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Metrics and Approaches for Quantifying Ecosystem Impacts and Restoration Success

By

Alexander Rubin

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Requirements for the degree of

Doctor of Philosophy

In

Landscape Architecture and Environmental Planning

in the

Graduate Division

Of the University of California, Berkeley

Committee in charge:

Professor G. Mathias Kondolf, Chair
Professor Mary Power
Professor John Radke

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Abstract

Metrics and Approaches for Quantifying Ecosystem Impacts and Restoration Success

By

Alexander Rubin

Doctor of Philosophy in Landscape Architecture and Environmental Planning

University of California, Berkeley

Professor G. Mathias Kondolf, Chair

Accurate quantification of ecosystem change is essential for effective environmental management. However, the selection of meaningful indicators of impacts to ecosystems and of benefits from restoration is not standardized. In this dissertation I investigate 1) the applicability of using sediment reduction as an indicator of the cumulative impacts of dams in the Mekong River basin; 2) review and evaluate the meaningfulness of common river restoration evaluation metrics such as macroinvertebrate diversity and richness in habitat heterogeneity projects, and 3) demonstrate the usefulness of prey availability as an indicator of restoration success in riparian restoration projects along the lower Colorado River.

1) The Mekong River, largely undeveloped prior to 1990, is undergoing rapid dam construction. Seven dams are under construction on the mainstem in China and 133 are proposed for the Lower Mekong River and tributaries. The question is what cumulative effect will these dams have on sediment movement in the watershed. There was a lack of data on sediment yields in some portions of the basin so we delineated nine distinct geomorphic regions, for which we estimated sediment yields based on geomorphic characteristics, tectonic history, and the limited sediment transport data available. We then applied the 3W model to calculate cumulative sediment trapping by these dams, accounting for changing trap efficiency over time and multiple dams on a single river system. Under a ‘‘definite future’’ scenario of 38 dams (built or under construction), cumulative sediment reduction to the Delta would be 51 percent. Under full build-out of all planned dams, cumulative sediment trapping will be 96 percent. That is, once in-channel stored sediment is exhausted, only 4% of the predam sediment load would be expected to reach the Delta. We then combined geomorphic assessments of the Mekong channel and delta with the 3W model's results of sediment trapping to forecast geomorphic change. We expect the biggest changes to occur along alluvial reaches, though stripping of thin sediment

deposits in bedrock reaches may also have significant consequences for benthic invertebrates, fishes, and other aquatic organisms dependent on the presence of alluvium in the channel. If all dams are built as proposed, the resulting 96% reduction in sediment supply would have profound consequences on productivity of the river and persistence of the delta landform itself and suggests that strategies to pass sediment through/around dams should be explored to reduce the magnitude and consequences of downstream sediment starvation. In this first case, we use sediment reduction as an indicator of watershed impairment. Though many complexities (e.g. oil, gas, and groundwater withdrawals, routing of sediment through deltas) influence coastal erosion, we found sediment reduction to be a meaningful worldwide indicator. We compiled sediment data from 24 worldwide deltas and results indicate a positive relationship of sediment reductions to deltas resulting in decreased rates of aggradation. In particular, sediment reductions of more than 80% are consistent in almost complete cessation in aggradation rates. The full-build scenario of Mekong dam building would result in 96% reduction in sediment delivery and we would then expect an almost complete cessation in sediment deposition in the delta.

