Recovery Potential Metrics Summary Form

Indicator Name: HYDROLOGIC ALTERATION

Type: Stressor Exposure

Rationale/Relevance to Recovery Potential: Several different forms of hydrologic alteration (i.e., mainly timing, magnitude and influences of flow on other natural processes) have resulted in dramatic shifts in river flow regimes, sediment transport and deposition patterns, temperature, nutrients, fish assemblages, floodplain isolation, altered high and low flow, and floodplain land use. Numerous additional effects are noted in the literature excerpts below. Most U.S. river systems are hydrologically altered by dams, but water diversions or withdrawals, channelization and human-made disruptions of overland flow also produce hydrologic alteration. Significant departure of an impaired waterbody from its range of natural flow variability is the most common mechanism among these that negatively influences recovery potential. However, dam removal, adjusting flow regulation at dams or the changing the seasonality or timing of withdrawals is often possible and can bring about recovery in many flow-altered waters.

How Measured: As channelization and general watershed influences on overland flow alteration are addressed in other metrics, this metric focuses on dams and withdrawals. Although upstream dam effects exist in the form of transformation from flowing water to pool habitat, other ecological and social values from this alteration are mixed and thus not easy to consistently interpret as stressor effects. Downstream dam effects are more easily evaluated as stressors on flowing waters, as are the sometimes similar effects below withdrawals. A scoring process of waterbody segments downstream of dams or withdrawals can consider dam sizes, active status, role on flow alteration, and feasibility of flow management.

A simplified scoring process of waterbody segments downstream of dams or withdrawals is:

- 0 – no hydrologic alteration from dams or withdrawals
- 1 – alteration from small, low-head dam or withdrawals, within natural flow range
- 2 – alteration from dam, natural flow range disrupted, managed flow feasible
- 3 – alteration from withdrawals, natural flow range disrupted
- 4 – alteration from dam, natural flow range disrupted, managed flow infeasible

Where information on the dam types is not available, the metric can be measured in terms of dam presence/absence.

Geo-Spatial Data Source: Aquatic barriers for fish passage are documented through the US Fish and Wildlife Fish Passage Decision Support System (See: http://fpdss.fws.gov/home). Major dams have been mapped through the US Army Corps of Engineers' National Inventory of Dams (See: http://www.usace.army.mil/Library/Maps/Pages/NationalInventoryofDams.aspx) but the large numbers of smaller dams on small to medium-scale streams and rivers are not uniformly documented. National Hydrography Dataset (NHD) contains data on dams and divergence structures (http://nhd.usgs.gov/). Data on water withdrawal locations may vary highly among states. An example of state withdrawal information can be found through the Michigan Department of Natural Resources and Environment (See: http://michigan.gov/deq/0,1607,7-135-3313_3677_3704-72931--,00.html).

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.

Comments: Relevant and applicable nationally in all waterbody types. Measurable in numerous ways but potentially easier to split out into several, more simple factors (channelization, dams, withdrawals) than to use as one metric. Can be data limited for some types of alteration.

Supporting Literature (abbrev. citations and points made):

- (Freeman et al., 2007) A compelling example of how important it is to consider the large-scale effects of altered hydrologic connectivity concerns alterations in the biogeochemical transport and cycling of silica as a result of the cumulative effects of dams. Rivers supply over 80% of the total silicate input to oceans (Treguer et al., 1995). Silicate stimulates production of diatoms, which fuel food webs and play a critical role in CO2 uptake (Smetacek, 1998). Increasing evidence links dam construction to decreased silicate transport and alterations in coastal food web structure (Conley et al., 2000). Moreover, reduced riverine inputs of other elements such as iron, may have far-reaching effects beyond coastal ecosystems (Hutchins and Bruland, 1998). Iron availability has been linked to patterns of silicate uptake. Therefore, reductions of riverine-transported iron (as a result of hydrological alterations) might also affect silicate uptake in nutrient- rich upwelling zones far from the coasts (Ittekkot et al., 2000). Further declines in the delivery of sediments, dissolved silicate, and other elements to estuaries and coastal oceans can be expected as new dams are constructed, with consequences to coastal food webs and wildlife.

  Environmental effects of altered nutrient transport in regulated rivers have emerged within the last two decades. This and other examples (e.g., mobilization of methylmercury in reservoirs) suggest that the current extent and magnitude of hydrologic alterations and pollutant loading will result in new, perhaps unexpected, environmental problems, and raise questions of the larger scale effects of other alterations in hydrologic connectivity (Pringle, 2003c) (7-8).
- (Freeman et al., 2007) Channelization, diversion through pipes (''piping''), impoundment and burial of headwater streams unavoidably impact stream systems by altering runoff patterns, fluxes to downstream segments, and by eliminating distinctive habitats (8).

- (Freeman et al., 2007) Headwater alteration affects ecological function at larger scales through the loss of unique functions and in relation to the importance of headwater connectivity to downstream and upland systems (8).
- (Freeman et al., 2007) Additionally, hydrologic alteration of headwater streams is generally accompanied by water quality impacts. Human activities commonly associated with headwater stream modification include land development, road construction, mining, agricultural drainage, and reservoir creation. Each activity entails significant water quality changes beyond those caused by the physical alteration of the headwater channels. Stream piping to create additional space for buildings, roads, or parking lots is accompanied by elevated streamflow, nutrients, pesticides, fecal coliforms, and pharmaceuticals that are associated with pavement, compacted soils, landscape management, domestic animal waste, and sewer leaks (Paul and Meyer, 2001). Strip mining and hilltop mining excavate some headwater streams and bury others in mine tailings. The downstream receiving waters are affected not only by the loss of the streams, but potentially by acidic ground water and streamflow created by the exposure of an enormous combined surface area of unweathered rock and the resulting oxidation of sulfides and pyrites. Stream systems altered by ditching to improve drainage from agricultural fields also receive high nutrient and sediment concentrations because of fertilizer or manure application and soil erosion. Small streams are often impounded to create ''farm ponds,'' or increasingly, to create ''amenities'' in residential developments; in both cases, the downstream drainage is influenced not only by replacement of the stream ecosystem with a reservoir but also by nutrient and sediment runoff from the landscape.

Finally, most U.S. (and other Holarctic) river systems are hydrologically altered by dams (Dynesius and Nilsson, 1994), an important fact for considering the emerging consequences of headwater disturbance. In effect, river systems are being squeezed from both ends – downstream by dams and levees that fragment mainstems and isolate channels from their floodplains, and upstream by disturbance and loss of headwater streams. The free-flowing, mid-sized river segments caught between downstream dams and impoundments and upstream headwater disturbance are frequently essential to sustaining aquatic biodiversity (see, e.g. Freeman et al., 2005) (8).

- (Freeman et al., 2007) Alteration of headwater ephemeral areas, wetlands, and streams for agricultural purposes in the Midwestern U.S. has significantly contributed to the seasonal occurrence of a large-scale (12,000-20,000 km2) recurring area of hypoxia (dissolved oxygen contents <2 mg L)1) in the Gulf of Mexico (Rabalais et al., 1996) (9).

- (Sondergaard and Jeppesen 2007) The construction of dams and reservoirs disturbs the natural functioning of many streams and rivers and shore-line development around lakes may reduce habitat complexity. New methods demonstrate how reservoirs may have a severe impact on the distribution and connectivity of fish populations, and new techniques illustrate the potential of using graph theory and connectivity models to illustrate the ecological implications (1089).

- (Sondergaard and Jeppesen 2007) Not least in arid areas, the construction of dams and reservoirs is one of the most important stressors of rivers (Gehrke *et al*. 2002; Schilt 2007). Besides affecting the natural flow, sediment transport and the pulse and water quality of the downstream river system, it also reduces the migration of the natural fish stock, leading to a fragmented fish distribution (1091).

- (Lake et al., 2007) Dams disrupt both longitudinal and lateral connectivity. Besides being barriers, dams with accompanying river regulation change flow regimes and by creating lentic reservoirs cause major changes in sediment, nutrient and organic matter dynamics and transport. As dams age, uses may alter and with public attitudes changing, dam removal is increasingly becoming a restoration strategy (Doyle et al., 2003). If the removal of small dams is carefully managed, it is quite possible to greatly reduce the harmful effects of nutrient and sediment release and to restore both habitat and connectivity (Hart et al., 2002; Stanley & Doyle, 2003) (600).

- (Andersen et al., 2007) Water resources development, particularly dam construction, is often cited as the most significant impact to rivers around the world (Dynesius and Nilsson 1994; Tockner and Stanford 2002). Tens of thousands of large and small dams have been built for water storage, power production, or flood abatement in the United States alone (Graf 1999). These developments have resulted in dramatic shifts in river flow regimes, sediment transport and deposition patterns, and floodplain land use (454).

- (Andersen et al., 2007) Flow alteration due to diversions and an upstream dam, combined with floodplain and in-channel gravel mining, have resulted in channel narrowing and forest expansion onto portions of what historically was a braided channel (Jaquette and others 2005). Expansion of riparian vegetation is common where processes maintaining braided channels have been reduced or eliminated (Friedman and others 1998) (465).

- (Freeman and Marcinek 2006) Ordination analysis of catch data showed a shift in assemblage composition at reservoir sites corresponding to dominance by habitat generalist species. Richness of fluvial specialists averaged about 3 fewer species downstream from reservoirs, and also declined as permitted withdrawal rate increased above about 0.5 to one 7Q10-equivalent of water. Reservoir presence and withdrawal rate, along with drainage area, accounted for 70% of the among-site variance in fluvial specialist richness and were better predictor variables than percent of the catchment in urban land use or average streambed sediment size. Increasing withdrawal rate also increased the odds that a site_s Index of Biotic Integrity score fell below a regulatory threshold indicating biological impairment (435).

- (Freeman and Marcinek 2006) Water withdrawals and diversions used to supply municipalities, industries, and agricultural irrigation have the potential to degrade aquatic habitats to the point that these systems fail to support native biota or to supply other ecosystem services (Moyle and Leidy 1992; Baron and others 2002; Naiman and others 2002). Prominent examples include conflicts between offstream water users and instream flow needs to sustain imperiled species (Collares-Pereira and others 2000; Cooperman and Markle 2003; Ward and Booker 2003), and collapse of fisheries and productivity in flow-deprived ecosystems (Postel 1996, 2000) (435).

