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Impacts of New Highways and Subsequent Landscape Urbanization on Stream Habitat and Biota

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New highways are pervasive, pernicious threats to stream ecosystems because of their short- and long-term physical, chemical, and biological impacts. Unfortunately, standard environmental impact statements (EISs) and environmental assessments (EAs) focus narrowly on the initial direct impacts of construction and ignore other long-term indirect impacts. More thorough consideration of highway impacts, and, ultimately, better land use decisions may be facilitated by conceptualizing highway development in three stages: initial highway construction, highway presence, and eventual landscape urbanization. Highway construction is characterized by localized physical disturbances, which generally subside through time. In contrast, highway presence and landscape urbanization are characterized by physical and chemical impacts that are temporally persistent. Although the impacts of highway presence and landscape urbanization are of similar natures, the impacts are of a greater magnitude and more widespread in the urbanization phase. Our review reveals that the landscape urbanization stage is clearly the greatest threat to stream habitat and biota, as stream ecosystems are sensitive to even low levels (<10%) of watershed urban development. Although highway construction is ongoing, pervasive, and has severe biological consequences, we found few published investigations of its impacts on streams. Researchers know little about the occurrence, loading rates, and biotic responses to specific contaminants in highway runoff. Also needed is a detailed understanding of how highway crossings, especially culverts, affect fish populations via constraints on movement and how highway networks alter natural regimes (e.g., streamflow, temperature). Urbanization research topics that may yield especially useful results include a) the relative importance and biological effects of specific components of urban development—e.g., commercial or residential; b) the scenarios under which impacts are reversible; and c) the efficacy of mitigation measures—e.g., stormwater retention or treatment and forested buffers.

Keywords road, urbanization, motorway, macroinvertebrate, fish, urban

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Introduction

Due to their large surface area, high traffic volume, and potential to induce urban development, the construction of large (≥ four-lane) paved roads (herein defined as highways), are often detrimental to local ecosystems. Stream ecosystems are particularly sensitive to the construction of new highways due to characteristics of the fluvial environment and biota. Downstream transport of water and sediment spreads chemical and fine sediment pollution, causing the ecological impacts of highways to extend farther in aquatic than in terrestrial environments (Forman and Alexander, 1998). Aquatic fauna often have a more difficult time avoiding spreading impacts than terrestrial fauna because their movements are generally confined to the narrow linear geometry of the stream channel. In addition, highways and urban development alter the hydraulic connection of streams to their watersheds, fundamentally altering processes, which control channel geomorphology, form habitat, and ultimately contribute to biotic integrity (Wang et al., 2001; 2003).

Angermeier et al. (2004) conceptualized the extent and nature of highway impacts on streams in three consecutive stages: initial highway construction, highway presence, and eventual landscape urbanization (Table 1). Because this framework reflects the spatial and temporal dimensions of impacts, it is useful for organizing, describing, and evaluating the environmental concerns of new highways. The initial phase, highway construction, includes all the short-term impacts from the construction process. These impacts are generally physical, temporary (i.e., subside through time), and local. The second phase, highway presence, encompasses secondary impacts that are chronically generated from the physical presence of the highway including chemical pollutants from automobile traffic and stream channel alterations. These chemical and physical impacts are regional and occur as long as the highway exists. Finally, landscape urbanization includes the impacts from general economic development and results in a variety of chemical and physical impacts that are widespread and chronic. Previous reviews have focused on single phases of highway impacts (Atkinson and Cairns, 1992; Little and Mayer, 1993; Forman and Alexander, 1998; Trombulak and Frissell, 2000; Forman and Deblinger, 2000; Paul and Meyer, 2001) but not clearly described or considered the inherent connectivity of the nature and scales of the impacts.

Table 1

Conceptual framework for primary physico-chemical impacts of highway construction. The scale and nature of primary environmental impacts change from the direct effects of highway construction to the secondary and indirect effects associated with the presence of a highway and urban development. These may be viewed as a gradient of changing concerns and impacts through time. All physico-chemical impacts have important consequences for stream biota but the degree of peer-reviewed investigation differs among stages

	Developmental stage		
Impact characteristics	Highway construction	Highway presence	Urbanization
Temporal extent	Temporary	Chronic	Chronic
Spatial extent	Local	Regional	Regional
Primary nature	Physical	Physical and chemical	Physical and chemical
Degree of investigation	Low	Moderate	High

The predictable effects of the three consecutive stages of new highway construction are seldom considered simultaneously in environmental assessments. The National Environmental Policy Act (NEPA), the Council on Environmental Quality (CEQ), and various state environmental laws (CEQ, 1997) require state and federal transportation agencies to consider the significance of anticipated impacts in environmental assessments (EAs) or prepare environmental impact statements (EISs) when significant impacts are anticipated. However, these assessments generally focus almost exclusively on short-term, localized impacts and ignore the long-term secondary and cumulative impacts (Spaling and Smit, 1993; McCold and Holman, 1995; Burris and Canter, 1997; Cooper and Canter, 1997; Angermeier et al., 2004) that are often primary concerns of the government agencies and civilian stakeholders reviewing these documents (e.g., NCWRC and NCDPR, 2002). Although European countries more rigorously apply ecological principles to transportation projects than the United States (Forman and Alexander, 1998), inadequate assessment of cumulative effects is a global problem (Cooper and Sheate, 2002).

Evaluations of the thoroughness of EISs and EAs are often limited due to a lack of published summaries of the impacts that may be expected from proposed projects. For example, evaluations of EISs and EAs have searched for the assessments for key words or concepts rather than assessing how meaningful and thoroughly probable impacts were considered (e.g., Burris and Canter, 1997; Cooper and Canter, 1997). A new review of the extent and nature of impacts from new highways that considers the stages and changing impacts identified by Angermeier et al. (2004) will assist reviewers of EISs and EAs in explicitly linking the successive stages, a step often ignored in assessment-proposed highway projects. Our review summarizes investigations that will help environmental and fisheries scientists consider potential impacts of proposed highway projects over multiple dimensions, but are often unavailable in field offices, and spread widely across academic disciplines. In addition EISs, EAs, and previous reviews often make assertions based on unpublished government reports which suffer the general inadequacies of "grey" literature (Collette, 1990). In contrast, we rely almost exclusively on published, peer-reviewed studies.