2) In a search for accountability, the effectiveness of many large restoration programs has been evaluated using standard such as acres or length of stream restored per dollar, but this was recognized to be inadequate. Another common restoration metric is based on the common goal of enhancing ecosystems by creating more complex and varied habitats. Although widely implemented, there is little understanding of the success to date of such projects. There is also little agreement on the best approaches and metrics for quantifying success. We reviewed the methods of 26 peer-reviewed evaluation studies and investigated the influence of study design on evaluation results. Of the 26 studies, many did not implement rigorous study designs. For example, only 46% of the studies used quantitative measures of habitat, 62% included only one year of post-project monitoring, 46% used zero or one control (unrestored) sites, and 62% did not include reference (best potential ecological condition) sites. Studies that used more rigorous designs (e.g. sampled more years, measured habitat quantitatively) were more likely to find increased ecosystem diversity and richness in response to heterogeneity enhancement. More fundamentally, all studies used macroinvertebrate diversity and/or richness as the measure of ecological success. We question the logic of assuming that reach-scale diversity or richness is useful as a universal measure of ecosystem integrity. Monitoring and evaluation should first establish hypotheses and conceptual models based on watershed perturbations and set specific milestones towards a sustainable, dynamic, and healthy ecosystem. Restoration targets should be defined based on regional, historical, and analytical reference conditions and by conducting manipulative experiments that can help predict ecosystem responses to restoration actions. It is important to understand if habitat heterogeneity projects are succeeding, but it is not yet

possible to draw general conclusions. Metrics to evaluate performance of stream restoration projects need more rigor and should be tied to project specific goals. Generic metrics may yield misleading results.

3) Below Hoover Dam, riparian vegetation along the Colorado River was extensively cleared for agriculture. Thus, large areas of habitat were lost to clearing. Moreover, the functions of the ecosystem were compromised as the connections of the river to its floodplain were severed by levees, flow reduction by dams and diversions, channel incision, and groundwater pumping. Subsequently, native species declined, including the southwestern willow flycatcher (*Empidonax traillii extimus*) that nests along rivers in dense riparian thickets. The Lower Colorado River Multi Species Conservation Program (MSCP) was established in 2005 to re-create habitat for 26 species including the flycatcher, but the benefits of these restoration sites for target species have not been quantified. Many MSCP projects have involved extensive plantings of willow (*Salix exigua, S. gooddingii*] and cottonwood [*Populus fremontii*) on high terraces disconnected from the river by levees. MSCP projects goals are specified as acres of habitat, but to support functioning food webs, riparian ecosystems in arid regions require a subsidy of aquatic insects. We documented prey availability for the southwestern willow flycatcher in constructed habitats as an indicator of their potential to support the species. The number of aquatic insects, proportion of aquatic insects, total number of insects, and number of insect orders all decreased with distance from the river, and the decrease occurred within the first 100 meters from the river. The MSCP cottonwood-willow plantation (more than 500 m from the river) at Cibola National Wildlife Refuge had 86% fewer total insects ($p=0.055$), 97% fewer aquatic insects ($p=.032$), and only half as many insect orders ($p=0.015$) as sites adjacent to the river. In the plantation, only 16% of the insects were aquatic vs. 59% aquatic at the river's edge ($p=.063$). Our results suggest that restoration success (and the recovery of southwestern willow flycatcher) may be limited by prey availability and that future riparian plantings should be concentrated along the river or tributary channels. Southwestern willow flycatchers have not been nesting in MSCP plantations. Thus the metric of "acres restored" is inadequate to capture ecosystem function. More meaningful metrics would identify potential limitations in ecosystems (such as prey availability) so that habitat suitability and functionality can be assessed and adaptively managed.

Together, the chapters of this dissertation highlight different approaches and considerations for the quantification of ecosystem impacts and restoration success. The field of ecosystem quantification is still far from adopting universally appropriate indicators of change, but this dissertation seeks to highlight problems in current approaches, and to demonstrate useful models and approaches.

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Acknowledgments

I am indebted to many people at Berkeley but primarily I would like to thank Matt Kondolf and Mary Power for their support, advice, insights, and commitment to protecting and restoring rivers. Through our many conversations I have developed the diverse range of projects, approaches, and ideas that are contained in this dissertation. Many were unexpected, and I am most appreciative for the opportunity to investigate such a wide range of projects. The journey has been eye opening and I will spend many more years considering some of the insights that weren't written into my dissertation. I am also deeply indebted to John Radke and Joe McBride for their thoughtful feedback and assistance through the course of my studies.