- (Freeman and Marcinek 2006) Population growth and urbanization are encroaching on aquatic habitats that support high levels of aquatic biodiversity and endemism, as well as supporting imperiled species (Abell and others 2000; Warren and others 2000). Threats to native biodiversity caused by altered runoff and pollution from urbanizing areas are likely to be compounded by water supply development, largely dependent on surface water in the Piedmont, unless specific management actions are taken to safeguard vulnerable streams (435).

- (Freeman and Marcinek 2006) Management of surface waters tapped for water supply has focused on protecting minimum flow levels, although ecologists have stressed the importance of flows across the range of the natural hydrograph for maintaining structure and function of aquatic ecosystems (Poff and others 1997; Richter and others 1997; Silk and others 2000; Bunn and Arthington 2002) (436).

- (Freeman and Marcinek 2006) Previous studies of flow regulation effects on fish assemblages have indicated greater detriment to fluvial specialists, i.e., species that require flowing-water habitats for at least a portion of their life-cycle (Kinsolving and Bain 1993; Travnichek and others 1995) or rheophilic species (Copp 1990), in comparison with habitat generalist species, which are able to maintain populations in lotic and lentic systems. A recent study of fishes in a flow-depleted river in the northeastern United States similarly has revealed a shift in species composition toward habitat generalists and a loss of fluvial specialists (Armstrong and others 2001) (436).

- (Freeman and Marcinek 2006) Reservoirs, by trapping and storing water during periods of higher runoff, potentially alter downstream flows over a broader range of the flow regime than direct withdrawals (436).

- (Freeman and Marcinek 2006) The ordination showed a shift in fish assemblage structure at reservoir sites away from reference sites (Figure 2), corresponding to dominance by habitat generalist taxa in reservoir site samples (441).

- (Freeman and Marcinek 2006) Estimated richness of FS fishes varied from 0 to 24 species across study sites and years, and was, on average, lowest at reservoir sites and highest at reference sites (Figure 3A) (442).

- (Freeman and Marcinek 2006) Drainage area, WI, and reservoir presence, together, accounted for 70.3% of the among-site variation in FS richness estimates (i.e., among-site variation declined from 20.30 to 6.02). FS richness declined with increasing WI and with reservoir presence (Figure 5A) (442-443).

- (Freeman and Marcinek 2006) In the wadeable, lower Piedmont streams included in our study, increasing the potential for water withdrawal and use of an instream reservoir were associated with a loss of native fish species that are dependent on flowing-water habitats (444).

- (Freeman and Marcinek 2006) Models using drainage area, WI, and reservoir presence to predict fluvial specialist richness were better supported by our data than models using drainage area alone or in combination with any of the other site-level variables tested (444).

- (Freeman and Marcinek 2006) Our results are consistent with the hypothesis that altering flow regimes will affect stream biota in relation to the degree of alteration (Poff and others 1997; Bunn and Arthington 2002). Raising the permitted water withdrawal rate potentially increases the proportion of flow removed across the hydrograph, reducing seasonal and interannual variability in baseflow conditions. Creating water storage with instream reservoirs further increases the capacity for altering flows.

  Reservoirs use runoff to refill depleted storage, thereby diminishing high flows (e.g., during wetter seasons or years). The effect on downstream fishes may be expected to vary as a function of reservoir volume relative to inflow and rate of water withdrawal. However, despite likely variation in reservoir operations, we observed a general effect on downstream richness of fluvial specialist species. In contrast, habitat generalist species displayed no association with either the rate of permitted water withdrawal or upstream reservoir presence.

  Reservoirs can also influence downstream water quality, depending on temperature and dissolved oxygen conditions within the reservoir and from what reservoir stratum water is released downstream. The water supply reservoirs in our study primarily release surface water. Surface releases can send warm water downstream during summer, whereas hypolimnetic releases may be cooler and low in dissolved oxygen (Baxter 1977; Collier and others 1996). Our data show higher average water temperatures downstream from reservoirs than from intakes, whereas we have observed instances of low dissolved oxygen below intakes as well as reservoirs. Elevated water temperatures would be expected to be more detrimental to fluvial specialists adapted to forested streams than to pond- and lake-adapted habitat generalists (Scott and Helfman 2001). Thus, increasing downstream water temperatures may be a mechanism by which water-supply reservoirs can cause a shift in fish assemblages. Reservoirs may also trap sediments, resulting in lower downstream turbidities (Collier and others 1996), although our turbidity measurements below intakes and reservoirs are variable with broadly overlapping ranges (444-445).

- (Freeman and Marcinek 2006) However, the similarities in observed conditions during low flows below intakes and reservoirs supports the hypothesis that the apparent reduction in habitat suitability for fluvial specialists downstream from reservoirs is not solely a function of altered water quality or low-flow habitat, but also results from alteration in flow regimes (445).

- (Freeman and Marcinek 2006) For example, isolation by reservoirs (upstream and downstream) as well as close proximity to downstream urban areas and point-source discharges are likely to diminish local species assemblages, whereas connections with nearby tributary systems having intact faunal communities are likely to augment local species richness, independently of flow alteration effects. The observation that water supply variables do improve predictive models for richness of fluvial-dependent species (or probability that a site scores as impaired) implies that decisions concerning how to supply water for offstream uses will have measurable consequences for biotic integrity, even though other landscape factors may add to or modify those effects (445-446).

- (Freeman and Marcinek 2006) Across our study streams, water supply variables appear more predictive of species richness than catchment urbanization or average bed sediment size, although the effects are not precisely known (446).

- (Freeman and Marcinek 2006) The question of protective minimum flow levels remains unanswered, except to note that there is no evidence that providing for a minimum flow of 7Q10 protects stream fish assemblages, either from our data or more generally (Stalnaker and others 1995). Higher minimum flow provisions may mitigate some effects of withdrawals and reservoirs, but only if periodic low-flow depletion is the primary pathway by which hydrologic alteration influences stream biota. If biotic integrity is diminished by flow reduction during periods of normally higher base flows, then requiring a protected minimum flow level will be insufficient to protect stream ecosystem integrity (Poff and others 1997; Richter and others 1997) (447).

- (Freeman and Marcinek 2006) Our results indicate that (1) increasing permitted water withdrawal levels is likely to result in local loss of stream fish species, specifically fluvial-dependent species, and (2) construction of instream water supply reservoirs is similarly likely to result in reduced richness of fluvial-dependent species. Based on our data, streams in the lower Piedmont may begin to experience species losses if permitted withdrawal exceeds about 0.5 to one 7Q10- equivalent of water (447).

- (Gregory et al. 2002) The fundamental geomorphic change associated with a dam's presence on or removal from a river is the alteration of the longitudinal profile of the river. Dams create a long, flat water surface marked by an abrupt drop in elevation at the dam. After a dam is removed, water levels and channel positions more closely resemble the original morphology of the river, and the sediments that had been stored behind the dam are sculpted by the subsequent river flow. This adjustment to a new longitudinal profile can cause major changes in the distributions of aquatic organisms (713).

- (Gregory et al. 2002) The potential for episodic flood erosion of these high terraces and incision of lateral channels into the terraces complicates the restoration of the river and its floodplain after dam removal (714).

- (Gregory et al. 2002) Removing a dam can release large volumes of sediment to downstream reaches over short periods of time and creates easily eroded floodplains (714).

- (Gregory et al. 2002) Most dams dampen high flows, thereby reducing the beneficial effects of flooding (Junk et al. 1989), such as transporting food into streams from the terrestrial ecosystem, providing floodplain areas for feeding during floods, scouring pools and creating riffles, cleaning silt and fine sediments from gravels, creating deposits of gravel for spawning, and creating complex wood accumulations (714).

- (Gregory et al. 2002) As a result [of dams], extreme flows, both low and high, are abbreviated, and their influence in shaping the composition of aquatic communities and ecological processes is greatly reduced (714).

- (Gregory et al. 2002) Interactions of geomorphic and hydrologic processes shape river channels through both erosional and depositional processes that occur during floods that fill the active channel and extend across river floodplains. If large floods are eliminated by dams, channels can incise and impede interaction with their floodplains. In the Willamette River in Oregon, more than 50% of the channel complexity has been reduced through active channel alteration, bank hardening, and hydrologic alteration through flood control (figure 2; Gregory et al. 2002) (715).

- (Gregory et al. 2002) Even if natural flows are closely simulated in dam operation, the geomorphic effects of trapping sediment behind the dam and loss of connectivity for migrating organisms persist (715).

- (Gregory et al. 2002) Installation of dams has caused the decline of indigenous aquatic fauna and changes to riparian vegetation worldwide (Li et al. 1987, Pfleiger and Grace 1987, Friedman and Auble 1999,Hughes and Parmalee 1999,Aparecio et al. 2000, Jansson et al. 2000, Penczak and Kruk 2000, Sharma 2001).Dams influence changes in species diversity in several ways. The stream and riparian habitats are changed by inundation, flow alterations, and influences on groundwater and the water table (Friedman and Auble 1999, Shafroth 1999,Rood and Mahoney 2000). Because dams are barriers that limit the dispersal of organisms and propagules, migration patterns are interrupted, breaking key links in the life history of riverine and aquatic organisms (Andersson et al. 2000, Jansson et al. 2000, Morita et al. 2000) (716).

- (Gregory et al. 2002) Dams are sediment traps that can keep nutrients such as silica sequestered behind dams, thereby changing community composition of phytoplankton downstream, as witnessed in the Black and Baltic Seas (Humborg et al. 2000).Retention of nutrients behind dams due to the reduced velocity and longer residence time of water in the reach changes the availability of nutrients and composition of plant and microbial communities. Sediment trapping by dams will accumulate and store toxic materials that are adsorbed physically on sediment particles or absorbed actively by the biota attached to the sediments (Dauta et al. 1999).Gravels and cobbles are sequestered behind dams, which limits their recruitment downstream and leads to habitat changes in streams and estuaries (Gosselink et al. 1974, Kondolf 1997). Dams can change the natural variation of stream temperatures, depending upon the dam's size and mode of operation. Releases of hypolimnetic water (the colder, most dense layer of water in a reservoir that is thermally stratified) from high dams can lower stream temperatures, thereby limiting the reproduction of warmwater fishes and shifting downstream communities to coldwater organisms (Clarkson and Childs 2000). Conversely, low-head dams can act as heat traps and shift community composition in the opposite direction (Walks et al. 2000) (716).