The purpose of this article is to review the impacts of new highways through undeveloped land. Although not the focus of this review, much of the information presented may be relevant to more common highway improvements, such as lane additions and surface upgrades. We focus on studies conducted in the United States, but some relevant international research is included to supplement sparsely researched topics. Following our conceptual framework (Table 1), we synthesize the scientific knowledge on physical, chemical, and biological responses of streams during 1) highway construction, 2) highway presence, and 3) watershed urbanization. The ultimate goals of this review are to provide information that will: 1) improve the ability of transportation planners to prepare more through, meaningful, and science-based EISs and EAs, and 2) spur research in subject areas where rigorous studies are lacking but information is needed for comprehensive impact assessment of new highways.

Highway Construction

Highway construction can be highly destructive to stream habitat and biota. Impacts on streams are primarily acute, local, and physical in nature (Table 1). Highway construction primarily degrades stream habitat locally but some of these impacts may transport downstream. In contrast to impacts of highway presence and landscape urbanization, many construction impacts may be temporary and streams can recover if not recurrently disturbed. In this section, we review literature assessing these acute impacts (e.g., sedimentation). Other impacts initiated during construction but posing long-term threats, such as channelization and culverts, are considered in the Highway Presence section.

Highway Construction and Physical Habitat

As with any earth-moving activity, the greatest threat of highway construction to streams is fine sediment pollution, which can cause a variety of problems for resident biota, including direct mortality, reduced reproductive success, and a reduction in the food base (reviewed in Waters, 1995). Fine sediment pollution originates as bare soil erodes into streams, usually after exposure to precipitation or flowing water. Streams impacted by highway construction accumulate (Clarke and Scruton, 1997) and transport (Weber and Reed, 1976) many times more sediment than undisturbed streams. Although a variety of erosion control procedures are available and often legally required, they are seldom evaluated for their effectiveness (but see Grace (1999), Benik et al. (2003a, 2003b)) and have a risk of failure seldom considered in published investigations or environmental impact statements. A Pennsylvania study found that, even in the presence of sediment control techniques, streams impacted by highway construction carried 5 to 12 times more fine sediment than a control stream (Weber and Reed, 1976). The suspended solids load of a Ontario stream increased from an average of 2.8 mg/l to 352.0 mg/l during the initial "clearing phase" and peaked at 1,390 mg/l during highway construction (Barton, 1977). In addition, Barton (1977) observed a 10-fold increase in fine sediment deposition following a highway construction channelization project but stream sediment loads approached preconstruction levels near the completion of the construction. Increases in suspended sediment are detectable for long distances (kms) downstream of construction sites (Wellman et al., 2000). These sediments deposit in downstream pools, riffles, and impoundments (Duck, 1985; Brookes, 1986).

Highway construction can result in a variety of other seldom studied physical habitat degradations by encroaching onto floodplains and damaging riparian areas. Heavy equipment accessing the stream may incidentally damage (Hubbard et al., 1993) or purposely remove (Stout and Coburn, 1989) riparian vegetation during highway construction. Riparian vegetation is a critical component of stream watersheds and performs many important functions for streams (see Urbanization section). Streams near highways are often channelized and the initial effect of heavy equipment modifying the stream channel may alter the dynamic equilibrium of streams and result in rapid channel reorganization, all of which can lead to additional sedimentation and erosion downstream.

Highway Construction and Stream Chemistry

We found no studies documenting chemical impacts of highway construction on streams. However, the use of heavy machinery in and around streams likely causes some chemical pollution. In addition, many highway construction materials are highly toxic to aquatic biota. For example, industrial waste materials and byproducts such as shredded tires, ashes, mining wastes, municipal sludge, and wood wastes may be used in highway construction (Eldin, 2002). These materials release heavy metals and hydrocarbons which are toxic to water fleas *Daphnia magna* and algae *Selenastrum capricornutum* (Eldin, 2002). The toxicity of these materials may be reduced when in contact with soil, and during typical construction these toxins are unlikely to reach detrimental levels in streams. Nevertheless, the proximity of toxic materials to streams increases the chances of accidental spills and releases.

Biological Effects of Highway Construction

The impact of highway construction on stream fishes and macroinvertebrates is rarely studied. Similar to other anthropogenic landscape changes, highway construction is difficult to research for several reasons. Highway construction consists of many individual impacts that occur concurrently; thus, specific causal mechanisms are difficult to establish. An additional obstacle to research is that construction timeframes are often unpredictable, and construction often takes longer than the tenure of a typical graduate student. In addition, highway construction presents statistical and study design difficulties; for example, treatments are difficult to replicate and meaningful controls difficult to establish.

We found only a few studies investigating the effects of highway construction on stream fishes and macroinvertebrates. However, fine sediment pollution occurs from a variety of anthropogenic sources and is widely studied outside the context of highway construction. The effect of fine sediments on stream biota has been recognized for decades (Ellis, 1936) and is the subject of many previous reviews (Chutter, 1969; Bruton, 1985; Ryan, 1991; Waters 1995; Wood and Armitage, 1997; Henley et al., 2000). Therefore, in addition to studies that directly focus on highway construction, this section includes more general investigations of the effect of fine sediment on stream biota.

Fine sediment pollution from highway construction can immediately alter macroinvertebrate and fish communities (Barton, 1977). Reductions in the abundance and diversity of macroinvertebrates may depend on the timing and duration of construction impacts (Cline et al., 1982). Stout and Coburn (1989) found an absence of macroinvertebrate shredders in pools below highway construction. Fine sediment from highway construction may result in reduced macroinvertebrate diversity and density (Lenat et al., 1981). Highway construction can immediately reduce the overall abundance of stream fishes by over 50% (Whitney and Bailey, 1959; Barton, 1977). Taylor and Roff (1986) reported that the abundance of bottom-feeding fishes is initially reduced, but recovers after fine sediment deposition rates decline. Fish and invertebrate communities begin recovering after the fine sediment loads are reduced and deposits wash downstream, but full recovery may require years (Taylor and Roff, 1986).