On a personal level, I could not have done this without the love and support of my exceedingly wonderful wife, Rainbow. She is an inspiration and lights up my life. My family has been ceaselessly supportive through many endeavors and I am again appreciative of the support offered over these past years.

Introduction

Because the mechanics of ecosystem impacts on physical and biological systems are relatively well understood, ecosystem impacts can often be quantified by measuring fundamental components of the system such as changes in flow regime [*Poff et al.*, 1997], sediment regime [*Wohl et al.*, 2015], water quality [*Vörösmarty et al.*, 2010], vertical and lateral connectivity [*G.M. Kondolf et al.*, 2006], and changes in biotic communities [*Carignan and Villard*, 2002]. The selection of metrics for quantifying ecosystem impacts is still a complex endeavor, but in many cases meaningful metrics can be applied across many different systems and the same metrics are appropriate for a variety of causes.

By contrast, the mechanics of ecosystem recovery and the appropriate indicators for quantifying restoration success are far from clear. Monitoring and assessment of restoration projects has become increasingly common, but evaluation has not been systematic or coordinated [*Bernhardt et al.*, 2005; *Wohl et al.*, 2005]. Beginning in the 1990's, a growing number of river restoration scholars, practitioners, funders, and regulators began advocating for monitoring and evaluation of completed projects, since few projects had any monitoring and evaluation at all [*Bernhardt et al.*, 2005; *Bernhardt et al.*, 2007; *G.M. Kondolf*, 1995; *G.M. Kondolf et al.*, 2007; *Palmer et al.*, 2007]. When conducted, reviews of restoration projects have found that designers deem a high percentage of projects "successful", but rarely have these judgments been based on specific ecological indicators [*Alexander and Allan*, 2007; *Jähnig et al.*, 2011]. Recent studies treat restoration more critically, questioning the mentality that drives restoration [*Katz*, 1992], when the "do nothing" approach is appropriate for letting the river heal itself [*G.M. Kondolf*, 2011], how restoration interacts with carbon cycling [*Madej*, 2010], the sustainability of restoration projects [*Palmer et al.*, 2005], and the societal benefits of restoration [*Dufour and Piégay*, 2009]. Although methods of achieving certain technical objectives such as how to remeander streams [*Shields et al.*, 2003], create riffles and pools [*Newbury*, 1995], and stabilize banks [*Li and Eddleman*, 2002] have occurred rapidly in recent decades, there are ongoing challenges to 1) define appropriate restoration objectives, 2) quantify success in achieving watershed-scale, long-term, or multi-organism objectives, 3) integrate small-scale projects to achieve watershed-scale goals, and 4) effectively meet multiple objectives including environmental flows, habitat, water quality, recreation, and economic opportunity, and 5) how to evaluate results from standardized monitoring protocols.

In Chapter 1 we use sediment reduction as an indicator of the downstream impact of dams in the Mekong basin. The theoretical basis for sediment reduction as an indicator has been well documented across different river systems [*Wohl et al.*, 2015]. The Mekong River is undergoing rapid dam construction. Seven dams are under construction on the mainstem in China and 133 are proposed for the Lower Mekong River and tributaries. The question is what cumulative effect will these dams have on sediment movement in the watershed. There was a lack of data on sediment yields in some portions of the basin so we delineated nine distinct geomorphic regions, for which we estimated sediment yields based on geomorphic characteristics, tectonic history, and the limited sediment transport data available. We then applied the 3W model to calculate cumulative sediment trapping by these dams, accounting for changing trap efficiency over time and multiple dams on a single river system. Under a ''definite future'' scenario of 38 dams (now built or under construction), the cumulative sediment reduction to the Delta would be 51%. Under full build-out of all planned dams, cumulative sediment trapping will be 96%. That is, once inchannel stored sediment is exhausted, only 4% of the predam sediment load would be expected to reach the Delta. We then combined geomorphic assessments of the Mekong channel and delta with the 3W model's results of sediment trapping to forecast geomorphic change and the potential implications for the ecosystem. We expect the biggest changes to occur along alluvial reaches, though stripping of thin sediment deposits in bedrock reaches may also have significant consequences for benthic invertebrates, fishes, and other aquatic organisms dependent on the presence of alluvium in the channel. If all dams are built as proposed, the resulting 96% reduction in sediment supply would have profound consequences on productivity of the river and the persistence of the delta landform itself and suggests that strategies to pass sediment through/around dams should be explored to reduce the magnitude and consequences of downstream sediment starvation. Portions of the research included in chapter 1 have been previously published ([*G. M. Kondolf et al.*, 2014; *Rubin et al.*, 2015] with contributions from Matt Kondolf, Toby Minear, and Paul Carling.