- (Gregory et al. 2002) Sediment trapping has clarified normally turbid streams in the Colorado and Missouri basins. One result has been that native fishes are now exposed to greater predation by piscivores (Pfleiger and Grace 1987, Johnson and Hines 1999, Petersen and Ward 1999). Dams in streams of the Columbia basin created migration bottlenecks for migrating salmonids, exposing them to greater contact time with native predators such as northern pikeminnow *(Ptychocheilus oregonensis)* and avian predators (Buchanan et al. 1981) (717).

- Gregory et al. 2002) Changes in flow, sediments, and temperatures when dams are removed may have noticeable effects on beds of freshwater bivalves. The hypolimnetic waters that are released by dams prevent gametogenesis and spawning of warmwater mussels (Neves 1999) (718).

- (Gregory et al. 2002) A study of historical patterns of survival of different stocks of chinook salmon in the Columbia River basin concluded that survival dropped sharply in reaches affected by dams soon after construction, but survival did not change abruptly in reaches not influenced by dam construction (Schaller et al. 1999). Removal of these dams might decrease the risk of extinction for these species (720).

- (March et al. 2003) The natural flow regime is very important in sustaining the native biodiversity and ecosystem integrity of rivers and streams (Poff et al. 1997).Yet it is well known that dams and water withdrawals alter the natural flow regime of rivers (Stanford et al. 1996) (1075).

- (March et al. 2003) A major problem with in-stream reservoirs, especially in tropical areas with high rainfall, is that they fill up with sediment, which decreases reservoir storage capacity and alters downstream sediment load (1075).

- (Pringle 2001) Human perturbations that alter hydrologic connectivity include dams, stream channelization, associated flow regulation, and water extraction (from both the stream channel and groundwater). Factors such as sediment transport, acid rain, and spread of pathogens and exotic plants along river and riparian corridors are not only perpetuated by hydrologic connectivity, but also their effects are often exacerbated by changes in this property (Pringle 2000*a*) (981-982).

- (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).

- (Pringle 2001) In many countries, water diversion from rivers is so extreme that many rivers no longer flow into the sea for large parts of the year, thereby negatively affecting the biological integrity of lower river deltas, estuarine, and marine ecosystems. Such rivers include: the Colorado River in the United States (Glenn et al. 1996, China's Yellow or Huang He River, and India's Ganges River (Brown et al. 1998). Decreasing freshwater inputs increase the salinity of estuarine areas, driving concomitant biological change such as declines in mangrove forests and fisheries. The Indus River (Pakistan, India) has experienced discharge reductions of ;80%, resulting in salinity increases and massive mangrove dieoff (McCully 1996).

  Sediments once carried by rivers to deltaic and coastal areas are now retained by upstream dams. Consequently, deltas and coastal areas are often ''sediment-starved,'' leading to severe coastal erosion. Sediment once carried to the Mediterranean Sea by the Nile River is now deposited behind upstream dams (e.g., the Aswan High Dam), producing shore erosion in the Mediterranean (near Baltin, Egypt) of ;151 m/yr (McCully 1996). This pattern is repeated throughout the world, from the highly eroded Mississippi Delta on the Louisiana coast to the great mangrove forests in the Niger Delta. The latter are being eroded at a rate of tens of meters per year, as a result of a ;70% reduction in sediment transport by the Niger River following construction of upstream dams (Hinrichsen 1998). In the United States, many coastal parks suffer shore erosion: Cape Cod National Seashore, Cape Hatteras National Park, and Indiana Dunes National Lakeshore (NRC 1988, 1990). The submergence of existing wetland habitat in coastal areas is a long-term issue being examined by U.S. land management agencies (NRC 1990) (983).

- (Pringle 2001) Until recently, one of the most serious threats to the park's biological integrity was proposed dams (Fig. 3A; see NPCA 1993) upstream of the park that could potentially change the timing and quantity of water flows, altering water temperatures and disrupting transport and distribution of sediments and organic matter. The Virgin spine dace (*Lepidomeda mollispiris*), a fish endemic to the Virgin River, is adapted to a ''flash flood'' regime and would potentially be negatively affected. The absence of flash floods would also encourage the invasion of exotic species in riparian corridors, as has occurred in other regulated rivers draining federal lands in the United States, e.g., the Colorado

River where it flows through Grand Canyon National Park (Johnson and Carothers 1987, Schmidt et al. 1998) (986).

- (Pringle 2001) Intensive upstream water abstraction has made some perennial rivers ephemeral (e.g., Letaba and Luvuvhu) for prolonged periods (Moore and Chutter 1988, Russel and Rogers 1989). Although the Sabie River is the only river in Kruger National Park that has never stopped flowing, the flow has been reduced during recent years, and drops in the water table adjacent to the river channel have killed riparian vegetation in many reaches (Venter and Deacon 1995, Rogers and Naiman 1997). Aquatic vertebrates such as the crocodile (*Crocodylus niloticus*) and the hippo (*Hippopotamus amphibius*) are very vulnerable to reduced water flows and water pollution. Both play important roles in structuring riverine ecosystems (Naiman and Rogers 1997, Davies and Day 1998), and their loss can trigger a cascade of events throughout both aquatic and terrestrial foodwebs (987).

- (Pringle 2001) Thus, management alterations of hydrologic connectivity (i.e., the creation of groundwater-fed watering holes between major rivers) have altered the population dynamics of many animal species as well as vegetation (987).

- (Pringle 2001) Several dams have no water below their intakes for much of the year, and all fish and shrimp larvae suffer direct mortality when sucked into water intakes during migration (988).

- (Pringle 2001) Olympic National Park (Fig. 4A; 373 km2), located in Washington, USA, provides a temperate example of how the loss of connectivity in lower watersheds has upstream effects. Threats include: existing dams and numerous proposed hydropower projects on rivers that flow out of the park; proposed offshore oil leasing; existing oil barge and tanker traffic; logging of lower watersheds; water withdrawals from streams outside of the park; and acid precipitation (NPCA 1993). Both the Elwha and Skokomish Rivers have been devoid of migratory fishes within Olympic National Park boundaries (Fig. 4A) since they were dammed. Dams block migration of several species of anadromous salmon and trout that, after maturing in the ocean, return to rivers in the park to lay their eggs or spawn. The dams modify downstream flow of nutrients, sediment, and woody debris necessary for successful spawning and rearing of juvenile fishes. Dams also inundate fish habitat and elevate downstream water temperatures (National Park Service 1995) (988).

- (Pringle 2001) The San Pedro Riparian National Conservation Area in western Arizona has become a critical stopover for migratory birds because riparian forests on other rivers (e.g., Rio Grande, Colorado River) have largely disappeared as a result of hydrologic alteration (Stromberg et al. 1996) (991).

- (Ward 1998) Anthropogenic impacts such as flow regulation, channelization, and bank stabilization, by (1) disrupting natural disturbance regimes, (2) truncating environmental gradients, and (3) severing interactive pathways, eliminate upstream-downstream linkages and isolate river channels from riparian/floodplain systems and contiguous groundwater aquifers. These alterations interfere with successional trajectories, habitat diversification, migratory pathways and other processes, thereby reducing biodiversity (269).

- (Ward 1998) Stream regulation by dams induces major discontinuities to resource gradients and zonation patterns along the longitudinal dimension (Ward and Stanford, 1995a). Biodiversity patterns along regulated rivers are characterized by major declines at riverine sites immediately downstream from dams, followed by relatively rapid increases concomitant with the recovery of environmental conditions (Fig. 5). Stream regulation alters virtually all environmental variables downstream; the sublethal effects (direct and indirect) of modified flow and temperature regimes are paramount in structuring biotic communities below many dams throughout the world (Ward, 1982; Petts, 1984; Walker, 1985; Dudgeon, 1992). The summer-cool water released from the bottom of the high dams on the Gunnison River, a major tributary of the Colorado River, has shifted the rhithral-potamal boundary downstream 60-70kin, a vertical drop of around 500m elevation (Ward and Stanford, 1991). As a result, salmonids have extended their ranges downstream and a trout fishery now occurs in an area previously inhabited by the endemic warm-water fishes of the Colorado River basin (Stanford and Ward, 1986) (272).

- (Ward et al., 1998) Many human-induced alterations to rivers have, intentionally or not, profoundly disrupted interactions along the lateral dimension. In Europe, for example, massive river training works were completed prior to the 20th century (e.g. Vischer, 1989). Coupled with flow regulation, wetland drainage, floodplain reclamation, and other practices, many segments of formerly dynamic anastomosed floodplain rivers became highly managed single-thread channels isolated from their floodplains (Petts *et al.,* 1989; Ward and Stanford, 1995b). This disruption of lateral connectivity is so pervasive, especially in Europe and North America, that many lotic ecologists failed to appreciate until quite recently the extent to which the lower reaches of managed river systems have been modified from the natural state. The hydrologic changes resulting from flow regulation provide a clear example of anthropogenic effects on downstream river-floodplain systems (Fig. 7). Most of these effects, and their interactions, reduce connectivity and all decrease spatiotemporal heterogeneity, which ultimately reduces biodiversity (273-274).

- (Ward et al., 1998) Results from a variety of gradient analysis techniques suggest that the distribution patterns of groundwater animals are not directly related to variables associated with elevation, such as temperature, and that site-specific geomorphic and hydrologic features are major structuring agents (275).

- (Ward et al., 1998) River regulation not only lowers the water table downstream (Fig. 7), but also reduces hydraulic conductivity leading to clogging of interstitial spaces (Schalchli, 1992). In addition to reducing living space for groundwater animals, low exchange rates lead to poorly oxygenated interstitial waters. In a segment of the Rhine regulated by a hydroelectric dam, microcrustaceans dominated interstitial assemblages; aquatic insects were rare and true groundwater forms were absent (Creuze des Chatelliers *et al.,* 1992) (275).

- (Ward et al., 1998) Anthropogenic regulation of river flow reduces or eliminates the natural disturbance regime, leading to a simplification of the floodplain vegetation as pioneer stages are eliminated and successional processes are truncated (Fig. 7; Dfcamps and Tabacchi, 1994) (275).

- (Ward et al., 1998) Anthropogenic activities such as flow regulation tend to isolate the river from its flood plain, partly by suppressing the temporal dynamics of flooding that are necessary to maintain a diversity of water bodies, each encompassing a range of successional stages. This lost connectivity arrests the formation of new floodplain water bodies and accelerates terrestrialization of extant water bodies (Fig. 7). The implications for biodiversity are exemplified by the following comparison of two Danube floodplains (Loffler, 1990), one isolated from the river channel, the other with connectivity largely intact: 20 species vs 60 species of macrophytes, respectively, in disconnected and connected floodplains; 16 species vs 35 species of molluscs; and 4 species vs 30 species of fishes (275).