Fine sediment pollution degrades stream biotic communities through a variety of mechanisms. Stream periphyton and macrophytes are abraded, suffocated, and shaded by fine sediment (Waters, 1995). Fine sediment loads impact macroinvertebrates by inducing catastrophic drift (Culp et al., 1986), damaging individual's respiratory structures (Lemly, 1982), and reducing habitat by clogging interstitial spaces in streambeds (Lenat et al., 1981). Fine sediment can also clog the gills of fishes and reduce the quality of their habitats for feeding by impairing visibility and reducing prey abundance (Bruton, 1985). It is possible that construction interferes with a variety of feeding strategies; Berkman and Rabeni (1987) found that fine sediment deposition reduced populations of both insectivorous and herbivorous fishes. In addition, fine sediment suspended in the water can lower reproductive success of fishes (Burkhead and Jelks, 2001). For example, egg survival of some species depends on substrate that is permeable to water flow (Kondou et al., 2001).

Highway Presence

Although highway construction can be highly detrimental to stream habitat and biota, construction sites are sparse compared to the land covered and increasingly affected by existing highways. Currently in the United States, there are 6.3 million km of public roads, 60% of which are paved with a surface area of about 50,000 km^2 (Eldin, 2002). At present, 20% of the United States' land area is directly affected by road presence (Forman, 2000), and 50% is within 382 m of a road (Ritters and Wickham, 2003). In contrast to the localized, temporary effects of highway construction, the extensive effects of highway presence are persistently generated by highways with direct hydraulic connections to streams (Table 1).

Highways and Physical Habitat

Although pulses of highway runoff can substantially affect stream channels, we found no studies of its impact on physical stream habitat. However, many investigators have examined the impacts of logging roads (see Gucinski et al., 2001). Although unpaved forest roads are not the subject of this review and differ from paved highways in many aspects, they are similar in that their impervious surfaces collect stormwater and route runoff into streams. Collecting and routing runoff to streams causes logging roads to increase the magnitude and frequency of stream flooding (King and Tennyson, 1984; Jones and Grant, 1996). These runoff changes are also characteristic of urban areas and cause a variety of physical changes to stream channels, such as channel widening and downcutting (see Urbanization section). However, because paved roads are only minor components of the total impervious surfaces of an urban watershed, the presence of a single highway in a watershed likely results in less changes to flow regimes and, ultimately, less severe changes to physical stream habitat than urban development.

Streams near highways are often channelized during construction. However, unlike many construction impacts such as fine sediment pollution, this modification has continual long-term consequences for physical stream habitat. Channelization increases channel slope, reduces base flows, increases peak flows, alters substrate composition, and severs floodplain links (Hubbard et al., 1993). Overall, channelization reduces the habitat diversity characteristic of natural streams by replacing coarse substrates with finer substrates, reducing depth and velocity heterogeneity, creating more laminar flows, removing cover, and eliminating natural pool-riffle sequences (Peters and Alvord, 1964; Narf, 1985).

If engineered properly, bridges may cause minimal impacts on the physical stream channel; however, through channelization or poor construction practices, bridges can destabilize stream channels. Although culverts are generally more detrimental to stream habitat and biota, they are often installed as a cheaper alternative to spanning structures. The presence of cluverts destabilizes stream channels by interrupting the downstream transport of woody debris, sediment, substrate, and water. Although few quantitative studies of the impact of culverts on physical stream habitat are available, Gubernick et al. (2003) provided a qualitative overview. Unlike dynamic natural stream channels, culverts are rigid and unaccommodating to changes in channel morphology. In addition, the stream channel is often widened above the culvert, reducing current velocities and forming a sediment trap. Although downstream sediment flow is reduced above the culvert, it continues or accelerates below the culvert causing channel downcutting and resulting in an elevation drop, even if initial construction put the pipe at stream level. Typically, culverts are sized to accommodate rare flood flows but are too small to allow passage of woody debris. Accumulations of woody debris near the inlet can starve downstream areas of this important component of stream habitat (see Urbanization section) and may plug the culvert, causing failure of road fill during floods and increasing the risk of catastrophic debris torrents.

Highways and Stream Chemistry

Highway surfaces collect a variety of chemical pollutants from automobile traffic and are disproportionate contributors to overall pollutant loads. For example, public highways cover 8% of Rhode Island, but produce 16% of the state's oil and grease pollution, and 77% of the state's zinc pollution (Hoffman *et al.*, 1985). These pollutants are mobilized by runoff water and transported to streams where they accumulate in sediments and biota and spread downstream, resulting in chronic and widespread effects. This runoff represents an important, but relatively unstudied, component of stream pollution (Wu et al., 1998).

Traffic residue adds a variety of metals to highway runoff, including iron, zinc, lead, cadmium, nickel, copper, and chromium. Tires contain up to 1% zinc by weight (Hedley and Lockley, 1975) and are a significant source of zinc in the environment (Davis et al., 2001). Brake pad dust contributes copper (Davis et al., 2001). These metals accumulate in roadside dust (Leharne et al., 1992), soil (Goldsmith et al., 1976; Garcia-Miragaya et al., 1981), and stream sediments (Van Hassel et al., 1979; Maltby et al., 1995b). The concentrations of metals in stream sediments are positively related to the volume of traffic (Van Hassel et al., 1980; Callender and Rice, 2000) and accumulate in proportion to the length of highway drained (Maltby et al., 1995b), suggesting that pollution will be most severe when large highways are drained by small streams.

