land use in the Yorkshire catchment featured a high proportion of crops, which in this system was not rated highly for buffer zone effectiveness, but the landscape was also characterized by positive attributes, namely gentle slopes and few rills or gullies (Figure 1) (262).
- (Ducros and Joyce 2003) The BZIEF indicated that buffer zones likely to meet their water quality and habitat objectives were limited to catchments with a range of stream channel and bank features, diverse and established vegetation, wide riparian corridors, and soil hydrological responses characterized by slow subsurface flow in saturated, carbon-rich soils. In contrast, catchments with intensive agricultural land use, low vegetation diversity, freely draining soils, and steep, eroding slopes were less likely, according to the BZIEF, to deliver habitat or water quality benefits (265).
- (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).
- (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).

- (Niyogi et al., 2007) Greater degrees of eutrophication, however, can lead to algal blooms that are stressful to most animals by causing low dissolved oxygen and poor habitat quality. Thus, certain biological responses to catchment development might follow a subsidy-stress pattern (Allan, 2004; Odum and others 1979; Quinn, 2000) because of nutrient effects on algae (Figure 1). On the other hand, increasing sediment will likely act as a stressor with increasingly negative effects on most ecological responses (Figure 1). Suspended sediment can reduce water clarity, leading to light limitation of primary producers (Ryan, 1991), and mobile sediments limit stream algae and further reduce primary production (Biggs and others 1999; Schofield and others 2004). Sedimentation also negatively affects many animals through reduced physical habitat, a decrease in food quality, and possible damage to taxa with delicate gills and mouthparts (Angradi, 1999; Lenat and others 1981; Nerbonne and Vondracek, 2001; Rabeni and Smale, 1995). Overall, sedimentation can affect production and diversity of animals both by direct (i.e., reduced habitat) and indirect pathways (i.e., reduced food from primary production) (Townsend and Riley, 1999) (214).
- (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).