- (Ward et al., 1998) A meandering reach will exhibit different responses from a braided reach to a given impact, such as damming or diversion, and position along the longitudinal profile may greatly influence response variables (Ward and Stanford, 1995a) (275).

- (Ward et al., 1998) It must be remembered that rivers are 'flood-dependent' ecosystems and that the flood plains are an integral part of the river. The biota, aquatic and terrestrial, that inhabit flood plains employ an amazing array of adaptive strategies to exploit the spatio-temporal dynamics. In a floodplain river, the absence of floods constitutes a disturbance (Sparks, 1995) (276).

- (Ward et al., 1998) Anthropogenic impacts on riverine landscapes, such as damming, dredging, and channelization, disrupt natural disturbance regimes, truncate environmental gradients, and sever interactive pathways (Ward and Stanford, 19891) (276).

- (Poiani et al., 2000) When a key disturbance regime such as flooding is pushed outside (typically below) its natural range of variation, ecosystems and species that depend on conditions associated with large floods may not be viable over the long term (Poff et al. 1997) (140).

- (Poiani et al., 2000) If the annual number of days in which streamflow exceeded 125% of the bankfull discharge were limited to 14 days or less, then the mean abundance of some patch types, such as mature cottonwood, would deviate outside of the 90% confidence limits of natural flow regime simulations (Figure 5; Richter and Richter in press). Thus, the natural variability in flood duration will need to be conserved in the future to maintain the current level of functionality at this site (143).

- (Pegg et al., 2003) Altered flow has been one of the primary consequences of impoundment and channelization. Impoundments designed primarily for flood control, navigation, and water supply tend to dampen natural flow variation by storing large amounts of water for later, controlled release (Bravard and Petts, 1996). Conversely, dams built for power generation tend to accentuate natural variability by creating daily high and low flow periods to meet electrical demands (Bravard and Petts, 1996). Channelization, accomplished by armoring the shorelines, diverting water out of side channels, and straightening the channel, also influences flow by facilitating rapid transport of water downstream. Other direct consequences of channelization include loss of river connectivity to the floodplain (Ward and Stanford, 1995), changes in water quality (Whitley and Campbell, 1974), and loss of aquatic habitat (Mosley, 1983).

  Flow in many large river systems is affected by a combination of alterations, including impoundments, channelized reaches, water diversions, and numerous landscape changes in the catchment. These alterations are likely to result in complex changes to the flow regime, and the precise nature of these changes may be difficult to predict. Many factors including flow reduction in impounded reaches, increased velocities in channelized reaches, loss of diverse habitat complexes, changes in runoff and sedimentation loading rates, and altered nutrient cycles, all a result of human alteration, create an environment seldom if ever historically experienced by the native fauna in these lotic systems (Ligon et al., 1995; Ibanez et al., 1996) (63-64).
- (Stanley and Doyle 2003) Four years later, the WCD concluded that, although dams have significantly contributed to human development and the benefits derived from dams have been considerable, the economic, social, and environmental price has been unacceptably high (WCD 2001) (15).

- (Stanley and Doyle 2003) By blocking flow, dams raise water heights, inundate surrounding terrestrial habitats, and slow the velocity of flowing water in rivers. Sediments and debris that would normally remain suspended in the water column and continue to move downstream instead settle out and collect within reservoirs. Accumulation is often so substantial that some reservoirs shift from their original function of water storage to becoming sediment storage devices (Figure 1). The filling process greatly decreases the functional lifespan of a reservoir (Palmieri *et al*. 2001) and increases the likelihood of eventual dam failure (Evans *et al*. 2000) (16).

- (Thompson et al. 2005) Dams impede the flux of water, sediments, biota, and nutrients, and can strongly alter the structure and dynamics of upstream and downstream aquatic and riparian habitats and biota (Ward and Stanford 1979, Petts 1984, Poff et al. 1997) (192).

- (Thompson et al. 2005) Benthic biota downstream of a dam are at particular risk of negative impacts following dam removal because of the potential for impoundment sediments to be transported downstream. We documented significant reductions in macroinvertebrate abundance and diatom richness at downstream sites that persisted for at least 12 mo after dam removal, but observed few changes in these attributes at upstream control sites. Algal biomass also was severely reduced at downstream sites in the year following dam removal, although less severe reductions also occurred upstream of the dam. However, the removal had no detectable effects on the composition or relative abundances of taxa within macroinvertebrate or diatom assemblages (199).

- (Thompson et al. 2005) [Following dam removal] The reductions in [downstream] macroinvertebrate abundance, diatom richness, and algal biomass coincided with, and were probably caused by, the downstream transport of sediments previously stored within the impoundment (199).
- Rahel 2007) One consequence of human-aided breaching of biogeographic barriers has been the spread of noxious species that have altered aquatic ecosystems and fisheries in ways that are undesirable to humans (696).
- (Rahel 2007) Another consequence of human-aided breaching of biogeographic barriers has been the homogenization of aquatic biotas. Homogenization occurs when a few cosmopolitan species come to dominate communities at the expense of unique native species (696).
- (Rahel 2007) The circumvention of biogeographic barriers promotes homogenization of aquatic biota. Homogenization refers to the increased similarity of biota over time and is typically the result of displacement of native species by a small set of non-indigenous species that have been widely introduced through human actions (Rahel, 2004). These cosmopolitan species bring sameness to faunas that were historically unique because of biogeographic isolation (McKinney & Lockwood, 1999) (701).
- (Rahel 2007) Canals have provided a mechanism for fish to bypass historic biogeographic barriers to movement within zoogeographic provinces. For example, the Chicago Sanitary and Shipping Canal was opened in 1900 and provided a linkage between Lake Michigan of the Great Lakes–St Lawrence Province and the Des Plaines River of the Mississippi Province. This canal breached a catchment divide that had been a biogeographic barrier to fish movement between these provinces for thousands of years. After the canal was opened, a group of aquatic species that included the round goby and zebra mussel used this route to move between provinces (Kolar & Lodge, 2000). Likewise, the Erie Canal provided a linkage between Lake Erie in the Great Lakes–St Lawrence River Province and the Hudson River of the Northern Appalachian Province. Among the aquatic species that used the Erie Canal to breach the historic biogeographic barrier between these faunal provinces are the white perch and the alewife (Mills, Chrisman & Holeck, 1999) (700).
- (Rahel 2007) Humans help species circumvent such biogeographic barriers by constructing canals around the barriers or by intentionally stocking species upstream of the barriers. One of the best known examples of the former is the Welland Canal that was

built in 1829 and allowed ships to bypass Niagara Falls (Mills et al., 1999). Being 49-m high, Niagara Falls was a formidable barrier to movement of aquatic organisms from the St Lawrence River and Lake Ontario upstream into the upper Great Lakes. The Welland Canal provided a mechanism for fish such as the sea lamprey and the alewife to bypass this barrier and colonise the upper portions of the Great Lake–St Lawrence River province (701).

- (Rahel 2007) Human-assisted breaching of biogeographic barriers is largely responsible for the increased exchange of aquatic organisms among historically isolated zoogeographic regions, provinces and catchments. Three major mechanisms by which aquatic species bypass historical barriers to colonisation are through direct human stocking, entrainment in ballast water and by way of canals (Fig. 1). These pathways will have to be controlled if future homogenization of aquatic communities is to be minimized (702).
- (Rahel 2007) Canals and water transport systems will continue to be a mechanism by which species can bypass historic barriers to colonisation (Mills et al., 1999) (705).
- (Light and Marchetti 2007) Many of California's native populations of freshwater fish are in serious decline, as are freshwater faunas worldwide. Habitat loss and alteration, hydrologic modification, water pollution, and invasions have been identified as major drivers of these losses (434).
- (Light and Marchetti 2007) Hydrologic modification (impoundments and diversions), invasions, and proportion of developed land were all predictive of the number of extinct and at-risk native fishes in California watersheds in the AIC analysis (434).
- (Light and Marchetti 2007) In a survey of fishes considered extinct, declining, or endangered in California, Moyle and Williams (1990) identified water diversions as the principal cause, followed closely by introduced species and other forms of habitat modification and more distantly by pollution and overexploitation. Similarly, habitat alteration is the most commonly cited cause of fish extinctions throughout North America (73%), followed by introductions (68%) (Miller et al. 1989) (435).
- (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).
- (Light and Marchetti 2007) Diversity loss among California freshwater fishes was strongly associated with the extent of invasions, hydrologic modification, and land-use disturbance (442).
- (Light and Marchetti 2007) The regression analysis gave nearly equal importance to hydrologic modifications (as a group) and invasions as predictors. In contrast, the path analysis identified invasions as the key direct driver of native fish declines and extinctions (442).
- (Light and Marchetti 2007) Hydroelectric power development of the Pit River drainage has led to the extirpation of rough sculpins from some dewatered reaches; meanwhile, it has successfully colonized several reservoirs, extending its former range some 22 km downstream (Moyle 2002) (442).
- (Light and Marchetti 2007) Other cases, however, demonstrate that local extinctions of California fishes can also be due to habitat alteration alone. Dam building with its associated diversions and dewatered stream reaches has led fairly directly to extirpation and decline of large-river salmonids, particularly certain runs of chinook salmon, *Oncorhynchus tshawytscha* (Moyle 2002) (443).