Highway surfaces also accumulate petroleum from automobile traffic. Motor oil accumulates from crankcase drippings, washes off the highway surface, and accumulates in stream sediments (Hoffman et al., 1985). Until the Clean Air Act of 1970 phased out leaded gasoline, lead was the most widespread metal pollutant from automobile traffic. Unleaded gasoline permits the use of catalytic converters, which convert gaseous exhaust pollutants such as carbon monoxide, nitrous oxides, and hydrocarbons to less toxic chemicals such as carbon dioxide, nitrogen, and water. The chemical reactions are catalyzed in automobile exhaust systems by platinum group elements (PGEs), including platinum, palladium, and rhodium, which are emitted on highway surfaces during operation. Since the introduction of catalytic converters, PGEs have become a new and relatively unstudied metal pollutant of stream sediments (Rauch and Morrison, 1999). In addition, iridium, rubidium, and osmium are common impurities in PGE catalysts and may also be deposited on highways (Rauch et al., 2004). Concentrations of PGEs in roadside soils are related to traffic volume and are increasing to such a degree that their recovery (i.e., mining roadside soil) may become economically viable (Ely et al., 2001).

In areas that undergo winter weather, deicing salt is another widespread, but little studied, chemical pollutant of streams. Deicing salt is spread on highways in anticipation of and during snow and ice accumulation, from where it washes directly into streams or is stored in the soil. A study in Pennsylvania found 20- to 30-fold increases in a stream's conductivity during winter thaws (Koryak et al., 2001). Although concentrations harmful to fish are considered rare (Transportation Research Board, 1991), few studies have addressed the effects of these "shock loads" of salt on stream biota. Koryak et al. (2001) observed only pollution-tolerant macroinvertebrates and stressed fish communities in areas receiving shock loads of deicing salt. Furthermore, deicing salt may be contaminated by metals and nutrients. Phosphorous, lead, and zinc were found in highway deicing salt and anti-skid sand in Minnesota (Oberts, 1986) and iron, nickel, lead, zinc, chromium, and cyanide in deicing salt in England (Hedley and Lockley, 1975). Road salt that does not run off directly into streams may still cause chronic problems through slow release into adjacent soils; chlorine ions from road salt have a soil residence time of at least 2 years (Mason et al., 1999).

Another concern associated with the presence of a highway is the inevitability of toxic chemical spills. In 1982, hazardous materials made up more than 25% of all domestic freight shipments (List and Abkowitz, 1986). Almost all types of hazardous wastes and 62% of all hazardous materials (by weight) are moved by truck (Abkowitz et al., 1989; Atkinson and Cairns, 1992). Unfortunately, accidental releases during shipping are not infrequent.

Between 1990 and 1994 an average of 10,000 accidents per year were reported, releasing 2,445 kl of hazardous materials annually on U.S. highways (USEPA, 1996).

Biological Effects of Highways

Highways have many detrimental effects on stream biota. Toxic chemical spills often occur from truck accidents, and can cause fish kills extending downstream for great distances (kms). Stream crossings may be especially vulnerable to spills because bridge surfaces encourage automobile accidents because during winter weather conditions by icing more frequently than terrestrial paved surfaces. Furthermore, the inherent vicinity of bridge accidents to streams increases that risk that spilled chemicals may enter streams before containment. Accidental spills are particularly devastating for isolated populations of rare species with limited potential for movement and recolonization, such as freshwater mussels. Although there are many documented cases of such acute effects (USEPA, 1996), we found no studies describing chronic changes in macroinvertebrate or fish communities resulting from repeated toxic spills. Studies examining streams after catastrophic toxic spills have documented eventual recovery and recolonization from adjacent areas (Ensign et al.,1997; Meade, 2004), emphasizing the importance of well-connected habitats to increase resilience of stream biota to the effects of highway presence. Thus, stream reaches that are isolated by culverts, dams, or natural barriers may be particularly vulnerable to spills.

Macroinvertebrates and fishes near highways may have elevated metal concentrations in body tissues. Levels of lead and zinc in fishes and aquatic macroinvertebrates may be locally related to the amount of traffic at upstream highway crossings (Van Hassel et al., 1980) and regionally related to highway densities across large areas (Stemberger and Chen, 1998). Fish species accumulate metals from highway runoff at differential rates (Ney and Van Hassel, 1983). Aquatic macroinvertebrates may absorb platinum from stream sediments (Rauch and Morrison, 1999). The accumulation of toxic chemicals in animal tissue likely results in widespread impacts that spread to terrestrial communities, particularly animals that feed exclusively on aquatic species (e.g., many members of the avian order Ciconiiformes).

Many components of highway runoff such as metals and petroleum are suspected toxicants to aquatic organisms. Although few studies have addressed the toxicity of highway runoff, the sediment from contaminated streams is considered more toxic than the water. Although a variety of potential toxicants, including hydrocarbons, copper, and zinc are found in highway runoff, polycyclic aromatic hydrocarbons (PAHs) in stream sediments may be responsible for the majority of macroinvertebrate toxicity (Maltby et al., 1995a). Boxall and Maltby (1997) confirmed that three specific PAHs, pyrene, fluoranthene, and phenanthrene, were major sediment toxicants for *Gammarus pulex* and accounted for >30% of the toxicity of runoff-contaminated sediments.

Comparisons of macroinvertebrate communities above and below highway crossings are rare but indicate that reductions in diversity and pollution-sensitive species below highway crossing are most pronounced where small streams receive runoff from large sections of highways (Maltby et al., 1995b). These patterns may reflect greater hydrocarbon pollution in sediments below road crossings. Reductions in pollution-sensitive shredders may result in slower leaf litter breakdown (Maltby et al., 1995b), altering stream productivity, nutrient cycling, and food webs.

In addition to chemical effects, highways also impact biota through physical changes to the stream channel. Channelization can have numerous effects on the physical structure and natural environmental regimes of stream systems; these dynamics provide a mosaic of habitats to support resident organisms (Stanford et al., 1996; Poff et al., 1997; Poole, 2002).

Moyle (1976) compared channelized and unchannelized sections of a California stream and found the biomass of fish and invertebrates in channelized locations was less than one-third of that in unchannelized locations. He also found differences in fish and macroinvertebrate species composition between channelized and unchannelized areas. Channelization may reduce the recruitment and production of fishes by eliminating nursery habitat. For example, removal of gradually sloping streambanks increases the area of unsuitable habitat with velocities greater than the swimming speeds of age–0 fishes (Copp, 1991, 1997; Scheidegger and Bain, 1995; Mann and Bass, 1997; Mérigoux and Ponton, 1999; Meng and Matern, 2001).