- (Ekness and Randhir 2007) The fragmentation of rivers by dams and other impediments impairs the distribution of vegetative species along a river's edge. A significant difference in water quality can exist along the longitudinal gradient from the headwaters to the outlet (1470).
- (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).
- (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).
- (Wall et al.,2004) Decline of this species has been attributed to habitat deterioration caused by siltation, channelization, and impoundments and predation by stocked fish (Tabor 1998; Schrank et al. 2001) (955).
- (Han et al., 2008) Fish and other biological invasions result from human activities including boat traffic (i.e., ballast water invasion), aquarium fish imports, aquaculture, releases for biological control, releases to develop new fisheries, irrigation and the construction of inter-catchment canals, pipes and tunnels (Allan and Flecker 1993; Padilla et al. 1996; Ruesink 2005; Brasher et al. 2006; Jeschke & Strayer 2006; Stohlgren et al. 2006). Exotic fishes generally have a positive spatial linkage with reservoirs, their tailwaters, impounded waterways and headwater above some large waterfalls and dams because of stocking those locations for sport fishing (Pringle et al. 2000; Leprieur et al. 2006). In particular, they have a higher abundance above than below large dams in tropical streams (Holmquist et al. 1998). Anthropogenic disturbances such as water abstraction for irrigation, damming without spillway and altered thermal regime by damming also facilitate their invasion and expansion. Intentional releases to establish a new species are most responsible for the widespread establishment of the non-native fish species (Marchetti et al. 2004; Ruesink 2005). It is therefore logical to hypothesise that successful fish invasions have a strong spatial linkage with locations of known fish releases, i.e., dams and associated reservoirs (Iguchi et al. 2004; Marchetti et al. 2004; Curry et al. 2007) (1-2).
- (Pringle 2001) Because all of the fishes and shrimps inhabiting the nine major streams draining the forest are migratory, water extraction associated with dams and pollution from sewage treatment plants (in rapidly developing coastal areas; Fig. 4B) can cause massive larval mortality (Benstead et al. 1999), potentially affecting upstream recruitment of adults and other ecosystem processes (988).
- (Pringle 2001) Most herbivorous mammals live within 6 km of surface water, and water dependence is a major factor limiting their abundance (Scholes 1995). To increase available habitat, park managers provided artificial watering holes in formerly waterless areas between major rivers. This altered the population dynamics of many species and resulted in catastrophic collapses of inflated herbivore populations (e.g., in 1966 and 1983), which became food limited (987).
- (Pringle 2001) Effects of the isolation of upper watersheds on biological integrity are not well understood. Modifications of lower watersheds such as water extraction, channel modification, land-use changes, nutrient discharge, and toxic discharge can initiate a cascade of events upstream that are often not immediately associated with these original downstream sources of disturbance (Pringle 1997) (987).

- (Pringle 2001) Correspondingly, groundwater depletion and stream dewatering are contributing to loss and alteration of wetland and riparian ecosystems throughout the world (e.g., Gremmen et al. 1990, Stromberg et al. 1996), with particularly strong effects in arid and semiarid regions due to high water demand by burgeoning human populations (990).
- (Pringle 2001) The San Pedro Conservation Area is threatened by external groundwater pumping that vastly exceeds recharge rates. Stromberg et al. (1996) predict that future declines in alluvial groundwater levels will cause desertification of the riparian flora and net loss of local biodiversity (991).
- (Pringle 2001) Drilling for hot water, oil, and gas can disrupt the flow of groundwater or release hydrostatic pressure critical to geyser eruption. Groundwater extraction can also deplete groundwater tables below levels necessary to maintain surface thermal features (NPCA 1993). For example, of the 10 major geyser areas in the world, all but three have been altered in recent years through nearby development (NPCA 1993). A well drilled on Royal Teton Ranch on the park's northern boundary (Fig. 5B) recently disrupted nearby flows to hot springs located just outside the park (991-992).
- (March et al., 2003) Although dams with spillways allow the passage of migratory aquatic biota, shrimp and fish abundance upstream of these dams is lower than in stream reaches downstream of dams or in comparable reaches without dams (Holmquist et al. 1998, Concepçion and Nelson 1999). Dams with spillways can also extirpate native faunas from upstream reaches if the native faunas are unable to climb or to migrate past lakelike reservoirs. For example, in Guam, the native fish *Kuhlia rupestris,* which does not have modified pelvic fins, is absent from streams upstream of Fena Dam (26 m high, with a spillway; Concepçion and Nelson 1999). Similarly, native neritinid snails are absent from streams above the dam. While neritinid snails can climb near-vertical surfaces, they also require flowing water as a directional cue to orient themselves upstream, which the low-flow conditions of the lake-like reservoir remove (Concepçion and Nelson 1999).

  Dams with hydroelectric facilities can disrupt the upstream migration of faunas by altering the location of freshwater flow into the ocean. Postlarval gobies in coastal areas use the input of flowing freshwater as a directional cue to locate rivers. When hydroelectric facilities discharge river water directly into the ocean, postlarval gobies can have difficulty differentiating between river outflow and hydroelectric facility discharge. For example, in Guadeloupe, West Indies, large upstream migrations of postlarval *Sicydium* (Gobiidae) have been observed entering canals leading to a hydroelectric facility that does not have access to the stream (Fievet and Le Guennec 1998) (1070-1071).
- (March et al., 2003) Large dams and associated reservoirs may also disrupt the downstream migration of fish and shrimp larvae by reducing water flow, thereby lengthening the time water takes to reach the estuary (1071).
- (March et al., 2003) Although reservoir-induced starvation has not been documented, it is highly probable. Furthermore, reduced flows through reservoirs may also increase predation on larvae.

  Small low-head dams also interfere with the migration of tropical island faunas (Benstead et al. 1999, Fievet et al. 2001a). The effects of low-head dams on the upstream migration of faunas appear to be similar to those of large dams with spillways (1071).
- (March et al., 2003) While fishes and shrimps can surmount the low-head dam when water is flowing over it, the dam does appear to slow their migration, resulting in increased densities below the dam compared with those above it (figure 8) (1073).
- (March et al., 2003) Reviews of studies of continental rivers suggest that reduction in river flow reduces the productivity of coastal fisheries (e.g., Loneragan and Bunn 1999) (1076).
- (Paul and Meyer 2001) Benthic feeders quickly reappeared as sedimentation rates declined after construction. Flow modification associated with urbanization also affects stream fish. In the Seine, modification of flow for flood protection and water availability

has affected pike (*Esox lucius*) by reducing the number of flows providing suitable spawning habitat. With urbanization, the river contains enough suitable spawning habitat in only 1 out of 5 years as opposed to 1 out of every 2 years historically (Boet et al. 1999) (352-353).

- (Light and Marchetti 2007) Quantitative analyses of native fish declines, invasions, and habitat alteration in this region reveal that dams, channelization, and water pollution are associated with both native species decline and with the richness and abundance of introduced fishes (Aparicio et al. 2000; Corbacho & S´anchez 2001; Clavero et al. 2004) (443).
- (Light and Marchetti 2007) In the analysis of variable importance, native richness was the most important single predictor of number of fishes of conservation concern in a watershed, entering every highly ranked model. The variable nonindigenous richness was next in importance, followed by the variables aqueduct density and dams (Table 3). In the analysis of category importance, the cumulative rank of all models including water development variables (0.919) was the highest, followed closely by models including nonindigenous richness (0.811) and more distantly by models including land-use variables (0.187) (441).
- (Han et al., 2008) Dams alter physical and hydrologic aspects of riverine systems affecting both quality and quantity of water flows. The majority of the largest river systems in the northern hemisphere are currently affected by fragmentation of river channels by dams (Dynesius & Nilsson 1994). Consequently, dams are considered one of the most significant obstacles in restoring the biodiversity and integrity of riverine systems (The Heinz Center 2002). A number of studies have demonstrated the negative effects of dams on native fish species. Dams can be a barrier to migration and degrade habitats because of altered flow and sediment regime (e.g., Morita & Yamamoto 2002; Wofford et al. 2005; Fukushima et al. 2007; Han et al. 2007; Isaak et al. 2007) (2).
- (March et al., 2003) However, the low-head dam did act as a bottleneck that increased the densities of upstream migrating animals below the dam. This high concentration of migrating juvenile fishes and shrimps attracted a variety of predators, such as green herons *(Butorides virescens),* adult shrimps and crabs, and mountain mullet *(Agonostomus monticola),* probably resulting in increased mortality of migrating fishes and shrimps (1071).
- (March et al., 2003) Upstream migrating animals at this site experienced increased difficulty during periods of low river flow, when no water was coming over the dam. During these periods,migrating shrimps lacked the directional cue provided by flowing water and became disoriented (Benstead et al. 1999).

  In addition to impeding the upstream migration of migratory faunas, water withdrawal at this site resulted in significant mortality of downstream migrating shrimp larvae (Benstead et al. 1999) (1071).
- (March et al., 2003) Long-term estimates of larval shrimp mortality were between 34% and 62%, depending on the amount of water withdrawn.

  Construction of low-head dams also alters the physical habitat and hydrological regime in reaches both upstream and downstream of dams (Fievet et al. 2001a).Areas downstream of low-head dams can experience decreased river flow and water depth (Fievet et al. 2001a). This decrease in freshwater can result in increased salinity, especially at sites near the upstream boundary of estuarine tidal influence. Areas upstream of low-head dams experience decreased flow rates and increased water depth (1071).
- (Novotny et al., 2005) The models [for assessing ecological integrity] (functions) link the individual risks and consider their synergy, addictivity, or antagonism. The risks include:
  - (1) Pollutant (chemical) risks, acute and chronic, in the water column
    - Key metrics: Priority (toxic) pollutants, DO, turbidity (suspended sediment), temperature, pH.
  - (2) Pollutant risk (primarily chronic) in sediment

Key metrics: Priority pollutants, ammonium, DO in the interstitial layer (anoxic/anaerobic or aerobic), organic and clay content.

(3) Habitat degradation risk

Key metrics: Texture of the sediment, clay and organic contents, embeddedness, pools and riffle structure, bank stability, riparian zone quality, channelization and other stream modifications.

(4) Fragmentation risk

Key metrics:

Longitudinal—presence of dams, drop steps, impassable culverts.

Lateral—Lining, embankments, loss of riparian habitat (included in the habitat evaluation), reduction or elimination of refugia.

Vertical—lack of stream-groundwater interchange, bottom scouring by barge traffic, thermal stratification/heated discharges, bottom lined channel (190).