Culverts are a feature of highway presence that can have a variety of negative impacts on stream biota. Culverts provide poor internal habitat due to low-bottom complexity and uniformly high-flow velocities inside culverts provide poor habitat (Slawski and Ehlinger, 1998), but most importantly, they are notorious fish movement barriers. The effects of highway crossings on stream fish movement depend on the swimming speed and behavior of individual species (Toepfer et al., 1999). Fish passage is obstructed by high current velocities and shallow depths inside culverts, as well as vertical drops at the culvert outflow (Baker and Votapka, 1990). In addition, concrete box culverts may develop internal gravel bars (Wellman et al., 2000) that impede fish movement. Warren and Pardew (1998) found that overall fish movement was an order of magnitude lower through culverts than through other crossing types or natural channels in small, warmwater Arkansas streams. Culverts throughout a tributary network can reduce production of species that require spawning migrations, such as coho salmon *Oncorhynchus kisutch*, by preventing adults from reaching spawning habitat (Beechie et al., 1994). Barriers can isolate populations, resulting in reduced genetic diversity and increased probability of extinction due to demographic instability and impeded recolonization. Most investigations of fish movement barriers have focused on economically important fishes with known migration patterns; for example, Belford and Gould (1989) determined combinations of water velocity and culvert length that prevented passage by brook trout *Salvelinus fontinalis*, rainbow trout *Oncorhynchus mykiss*, and brown trout *Salmo trutta*. However, entire fish communities are vulnerable to highway crossing movement barriers (Jackson, 2003) and the importance of movement and movement barriers to nongame fishes and fish communities is poorly understood. In one of the few published studies for a nongame species, Schaefer et al. (2003) found that a variety of culvert types significantly decreased the probability of movement of the federally threatened leopard darter *Percina pantherina* between habitat patches. Although culverts present a variety of obstacles to fish movement, engineers designing passable culverts may narrowly focus on the effects of singular parameters such as vertical outflow drop distance or current velocity (e.g., Rowland et al., 2003) and not consider the cumulative effects of multiple passage inhibiting features.

Urbanization

Urbanization is difficult to define, as the meaning of "urban" varies across disciplines (Paul and Meyer, 2001). We modify the definition by Kemp and Spotila (1997) and define urbanization as development in a watershed, such as building construction, that changes land use typical of rural areas (e.g., farming, grazing) to uses more typical of residential and industrial areas (e.g., retail, suburban residential areas, plants and factories). This definition describes the general process of watershed-altering development that is characteristic of the urban landscape and the focus of this review.

The construction of new highways is the "quintessential public sector investment" by which government attempts to encourage economic growth in rural areas (Chandra and Thompson, 2000). At the state level, new highways are ineffective at increasing economic activity (Evans and Karras, 1994; Holtz-Eakin, 1994; Dalenberg and Partridge, 1997), but they effectively redistribute economic activities among locales. New highways reduce traditional rural economic activities of nearby counties such as agriculture, but enhance and concentrate urban economic activities such as manufacturing and retail in the county the highway intersects (Rephann and Isserman, 1994; Chandra and Thompson, 2000). The construction of new highways also increases the price of agricultural land (Shi et al., 1997), thereby encouraging development. Shifting the economic productivity from rural to urban activities promotes urbanization, at least in the county the highway passes through, as reduced travel time to cities encourages establishment of commerce in previously undeveloped areas. Although final decisions for urban development are made by local governments, new highways clearly and purposely provide impetus for urban development.

Much of our understanding of the relation between new highways and urban development is based on patterns observed at interstate highway exits. Key characteristics of interstate exits related to the rate and nature of urban development in North Carolina include traffic volume of the interstate and crossroad, location, and population of nearby communities, distances to urban centers, degree of preexisting development, and distance to the next interchange (Hartgen et al., 1992). A later study concluded that these relationships were consistent for interstate exits nationally (Hartgen and Kim, 1998). Hartgen et al. (1992) described the general requirements, stages, and potential paths of interchange development, and all potential paths predict that interstate construction leads to the conversion of forested and agricultural areas to commercial or residential development. Improvements to existing highways, such as lane additions, also increase development activity along the highway corridor (Cervero, 2003).

Other studies have documented land use change induced by the presence of nearby highways. Although these studies do not address the construction of new highways, they provide insight into the relationship between a new highway and landscape urbanization at points other than exits. Bradshaw and Muller (1998) forecasted the conversion of farmland to urban areas in California, describing highways between cities as "magnets for decentralized growth." In the southern Appalachian Mountains, areas close to highways were more likely to experience development (Wear and Bolstad, 1998).

Although the relation between the construction of a new highway and urban development is intuitive, predictable, and often a declared political goal, few investigators have examined this connection. Indeed, this connection is a contentious issue for transportation planners. The position that new highways do not result in urban development, although apparently ubiquitous among transportation planners, is not well-supported by published studies and our literature review failed to produce any peer-reviewed examples refuting a positive relation between highway construction and urban development (but see Hartgen (2003a, 2003b)).

A stream's physical habitat and chemical environment are largely products of its watershed. Thus, as a watershed urbanizes, changes occur in stream habitat, water chemistry, and ultimately biota. Similar to the presence of a hydraulically connected highway, urban development continually affects streams and causes extensive and chronic impacts (Table 1), but at greater magnitudes. Runoff from urban areas contains all the chemical pollutants from automobile traffic as well as those from urban sources. In addition, urbanization drastically alters how a watershed produces stream flow, resulting in many changes in physical habitat.

Urbanization and Physical Habitat

Undeveloped watersheds are characterized by land surfaces that are pervious to precipitation. Rain falling in undeveloped watersheds infiltrates the soil and reaches streams slowly as subsurface flow. The urban landscape, however, is characterized by rooftops, asphalt, compacted soils, and other highly impervious surfaces (Schueler, 1994, 1995). These impervious surfaces with direct hydraulic connections to streams (Schueler, 1994; Wang et al., 2001, 2003), capture precipitation and route it quickly into storm sewers and gutters and, ultimately, into streams (Hollis, 1975). Similarly, precipitation falling on impervious surfaces without direct hydraulic connections to streams may reach streams quickly as overland flow (Horton, 1945; Leopold, 1973). Thus, urbanization fundamentally alters the delivery of water to streams (Booth, 1991).