- (Pringle 2001) One proposed project on the Sarapiqui River, just west of La Selva (Fig. 6A), would remove ; 90% of the river water over a distance of 4 km by diverting it into a large metal pipe. Cumulative effects of these projects may affect the biological integrity of La Selva Biological Station and Braulio Carrillo National Park. Alterations of river flow regimes as a result of this recent hydropower development could cause negative physical, chemical, and biological changes in stream ecosystems, riparian zones, and floodplain swamp forests (Pringle 2000*a*). (994).
- (March et al., 2003) For example, a slow-flowing pool extends as far as 170 m upstream of the low-head dam, with reduced channel depth and slope profile. Channel depth increases directly in front of the dam wall because of displacement of sediment during turbulent flow associated with large spates. Spates and associated scour also account for increased channel depth just below the dam (figure 7a) (1073).
- (March et al., 2003) The low-head dam also appears to entrain more downstream-drifting shrimp larvae. As mentioned previously, during a 69-day study the low-head dam caused direct mortality of 42% of downstream-drifting larvae (Benstead et al. 1999). Furthermore, during periods of low river flow, there was no discharge over the dam, causing 100% mortality of drifting larvae (Benstead et al. 1999) (1073).
- (Andersen et al., 2007) The universal form of flow alteration found on the rivers in this study—reduction in annual peak flow—can affect forest extent differently on different rivers, depending on the historic timing of the annual flood peak. Where the peak flood occurred in spring, its reduction slows creation of new cottonwood germination surfaces and precludes cottonwood recruitment at high positions within the historic array of potential recruitment sites (Cooper and others 1999, 2003; Fenner and others 1985). Where total annual flow has not been greatly affected, base flows are typically increased, further constraining recruitment sites (Merritt and Cooper 2000). Thus, the ‘‘recruitment box’’ (Mahoney and Rood 1998) is narrowed and the potential spatial extent of postalteration cohorts is severely reduced compared to the extent of prealteration cohorts (463).
- (Andersen et al., 2007) They attributed these patterns to the independent adverse effects of hydrologic alteration and floodplain development on regeneration. The likelihood of the presence of regeneration was reduced 65% by development.

  Flow regime alterations on rivers featuring spring floods also can stress mature (preexisting) trees, with responses that vary from widespread branch and root system dieback (Williams and Cooper 2005) to death of some individuals. Nonlinear responses to flow alteration are likely, with thresholds for major effects perhaps dependent on prior natural or anthropogenic variability in the flow regime (Andersen 2005) (464).
- (Andersen et al., 2007) We suggest that the large area of CF with <5% canopy cover is a result of an accelerated thinning process initiated in mature stands when flow regimes were altered (464).

- (Andersen et al., 2007) Unless specifically designed to avoid it, flow alterations associated with new projects will almost certainly lead to further loss of cottonwood forest (465).
- (Rahel 2007) Through a variety of mechanisms, humans have increased the connectance among aquatic systems that were historically isolated by biogeographic barriers to movement. This human-aided breaching of biogeographic barriers has led to significant homogenization of aquatic biotas (706).
  Factors that promote the breaching of biogeographic barriers include the increasingly global nature of commerce, the movement of water through canals and growth in the aquaculture and ornamental fish industries (706).
- (Han et al., 2008) The mean native species richness (± SD) was significantly higher at grids with no dams (3.62 ± 2.63) than sites in inlet streams (2.73 ± 1.71) and lowest at sites in reservoirs (2.44 ± 0.14; Kruskal–Wallis rank sum test, v2 = 91.5, d.f. = 2, P < 0.001). In contrast, the mean non-native species richness was significantly higher at sites in reservoirs (1.20 ± 1.09) compared to sites in inlet streams (0.53 ± 0.74) and lowest at sites with no dams (0.42 ± 0.86) (Kruskal–Wallis rank sum test, v2 = 160.7, d.f. = 2, P < 0.001) (4).
- (Han et al., 2008) The dam variable however was a significant predictor for non-native species richness and proportion of native species. Sites in reservoirs had the greatest nonnative species richness followed by sites in inlet streams after the effects of all the other environmental variables were taken into account (Fig. 2t). Conversely, reservoirs were predicted to have the smallest proportion of native fish species followed by sites in inlet streams (Fig. 2u) (4-5).
- (Han et al., 2008) While elevation, a natural environmental variable was the second most important predictor for native species richness after survey year as the most significant variable, variables related to human activities, dam construction and land use patterns were more important to determine non-native species richness (7).
- (Han et al., 2008) Non-native fish species richness had a clear linkage with the presence of dams. The non-native species richness was highest in reservoirs, followed by inlet streams above reservoirs. It was lowest in reaches with no dams. This is consistent with the findings from previous studies (Holmquist et al. 1998; Pringle et al. 2000; Leprieur et al. 2006), where exotic fishes generally are introduced in reservoirs and favoured by regulated flow. The spatial linkage must be interpreted cautiously, however, because it did not apply to non-native fish species in general but was only specific to salmonids. Numerous reservoirs in Hokkaido have experienced prolific releases of rainbow and brown trout for sport fishing (Takami & Aoyama 1999; Takayama et al. 2002). The introduction of large non-native piscivores such as salmonids can lead to dramatic shifts not only in the fish assemblage structure but also the entire food-web structure as a result of the cascading nature of trophic levels (Carpenter et al. 1985). This may be especially true for Japan where native large piscivores were originally absent (Iguchi et al. 2004) (7).
- (Han et al., 2008) Our hypothesis that the invasion of non-native fish species is spatially linked to the location of dams was partly supported. Although species richness of nonnative fishes was significantly higher above dams, the species that increase in reservoirs and inlet streams comprised only salmonids. Non-native salmonids can be a significant threat to the indigenous fish fauna of Hokkaido, especially native salmonids (Hasegawa et al. 2004). Located higher in latitude, Hokkaido has historically supported healthy populations of various native salmonids, such as Sakhalin taimen (Hucho perryi), whitespotted char (Salvelinus leucomaenis), Dolly Varden (S. malma), masu salmon (O. masou), chum salon (O. keta) and pink salmon (O. gorbuscha) (Miyadi et al. 1996). Large dams have land-locked some of the native salmonid populations in reservoirs and inlet streams (Edo et al. 2000; Tamate & Maekawa 2002), potentially intensifying the interaction between native and non-native salmonids. Competition with predation by and hybridization with the introduced non-native salmonids could increase the risk of extinction of the land-locked native salmonids. Constructing fish ladders may not provide

a solution to dams because it also enables non-native fishes to escape from reservoirs and to enlarge their distribution. It is most desireable to prevent the introduction of non-native species. If this occurs, their populations can multiply at the expense of native species (7-8).

- (Bernhardt and Palmer 2007) Infrastructure thus limits site-specific options, but it also reduces connectivity between segments of river networks, with important implications for populations of stream biota dependent on upstream–downstream dispersal (746).
- (Pringle 2001) Diversion of geothermally modified waters for potable water supplies may affect the structure and function of stream ecosystems at La Selva. Moreover, land use and other hydrological alterations in the landscape have the potential to alter interbasin transfers of geothermally modified groundwater to La Selva by affecting the quality and quantity of groundwater recharge (Genereux and Pringle 1997). These interbasin transfers are responsible for large fluxes of water and solutes to lowland streams, and can account for *over half* of the stream discharge and major cations at some times of year (Wood et al. 1998). La Selva is therefore very vulnerable to water quality and quantity alterations via regional hydrological connections that extend well beyond its boundaries and watershed (994).
- (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).
- (Han et al., 2008) Damming generally exerts negative effects on native fish species, especially on migratory species (Holmquist et al. 1998; Joy & Death 2001; Fukushima et al. 2007). In this study, however, native species richness was not significantly correlated with the presence of dams (7).
- (Pringle 2001) The isolation of upper watersheds within reserves can sometimes be used as an opportunity to reintroduce and/or manage endangered species (990).
- (Morita and Yamamoto 2002) Exotic fishes approach just below the dams, turning dammed-off habitats into refuges for native fishes in some rivers (e.g. Takami et al. 2002). Therefore, managers should consider this potential benefit of dams before fish ladders are installed (1322).
- (Palmer et al., 2005) Degraded running water systems (e.g. following dam construction) are typically characterized by a major reduction or alteration in variability (Baron *et al*. 2002; Pedroli *et al*. 2002). Often the limits have been so far exceeded that resilience has been lost (Suding, Gross & Housman 2004). Unless some level of resilience is restored, projects are likely to require on-going management and repair, the very antithesis of self-sustainability. Thus, we argue that, to be ecologically successful, projects must involve restoration of natural river processes (e.g. channel movement, river–floodplain exchanges, organic matter retention, biotic dispersal). Restoring resilience using hard-engineering methods should not be the first method of choice as they often constrain the channel. However, there are situations in which engineered structures may enhance resilience (e.g. grade restoration facilities that prevent further incision and promote lateral channel movement, Baird 2001; projects providing fish access to spawning reaches through culvert redesign or by establishing pathways to the floodplain, NRC 1992) (211-212).
- (Filipe et al., 2004) Once reserve areas have been selected, they must be integrated within a basin management approach to harmonize development opportunities and exploitation of aquatic resources (Meffe 2002). There is also a need for ecologists, conservationists, social scientists, and stakeholders to negotiate use rights (Cullen et al. 1999). In multinational water bodies, such as the Guadiana River basin, international collaboration is needed and all social, economic, and political constraints should be considered. Additionally, the establishment of discrete reserves is not enough to protect freshwater fishes (Angermeier 2000; Meffe 2002). Interventions upstream or downstream must be considered in the management of reserves because these activities could have implications for the species for which the reserve is designed (Cowx &

Collares-Pereira 2002). In particular, the construction of a dam outside of the reserve network has implications for the recolonization of each reserve area because it may disrupt migration pathways. Similarly, the introduction of alien species elsewhere in the watershed may have long-term implications if the introduced species is able to disperse into the reserves. In our case study, the Alqueva and Pedrogao reservoirs will create unsuitable habitats for native fishes by affecting their movement and enhancing the populations of exotic species. In addition, the lack of facilities for fish passage around Alqueva has permanently isolated the populations upstream and downstream of the dam (197).

- (Ekness and Randhir 2007) Lateral [riparian] and longitudinal [stream order] connectivity and flow regime are critical factors that influence watershed health. The latter can be impaired by land and water use practices that affect biotic diversity, water quality, esthetics and hydrology (Brooks et al., 1997) (1469).
- (Palmer et al., 2005) Some relatively undisturbed river ecosystems are impacted by upstream impoundments or water withdrawals. In these systems, ecologically effective restoration will move the system closer to the natural hydrograph. Ecologically ineffective restoration will focus exclusively on maintaining some minimum instream flow, but will fail to re-establish the natural flow regime. The first approach will be successful in that it may restore cues for fish spawning and riparian plant germination, high flows for nutrient regeneration and channel maintenance, and groundwater connectivity. The latter approach will maintain the river channel but without re-establishing these additional ecosystem benefits (213).
- (Pegg et al., 2003) Visual inspection of Figure 2 indicates that flows in the middle reaches of the river (i. e., the interreservoir zone and the upper portions of the channelized zone) have changed dramatically between the pre- and post-alteration periods as evidenced by a decrease in flow variability during the post-alteration period. In contrast, flow variability has maintained some integrity between the two periods at Fort Benton (i.e., unaltered zone), the upper most gauge, and Hermann (i.e., channelized zone), the lower most gauge on the Missouri River (Figs. 1 and 2) (66).
- (Pegg et al., 2003) As with mean flows (Fig. 2), the amount of variability in the residual plots is lower after alteration in the inter-reservoir and upper channelized zones of the river as represented by Bismarck, ND, and Omaha, NE (Fig. 3). Conversely, in the extreme upper and lower gauges on the river, residual variability is similar in pre- and post-alteration periods.