These changes in precipitation delivery alter stream flow regimes. As a watershed urbanizes, peak flow volume from precipitation events increase (Hollis, 1975; Beard and Chang, 1979; Neller, 1988; Booth, 1990; Clark and Wilcock, 2000; Rose and Peters, 2001), thereby increasing the frequency of bankfull flows (Leopold, 1973; Hollis, 1975; Arnold et al., 1982; Moscrip and Montgomery, 1997). Even low levels of paving increase the magnitude of frequent floods (recurrence interval $\leq$ 1 year); for example, paving 20% of the watershed can increase the peak discharge of the mean annual flood by an order of magnitude (Hollis, 1975). Thus, discharge rates that previously occurred once every 2 years may double in frequency following watershed development (Booth, 1991). Ten-year floods may occur 2.5 to 10 times more frequently following watershed urbanization (Moscrip and Montgomery, 1997). In addition, precipitation events that produced no increase in stream flow prior to urbanization may generate substantial flooding following watershed urbanization (Booth, 1991).

These changes in flood frequency and magnitude result in a variety of changes to physical features of streams. Bankfull and greater flows cut and form stream channels and adjust channel capacity such that bankfull conditions occur on an average of once every 1 to 2 years (Wolman and Miller, 1960; Leopold, 1973). The increased frequency of bankfull flows following urbanization causes a stream to increase its channel capacity by eroding its banks, downcutting its channel, or both (Hammer, 1972; Leopold, 1973; Arnold et al., 1982; Allen and Narramore, 1985; Booth, 1990, 1991; Gregory et al., 1992; Pizzuto et al., 2000). Thus, urban streams are wider and deeper than unaffected channels.

Impervious surfaces increase peak flow at the expense of base flow. Base flows result from subsurface flow and groundwater that steadily contributes to streams between precipitation events. Because impervious surfaces prevent precipitation from infiltrating below the surface, urban streams are characterized by low base flows (Simmons and Reynolds, 1982; Wang et al., 2001, 2003). Low flows combined with the effects of channel enlargement, results in urban streams that feature oversized stream channels with little water between runoff events.

Streams in urbanized watersheds enlarge their channels by eroding their banks. Bank erosion as well as runoff from urban construction activities adds fine sediment to the receiving stream (Waters, 1995; Trimble, 1997). Typically, fine sediment is a minor component of pristine streams. For example, a stream flowing through a completely forested watershed receives about 11.3–33.8 metric tons per ha of sediment annually; in contrast, an urbanizing watershed may receive more than 226,000 metric tons per ha annually (Wolman, 1967; Wolman and Schick, 1967). This dramatic increase in fine sediment can devastate, and ultimately, extirpate stream biota (see Highway Construction section). Channel enlargement may be balanced by rapid loading of fine sediment during the initial phases of urbanization. For example, Wolman (1967) hypothesized that high sediment loads from the construction phases of urbanization could temporarily clog and constrict stream channels, a phenomenon later observed by Leopold (1973).

When the extent of urbanization in a watershed stabilizes, stream channel enlargement may cease, and the channel banks may restablize. In addition, as the rate of urban development declines, fine-sediment loading may be greatly reduced as construction site soils are stabilized via revegetation or pavement, and prior deposits may be removed by scouring during subsequent flooding (Wolman, 1967; Clark and Wilcock, 2000; Finkenbine et al., 2000). However, the process of bank erosion, downcutting, and channel adjustment may continue for several decades, and some streams never stabilize (Henshaw and Booth, 2000).

Urbanization typically results in loss of streamside (riparian) vegetation as areas near streams are cleared. The degree of riparian disturbance varies with type of urban land use. For example, Thibault (1997) found the land used for transportation, schools, and industry had more intact riparian areas than residential, commercial, and recreational land. Riparian vegetation is a critical component of the watershed (reviewed by Karr and Schlosser, 1978; Gregory et al., 1991; Naiman and Décamps, 1997; Pusey and Arthington, 2003) and, although they cover a small percentage of the watershed, riparian areas are disproportionately important for stream health. Intact riparian areas absorb and filter out metals, fine sediment, and nutrients from overland runoff (Castelle et al., 1994) and generally mitigate the physical and chemical effects of urbanization (May et al., 1997). Riparian vegetation stabilizes streambanks and reduces bank erosion (Whipple et al., 1981; Beeson and Doyle, 1995; Finkenbine et al., 2000), and helps moderate urban stream temperatures (LeBlanc et al., 1997).

Riparian vegetation contributes leaves, wood, and other organic debris to streams. The biota of small (≤ fourth-order) streams, such as those generally associated with urban areas (Heaney and Huber, 1984), depend on leaves and organic inputs as their energy base (Vannote et al., 1980; Hawkins and Sedell, 1981). Large woody debris is an important component of stream channels because it stabilizes stream banks (Keller and Swanson, 1979; Booth, 1991; Gregory et al., 1991; Finkenbine et al., 2000), creates pools (Keller and Swanson, 1979; Larson et al., 2001), and provides habitat for macroinvertebrates (Benke et al., 1985) and fishes (Angermeier and Karr, 1984; Flebbe and Dolloff, 1995). In urban areas, recruitment of woody debris declines as development removes floodplain trees and instream abundance is typically reduced by intentional debris removal (Larson et al., 2001).

Stream water temperature is a major determinant of the distribution and abundance of aquatic biota and is primarily regulated at two spatial scales, the riparian and the watershed. Riparian vegetation shelters streams from warming by absorbing or reflecting sunlight before it reaches the water. Thus, loss of riparian vegetation contributes to the warming of urban streams (Barton et al., 1985; LeBlanc et al., 1997). At the watershed scale, impervious surfaces, especially parking lots, collect and heat runoff water before it reaches streams. For example, Van Buren et al. (2000) developed a computer model to predict runoff temperature and observed that a parking lot produced runoff 5.9°C warmer than summer rainfall. The maximum daily water temperature in Wisconsin and Minnesota streams increase by 0.25°C with every 1% increase in the impervious area of the watershed (Wang et al., 2003). In addition to increases in average water temperature, urban streams exhibit increased temporal variability (Moglen et al., 2004).