  Daily mean flows were significantly higher during the post-alteration period at all gauges ($P < 0.10$; Table 1). Post-alteration daily flows averaged 16% higher than the pre-alteration flows at Bismarck, ND, and 10% higher at Yankton, SD (Table 1). The remaining gauges had daily mean flows during the post-alteration period that averaged from 30 to 45% higher than pre-alteration flows (66).
- (Pegg et al., 2003) Graphical comparisons of pre- and post-alteration daily mean flows during the spring fish spawning season were qualitatively similar to those made over the entire year, with the most obvious changes appearing in the middle sections of the river (66).
- (Pegg et al., 2003) Post-alteration, spring daily mean flows at the two uppermost gauges (Fort Benton, MT, and Wolf Point, MT) and the two lower most gauges (Boonville, MO, and Herman, MO) were not significantly different between the two time periods (Table 2). In contrast, gauges located in the middle portion of the river (Bismarck, ND, to Kansas City, MO) did significantly differ ($P < 0.10$) among flow periods, and percent change appeared to follow a longitudinal gradient from a high-negative to a low-positive. Spring spawning daily flows at Bismarck, ND, averaged 32% lower, Yankton, SD, averaged 28% lower, and Omaha, NE, averaged 5% lower during the post-alteration period; whereas, flows from Nebraska City, NE, to Kansas City, MO, averaged 5 to 7% higher during the post-alteration period (67).
- (Pegg et al., 2003) Additionally, variation was markedly reduced in the post-alteration period due to the regulation of flows from impoundments. Consequently, daily mean flows

were significantly higher during the post-alteration period at all gauges when analyzed at the annual scale. There were also significant differences at the most strongly human-influenced gauges during the spring flow period (Table 2) (68).

- (Pegg et al., 2003) Because water is held back in each of the six mainstem reservoirs during spring flooding, we would expect the spring spawning flows in the inter-reservoir reaches and other areas influenced by dam operations to be lower than the pre-alteration period. Our findings support this prediction in that the Bismarck, ND, and Yankton, SD, gauges (Fig. 4) experienced a marked decrease in spring spawning flows during the post-alteration period. The Omaha, NE, gauge also experienced slightly lower spring fish spawning flows, indicating that the river is still influenced by reservoir operations roughly 250 km downstream of the last impoundment. However, moving downstream from these impoundments appears to mediate flow differences between the two periods due to inputs from relatively large tributaries (Galat and Lipkin, 2000; Pegg and Pierce, 2002) (69).
- (Pegg et al., 2003) Spring flows are important to the ecology of large rivers and is an area of strong concern when addressing biological problems throughout the Missouri River system (Galat et al., 1996). The flood-pulse concept (Junk et al., 1989) is based on the theory that biological communities in large floodplain rivers have evolved to utilize the timing, duration, and water level changes generally associated with spring flooding. These floods trigger fish spawning events, and provide food and nursery areas in addition to maintaining diversity within the system (Johnson et al., 1995). Therefore, basic biological functions such as spawning and recruitment may be curtailed, causing negative responses in diversity and density of native fishes due, in part, to the removal of flooding events as seen in the middle reaches of the river. Fish community information from the Missouri River suggests that species richness and abundances are much lower than would have been expected along a natural species gradient in the impoundment influenced reaches of the Missouri River compared to other reaches (Pegg, 2000) (69).
- (Pegg et al., 2003) Analyses from studies investigating other aspects of flow on the Missouri River have generally reached the similar conclusion that the middle portion of the river has been most significantly altered (e.g., Galat and Lipkin, 2000). Using the IHA approach over a similar time period, Galat and Lipkin (2000) found that the relatively unaltered areas of the upper Missouri River, and to some extent the lower 600 km before joining the Mississippi River, maintained a certain degree of natural variability after impoundment, whereas the middle portion of the river was substantially altered. Coupling these findings with our study demonstrates a consistent trend in flow alteration that is most pronounced in the middle portion of the river (69).
- (Palmer et al., 2005) We propose that the first step in river restoration should be articulation of a guiding image that describes the dynamic, ecologically healthy river that could exist at a given site. This image may be influenced by irrevocable changes to catchment hydrology and geomorphology, by permanent infrastructure on the floodplain and banks, or by introduced non-native species that cannot be removed. Rather than attempt to recreate unachievable or even unknown historical conditions, we argue for a more pragmatic approach in which the restoration goal should be to move the river towards the least degraded and most ecologically dynamic state possible, given the regional context (Middleton 1999; Choi 2004; Palmer *et al*. 2004; Suding, Gross & Housman 2004) (210).
- (Stan et al., 2002) In addition, subsequent erosion of sediment deposits behind the dam results in frequent and complex channel change within the reach upstream of the dam (714).
- (Stan et al., 2002) Dams are a primary cause of its severe decline. Many of these dams did not provide fish ladders, thus blocking passage to spawning areas upstream, and altered habitat conditions for pelagic eggs and shad larvae (figure 3a) (Walburg and Nichols 1967) (717).

- (Stan et al., 2002) daily hydrological regimes are modified by the dams. But these dams block fish passage and trap more than 13 million m3 of sediment, mostly behind Glines Canyon Dam (719).
- (Stan et al., 2002) The courts determined that the dams caused temperature increases and gas supersaturation that exceeded limits under the Clean Water Act (720).
- (Stan et al., 2002) Return of anadromy could also affect food webs upstream. For example, resident steelhead and Dolly Varden would lose some spawning habitats associated with reservoirs and also be subject to greater competition and predation by juveniles of other salmonid species (720).
- (Stan et al., 2002) The lake-like conditions of the reservoir reaches have created favorable conditions for almost a century for some plants and animals that will be adversely affected by dam removals. Shoreline cover along Lake Aldwell will greatly diminish and thus significant habitat for lacustrine mink will be removed (FERC 1991). Surprisingly, beaver are likely to increase with recolonization of hardwoods along riverine terraces. Wetland biomes that have developed along lake edges will disappear with their associated plants, one of them a bicolored linanthus unique to the Elwha valley (FERC 1991). Eventually other wetlands are expected to develop along stabilized backwater and meanders of the reestablished floodplains (720).
- (Stan et al., 2002) Despite dam passage improvements that have dramatically mitigated direct mortality associated with dams, the NMFS concluded that the removal of the dams would not reduce the risk of extinction under current conditions (721).
- (Stan et al., 2002) One of the major ecological benefits of dam removal is the restoration of hydrologic regimes, particularly in the local reach and immediately downstream(715).
- (Stan et al., 2002) Dam removal potentially restores hydrologic conditions and permits more dynamic channels (715).
- (Stan et al., 2002) Dams in northwestern rivers influence salmonids and other species by eliminating spawning and rearing habitats in the area covered by reservoirs, changing water velocities that influence migration rates, altering currents that are attractants for migrating fish, forcing some fish through turbines where they experience extreme pressures, increasing river temperatures as the sun warms the slower waters of the reservoir, exposing migrating juvenile fishes to fish and avian predators, and modifying flood patterns that shape river habitats and maintain spawning gravels.Removal of dams potentially restores river temperature patterns, flow patterns for migrating fish, and flood dynamics. The potential negative impacts of dam removal on salmonids are associated primarily with the instabilities of sediments and terraces stored behind the dam (716).
- (Stan et al., 2002) Because mussels and other bivalves depend on flowing water and unimpeded movements of host fish, dam removal may allow reconnection of populations of bivalves fragmented by lentic waters behind dams (718).
- (Stan et al., 2002) The short-term effects of dam removal will include the redistribution of large volumes of silt downstream (Stoker and Harbor 1991), but eventually additions of gravels will open up extensive reaches of usable spawning habitat in the middle reach (719).
- (Stan et al., 2002) Overall, removing the dams will greatly enhance anadromous fish runs and, consequently, food chains. Dramatic increases in salmon carcasses are expected to provide nutrients and food resources to juvenile fishes and other aquatic predators. Changes in hydrology and return to natural flow patterns will influence downstream temperatures and instream dynamics (720).
- (Stan et al., 2002) these levels are most likely harmful to fish eggs. These elevated water temperatures may increase the infection rate of *Dermocystidium* bacteria, which attack salmonids as they come from marine systems into fresh water (FERC 1991). Lowered water temperatures after dam removal would decrease the incidence of the disease and thus potentially increase salmonid survival (720).
- (Stan et al., 2002) Though the dams were managed as run-of-the-river flows, daily fluctuations were not conducive for stable invertebrate populations, especially organisms

in habitats associated with the river margin,with potentially similar effects on young fish.More naturally predictable flows will contribute to increases in productivity at all levels (720).

- (Stan et al., 2002) When the Elwha and Glines Canyon dams are removed, an estimated increase of 160,000 m3 in sediments will be supplied at the mouth of the river (FERC 1991). It is possible that these sediments will have shortterm impacts on downstream communities and nearshore marine benthic communities and shellfish at the mouth of the Elwha River (720).
- (Stanley and Doyle 2003) Dam removal can result in decades of accumulated material being released downstream in a rapid and catastrophic fashion (17).
- (Stanley and Doyle 2003) Unfortunately, despite awareness of the importance of sediment management, there is remarkable uncertainty regarding patterns and rates of sediment transport following dam removal (Rathburn and Wohl 2001; Pizzuto 2002) (17).
- (Stanley and Doyle 2003) Problems can arise when channel formation processes expose and transport material previously stored behind the dam (18).
- (Stanley and Doyle 2003) Following partial removal of the Fort Edwards Dam in 1973, large quantities of oils and sediments rich in polychlorinated biphenyls (PCBs) were released into the river, requiring a costly cleanup effort (Shuman 1995). The sediment moved into the river, where it restricted flow and blocked the navigation channel and access to adjacent riverside businesses. The altered flow created an additional health hazard when sewage, discharged into the river by the town of Fort Edwards, could not be conveyed downstream (Heinz Center 2002). A second wave of contaminated sediments was mobilized in 1991, when the remaining structure was removed. The following year, average PCB concentrations in striped bass had doubled (HRF 2002). Many US dams were originally built for industrial purposes, or to act as focal points for urban growth. This means that sediment contamination may not be unusual in these older reservoirs, adding additional costs and urgency to a removal process (Lenhart 2003) (18).
- (Stanley and Doyle 2003) In many Midwestern states, reservoir sediments frequently contain a similar chemical legacy in the form of nutrient-rich particles derived from past and present agricultural activity (Stanley and Doyle 2002). Removal may then reintroduce nutrients that had been stored for decades, causing enrichment of downstream rivers, lakes, and even coastal areas. In support of this prediction, Gray and Ward (1982) found that the flushing of sediments from the Guernsey Reservoir on the North Platte River caused a sixfold increase in downstream phosphorus concentrations and stimulated the growth of large filamentous green algal mats (18).
- (Stanley and Doyle 2003) Thus, there is the very real possibility that by adding to already elevated nutrient concentrations in rivers, dam removal will be at odds with nutrient management strategies in some parts of the US (Stanley and Doyle 2002) (18).
- (Stanley and Doyle 2003) Sediments mobilized by channel formation processes in the reservoir are transported downstream, where they settle on channel beds and banks (Figure 6). The amount of suspended sediment increases greatly during and after drawdown and removal, often by three to five orders of magnitude (Childers *et al*. 2000; Doyle *et al.* in press *a*), and conditions of high turbidity may persist for months (Perrin *et al*. 2000). Because reservoirs trap fine particles, released material can remain suspended in the water column for several kilometers (Gray and Ward 1982). The ecological impact of suspended sediment specifically released by dam removal has not yet been considered, but many negative effects of both pulsed and sustained inputs of sediments to stream biota are well documented (Waters 1995) (18).
- (Stanley and Doyle 2003) Decreased streambed particle size, sediment deposition on lateral and in-channel bars, and filling of a downstream impoundment followed three small dam removals in Wisconsin (Stanley *et al*. 2002; Doyle *et al*. in press *a*). However, downstream sediment deposition does not always produce detectable changes in algal or invertebrate communities (Stanley *et al*. 2002, Bushaw-Newton *et al*. in press), either because the magnitude of impact is minimal or because the rate of recovery is rapid. In contrast, sediments released from a Colorado reservoir filled pools and clogged the

interstitial spaces between coarse sediments in the channel bed up to 12 km below the reservoir (Wohl and Cenderelli 2000) and killed over 4000 fish (Rathburn and Wohl 2001). Declines in densities and shifts in species composition of macroinvertebrate communities were also observed (Zuellig *et al*. 2002). Similar patterns of fish and invertebrate mortality were reported by Doeg and Koehn (1994), following the desilting of a small reservoir in Australia. A second reduction in fish and invertebrate numbers also occurred several months later, demonstrating that the downstream effects of sediment releases may be prolonged as the material works its way through the system (19).

- (Stanley and Doyle 2003) For instance, we have observed that deposition of fine sediments caused localized mortality of freshwater mussels following a dam removal in southern Wisconsin, a worrisome observation given the precarious conservation status of this group (Master 1990) (19).
- (Stanley and Doyle 2003) Despite these apparent successes, removal of dams as a means of restoring fish species that migrate up rivers to breed has been an area of contention in dam and fisheries management (19).
- (Stanley and Doyle 2003) First, dam removal occurs in circumstances far different from dam construction, since not only do dams change rivers over their lifetimes, but the area surrounding the dam also changes. Rather than erasing past environmental legacies, dam removal creates a new ecological template upon which subsequent physical, chemical, and biological processes will be played out. Second, regardless of the long-term outcomes, removing a dam is not a gentle process. It disrupts and reconfigures the existing physical environment and eliminates an entire ecosystem. Dam removal should therefore be considered a disturbance in the strict ecological sense of a “discrete event in time that disrupts ecosystem, community, or population structure, and changes resources, substrate availability, or the physical environment” (White and Pickett 1985). Also, because it is a disturbance, we should expect substantial changes in many ecological variables, including the loss of resident flora and fauna and the disruption of ecosystem processes, at least in the short term (20).
- (Stanley and Doyle 2003) The specific nature of the trade-offs [when deciding whether ot not to remove a dam] will depend on the size and configuration of the dam and reservoir, local legacies, and the composition of the resident biota (20).
- (Stanley and Doyle 2003) Mortality rates of virtually all reservoir populations, except fish, will be extremely high and can be expected to approach 100% if dewatering is rapid. For some groups of organisms, replacement of reservoir assemblages by more typical riverine taxa can occur relatively quickly after the dam is taken out. For example, fish and macroinvertebrates adapted to slow-moving water and silty sediments gave way to riverine taxa within a year of removal of two separate dams in Wisconsin (Kanehl *et al*. 1997; Stanley *et al*. 2002). Much to the delight of local anglers, changes in the fish community included declines in common carp (*Cyprinus carpio*) and increases in smallmouth bass (*Micropterus dolomieu*) and darters (*Etheostoma* and *Percina* spp). In both studies, the recovery of riverine taxa reflected both recolonization of individuals that had previously resided upstream or downstream from the dam and successful reproduction within this newly created habitat (16).
- (Stanley and Doyle 2003) One of the most widely publicized ecological aspects of dam removal is the elimination of barriers to fish migration (19).
- (Stanley and Doyle 2003) Following the removal of the Edwards Dam in Maine’s Kennebeck River, striped bass, alewife, shad, Atlantic salmon, and sturgeon all traveled past the former dam site (American Rivers 2002) (19).
- (Thompson et al., 2005) Removal of small dams can be expected to restore lotic habitat within the former impoundment (Bushaw-Newton et al. 2002, Stanley et al. 2002), and may improve fish passage (Stanley and Doyle 2003), but downstream benefits are less certain. For example, removing low-head, run-of-river dams that have short hydraulic residence times and limited storage volumes may have little impact on downstream water quality, thermal dynamics, or flow regimes (Hart et al. 2002). Downstream biota, particularly benthos, will not necessarily benefit from such removals. Small dam

removals may have negative effects on downstream biota. In particular, the downstream transport of sediments previously stored in impoundments has potentially serious consequences for downstream communities (Shuman 1995, Wood and Armitage 1997, Bednarek 2001, Poff and Hart 2002). Severe depletion of downstream benthos could reduce the effectiveness of dam removal as a restoration method. For example, the benefits associated with increased access by fish to upstream habitats following dam removal might be offset by corresponding reductions in food availability within downstream habitats (193).

- (Thompson et al., 2005) In the context of restoration, the downstream effects of removing the Manatawny dam are unlikely to have serious implications. The structure of macroinvertebrate and algal assemblages was largely unaffected, and abundances are likely to recover rapidly once fine sediment loads are reduced in downstream riffles. Stanley et al. (2002) also found no change in macroinvertebrate assemblage structure or condition metrics following dam removals in a Wisconsin river. These results, and observations that even severely depleted benthic assemblages often recover rapidly once sediments released from reservoirs are flushed from the system (Gray and Ward 1982, Doeg and Koehn 1994, Stanley and Doyle 2003), suggest that small dam removals are unlikely to have long-term deleterious impacts on downstream benthic communities as long as highly vulnerable species (e.g., endangered freshwater mussels; see Sethi et al. 2004) are not at risk. If downstream effects are ecologically neutral in the long-term, then small dam removals will often have a net ecological benefit. That is, the long-term benefits associated with the reconnection of downstream and upstream reaches and the restoration of lotic habitats in the former impoundment (Stanford et al. 1986, Bednarek 2001, Stanley et al. 2002) will often outweigh the relatively short-term ecological impacts of downstream sedimentation following removal. (201-202).
- (Thompson et al., 2005) sediments deliberately or accidentally released from reservoirs can cause pronounced reductions in benthic densities and diversity (Gray and Ward 1982, Marchant 1989, Doeg and Koehn 1994). It is possible that some of the apparent impacts on downstream benthic assemblages were caused directly by the floods in Stage 2, including the large December 2000 event, rather than by dam removal. For example, reductions in algal biomass observed at all sites in Stage 2 may have been caused by increased frequency of scouring flows. Floods affected all sites, however, whereas persistent reductions in macroinvertebrate density and diatom richness were only observed downstream of the dam, providing evidence of substantial dam removal effects. We cannot separate completely dam removal effects from flood effects because these factors interacted, as will often be the case. A major effect of dam removal is downstream sedimentation, and sediment can only be transported downstream by flows of sufficient power (200).
- (Thompson et al., 2005) In contrast, marked effects on both algae and macroinvertebrates occurred after the Stage 2 removal [of the dam], when downstream sediment transport increased dramatically (200).
- (Thompson et al., 2005) Macroinvertebrate and algal assemblages in riffles downstream of the dam are likely to recovery rapidly once fine sediment loads are reduced (Gray and Ward 1982, Marchant 1989, Doeg and Koehn 1994, Wood and Armitage 1997), especially given that species composition was largely unaffected for both groups. Geomorphic responses to dam removal remain poorly understood, however, and recovery rates are difficult to predict. Surveys downstream of the dam during 2004 suggest that bed sediments are yet to recover from the effects of dam removal (J. Pizzuto, University of Delaware, personal communication), and that sediment transport and deposition are continuing to have adverse effects on fish assemblages (R. J. Horwitz, Patrick Center for Environmental Research, personal communication). These results suggest that geomorphic adjustment, and therefore biological recovery, following small dam removal could take many years (201).
- (Thompson et al., 2005) Although algal biomass was lower after dam removal, there was no evidence of improvement in macroinvertebrate metrics up to 1 y after removal (201).

- (Thompson et al., 2005) It is unlikely that dam removal alone will cause a substantial improvement in downstream ecological integrity because the Manatawny dam had minimal downstream effects on water quality (Bushaw-Newton et al. 2002), and urban impacts continue Unabated (201).
- (Stanley and Doyle 2003) Because taking out dams creates “new” habitat, and because sediments are amenable to plant growth, dam removal may be a valuable tool for riparian restoration (Shafroth *et al*. 2002). However, widely available and often nutrient-rich sediment also represents prime habitat for invasion of weedy and exotic species that are generally considered undesirable (Shafroth *et al*. 2002). Observations of plant communities at several Wisconsin dam-removal sites show that species such as stinging nettle (*Urtica diocia*) and the invasive reed canary grass (*Phalaris arundinaceae*) are often abundant (CH Orr pers comm; Figure 5) (17).