Urbanization and Water Chemistry

Urban runoff contains a variety of chemical pollutants including petroleum, metals, and nutrients. Rivers and streams receive the majority of urban runoff (84%) (Heaney and Huber, 1984) and chemical pollutants are often stored in stream sediments. House et al. (1993) reviewed the constituents and impacts of urban runoff on receiving waters.

Oil and grease enter urban runoff from a variety of sources including deliberate dumping, automobile engine emissions, and chemical spills; however, the majority originates from automobile crankcase drippings (Hoffman et al., 1982). Parking lots accumulate oil and grease deposited by parked vehicles and become the primary land use source of oil and grease in urban runoff. Stenstrom et al. (1984) observed concentrations of oil and grease up to 15 mg/l in parking lot runoff. Automotive sources of metals in urban runoff include zinc from tire wear (Hedley and Lockley, 1975) and motor oil (Davis et al., 2001), platinum from catalytic converter emissions (Rauch and Morrison, 1999), and lead from motor oil (Davis et al., 2001).

In addition to automotive sources, urban runoff accumulates metals from a variety of other sources. For example, iron originates from the corrosion of steel (Characklis and Wiesner, 1997), zinc from the corrosion of galvanized metals (Hedley and Lockley, 1975), roofing, and painted wood (Davis et al., 2001), and lead from brick and painted surfaces (Davis et al., 2001). Other metals in urban runoff include chromium and nickel (Klein et al., 1974; Helsel et al., 1979; Rhoads and Cahill, 1999). Metals from urban runoff accumulate in stream sediments (Garie and McIntosh, 1986; Rauch and Morrison, 1999), where concentrations are related to both population and traffic densities (Callender and Rice, 2000).

Urban runoff is high in nutrients such as nitrogen and phosphorous that can result in detrimental algal blooms and decreased dissolved oxygen levels. Nutrient levels in streams are typically predictable from land use (e.g., Herlihy et al., 1998). For example, the risk of nutrient pollution increases as nonforest land cover reaches 10% of the watershed (Wickham et al., 2000). Historically, nutrient pollution has been associated with agricultural land use, but urban land often produces greater nutrient loading. For example, concentrations of total phosphorous and total nitrogen in urban streams were 2 to 10 times higher than agricultural and forested streams in Missouri (Smart et al., 1985). Other studies have reported higher concentrations of nitrogen and phosphorous in urban streams than in agricultural and forested streams (Osborne and Wiley, 1988; Wahl et al., 1997).

Biological Impacts of Urbanization

Altered and impaired biotic communities are characteristic of urban streams. Urban macroinvertebrate communities have reduced taxa richness (Garie and McIntosh, 1986; Jones and Clark, 1987; Kemp and Spotila, 1997), reduced density (Garie and McIntosh, 1986), lower index of biotic integrity (IBI) scores (Steedman, 1988; Kennen, 1999), lower functional diversity (Pedersen and Perkins, 1986), and lower taxonomic diversity (Pratt et al., 1981; Shutes, 1984; Pedersen and Perkins, 1986). In an extensive survey of New Jersey streams, Kennen (1999) found that locations with severe macroinvertebrate community impairment were most commonly downstream from urban areas. Urbanization reduced taxa-diversity and richness by reducing the density of pollution intolerant taxonomic orders (Ephemeroptera, Coleoptera, Megaloptera, and Plecoptera) and increasing the density of pollution tolerant Diptera in Virginia streams (Jones and Clark, 1987). Macroinvertebrate diversity may decline progressively as streams flow through urban areas (Pratt et al., 1981).

Macroinvertebrate diversity was reduced to taxa-tolerant of physical disturbances in an urban Washington stream (Pedersen and Perkins, 1986).

Fish communities are similarly impaired by urbanization. Urban stream fish communities have lower overall abundance (Weaver and Garman, 1994; Albaneese and Matlack, 1998; Wang et al., 2000, 2003), diversity (Tramer and Rogers, 1973; Klein, 1979; Scott et al., 1986; Weaver and Garman, 1994; Onorato et al., 2000; Wang et al., 2000), IBI scores (Schleiger, 2000; Wang et al., 2003), taxa richness (Weaver and Garman, 1994; Albanese and Matlack, 1998; Schleiger, 2000; Wang et al., 2000), darter species richness (Weaver and Garman, 1994; Albanese and Matlack, 1998; Schleiger, 2000), and darter abundance (Kemp and Spotila, 1997; Onorato et al., 2000), and are dominated by pollution tolerant species (Wichert, 1994, 1995; Kemp and Spotila, 1997; Albanese and Matlack, 1998; Wang et al., 2003). Lead content in fish tissue is higher in urban areas (Stemberger and Chen, 1998). Furthermore, the proximity of urban streams to humans increases the risk of nonnative species introduction and establishment.

Although many studies describe the alteration of stream macroinvertebrate and fish communities by urbanization, the mechanisms linking specific urban impacts to specific community responses are largely unknown. Since multiple chemical and physical impacts of urbanization occur simultaneously, it is difficult to determine how specific environmental stresses affect biotic communities. However, changes in physical habitat likely impacts biotic communities more than changes in water chemistry. For example, fish and macroinvertebrate communities become impaired at the onset of urbanization (Klein, 1979), when physical changes are more prevalent than water chemistry changes. Most water chemistry changes are not detectable until urban land cover exceeds 40% of a watershed (May et al., 1997).

Threshold Effect of Urbanization

In the last 100 years, the field of stream ecology has expanded its spatial focus from small habitat patches to entire watersheds (Miranda and Raborn, 2000). Consistent with this paradigm shift and advances in geographic information systems and remote sensing, recent studies have addressed how different spatial configurations of urbanization affect stream communities. For example, investigators have documented relations between percent urban land cover (ULC) (Steedman, 1988), percent impervious area (Klein, 1979; Booth and Jackson, 1997; Wang et al., 2000), percent impervious area with direct connections to streams (Booth and Jackson, 1997; Wang *et al.*, 2001, 2003), and biotic parameters. These studies overwhelmingly conclude that very low levels (8–10%) of ULC (or surrogate measures) result in highly altered fish and macroinvertebrate communities. Even after this low level of development, successful restoration of these communities back into preurban conditions may be near impossible, as this small change could result in a shift into a new, less desirable, stable state that is difficult to reverse (Mayer and Rietkerk, 2004).

Initial watershed urbanization following the construction of a new highway is more damaging to stream ecosystems than later, more extensive, development. In macroinvertebrate and fish communities, pollution- and stress-tolerant species rapidly replace intolerant species as ULC approaches 10%. After ULC exceeds 10%, further increases result in little or no fish community changes (Schueler, 1994; Booth and Jackson, 1997; Wang et al., 1997, 2000, 2001). For perspective, 10% ULC is characteristic of areas typically considered "suburban" rather than "urban" (Wang et al., 2000). Although agriculture can have similar effects, streams may support relatively healthy fish communities until agricultural land cover exceeds 80% of the watershed (Wang et al., 1997). Because fish communities in currently undeveloped or agricultural watersheds are likely to be severely degraded by the onset of urbanization (Wang et al., 2000), protection against urbanization impacts should focus on watersheds where urbanization has not yet begun (May et al., 1997). In the context of highway impacts, this means that the greatest damage to stream health is inflicted by building new highways through undeveloped watersheds, which, ultimately, become subject to urban sprawl.

Conclusions

The short-term environmental consideration of transportation projects in EISs and EAs focuses on the initial construction impacts. However, the most serious threats to stream ecosystems are the long-term secondary effects of a highway's presence in the watershed and the cumulative effects of urban development. For example, the biotic integrity of streams in undeveloped (primarily forested or agricultural watersheds) is substantially degraded by the onset of urbanization, thus, streams in undeveloped watersheds are more sensitive to the construction of new highways than streams in urban watersheds. Because many aquatic impacts from the existence of the highway and urban development are long-term considerations, the narrow, short term focus of EISs and EAs provides inadequate protection for stream ecosystems. As new highways continue to diminish the percentage of the landscape that is unaffected by roads, expanding the spectrum of environmental impacts considered for highway projects is increasingly important.

Highway construction and highway presence impose a variety of impacts on stream habitat and biota. Urban development results from the construction of new highways and is clearly the most pernicious threat, as stream habitat and biota are sensitive to even low levels ($<10\%$) of development in a watershed. Watershed urbanization is a predictable indirect or secondary effect of the construction of new highways and NEPA, the CEQ, and various state environmental laws require consideration of indirect and cumulative effects in EISs and similar documents (CEQ, 1997). Although secondary and cumulative impacts are often important considerations of environmental agencies that comment on such assessments (e.g., NCWRC and NCDPR, 2002), landscape urban development resulting from the construction of new highways is generally ignored by the transportation agencies preparing the assessments. The importance of considering the impacts of landscape urban development during initial planning is amplified because this is the final opportunity to consider all effects cumulatively. Landscape urbanization ultimately results from the "tyranny of small decisions" (Odum, 1982) on many individual projects, the cumulative impacts of which are overlooked by the Clean Water Act section 404 permitting process (Stein and Ambrose, 2001), as well as other regulatory mechanisms.

Given the severity and extent of highway impacts on stream biota, we were impressed by the paucity of peer-reviewed literature on many aspects of those impacts. We believe the lack of published studies demonstrates a failure of both management agencies and academic researchers to address a severe and politically thorny environmental issue. Well-designed descriptive studies, in addition to conceptual or theoretical investigations, could contribute substantially to how society views and manages highway impacts. We urge scientists, managers, and policymakers to cooperate more closely to generate comprehensive knowledge about how highways affect ecosystem operation, make that knowledge available to the public (e.g., in EISs and EAs), and apply that knowledge to policy decisions regarding development of sustainable transportation systems.

Although highway construction is ongoing, pervasive, and has severe biological consequences, we found few published investigations of its impacts on streams. We encourage environmental and fisheries scientists to pay closer attention to the effects of new highway construction or highway improvements on streams. Carefully designed, comparative investigations could contribute substantially to our understanding of the differential impacts of various construction techniques, as well as the efficacy and risk of failure of various mitigation practices.

There are many unexploited opportunities to investigate the impacts of highway presence on stream biota. Researchers know little about the occurrence, loading rates, and biotic responses to specific contaminants in highway runoff. Understanding the dynamics and roles of specific pollutants could facilitate more effective mitigation. Future investigation should address the relative importance of chronic pollution, such as metals accumulated in stream sediments, versus acute impacts such as pulses of petroleum and deicing salt. Additional research is also needed to understand how highway crossings, especially culverts, affect fish populations via constraints on movement and how highway networks alter flow regimes of watersheds.

Impairment of stream biotic communities due to urbanization is severe and widely studied. However, opportunities still exist for relatively simple descriptive investigations. For example, we are impressed but the paucity of studies addressing stream thermal pollution from urban runoff and reduced riparian areas. In addition, techniques for minimizing impact or restoring biotic integrity are poorly developed. Research topics that may yield especially useful results include a) the relative importance and biological effects of specific components of urban development: e.g., highway, commercial, or residential, b) the scenarios under which impacts are reversible, and c) the efficacy of mitigation measures: e.g., stormwater retention or treatment and forested buffers. Finally, comprehensive risk analyses that incorporate both social and biotic components are badly needed to examine potential for catastrophic events during all phases of new highway impacts. Risks include mitigation failures and catastrophic spills during the highway construction, presence, and urbanization phases. Depending on the nature of the biotic community (e.g., is it isolated, is the stream small, does it contain sessile species), it may be more or less vulnerable to these kinds of events. Without a spatially explicit, rigorous risk analysis framework, managers cannot properly weigh the risks and benefits of road projects proposed in their areas and have no scientific basis for proposing alternatives that may be less damaging to stream ecosystems.

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