By contrast, the theoretical and practical considerations for quantifying restoration are much more complex. One fundamental problem is that the metric that is used to quantify success can be achieved through many pathways. In practice, this problem of equifinality means that one might "restore" the sediment load of the Mekong River in several ways and not all ways will be functionally equivalent. Increasing erosion in tributary river basins downstream of dams by clearing forests may recover the total load of the river, but the caliber of that load will likely be different. Essentially, while the question of impacts could be assessed with the simple metric of total annual sediment load, quantifying restoration of sediment may require consideration of the size, timing, mobility, sorting, and chemistry, and habitat suitability of that sediment, a much more difficult problem.

In Chapter 2 we investigate the complexities of quantifying ecosystem restoration by focusing on stream habitat heterogeneity enhancement projects and evaluations. Managers and scholars are increasingly interested in quantifying the effectiveness of ecosystem restoration projects, yet appropriate metrics are challenging to identify. Many restoration projects are implemented to benefit specific species, yet because populations may take decades or longer to respond to restoration activities, and because populations of target species may be strongly affected by factors unrelated to restoration actions, measures of populations are often not appropriate ways to promptly evaluate restoration. Therefore, many monitoring programs use surrogate, and potentially irrelevant, metrics to evaluate restoration project performance. Stream restoration projects commonly attempt to enhance ecosystems by creating more complex and varied habitats. Although widely implemented, there is little understanding of the success to date of such projects. There is also little agreement on the best approaches and metrics for quantifying success. We reviewed the methods of 26 peer-reviewed evaluation studies and investigated the influence of study design on evaluation results. Most of the 26 studies did not implement rigorous study designs: only 46% of the studies used quantitative measures of habitat, 62% included only one year of post-project monitoring, 46% used zero or one control (unrestored) sites, and 62% did not include reference (best potential ecological condition) sites. Studies that used more rigorous designs (e.g. sampled more years, measured habitat quantitatively) were more likely to find increased ecosystem diversity and richness in response to heterogeneity enhancement. More fundamentally, all studies used macroinvertebrate diversity and/or richness as the measure of ecological success, though the meaningfulness of reach-scale diversity/richness as an indicator of ecosystem condition is not clear. While protecting biodiversity is indeed an important societal goal, some systems may support only a few specialist/endemic species, and increasing diversity in that system (at the detriment of endemics) hinders regional conservation goals. Monitoring and evaluation should first establish hypotheses and conceptual models based on watershed perturbations and set specific milestones towards a sustainable, dynamic, and healthy ecosystem. Restoration targets can be defined based on regional, historical, and analytical reference conditions and by conducting manipulative experiments that can help predict ecosystem responses to restoration actions.

Societal values, as much as scientific theory, set conservation and restoration priorities. Societies may value and protect certain species or regions while neglecting others. The large canyon of the Colorado River was considered a wasteland by early visitors and is now awarded special status and protected as Grand Canyon National Park [*Pyne*, 1999]. This question of values and assumptions is particularly evident in efforts to quantify and evaluate conservation and restoration. For example, restoration evaluations [e.g. *Moerke et al.*, 2009; *Purcell et al.*, 2002; *Tullos et al.*, 2009] commonly use habitat assessment metrics such as the EPA's Rapid Bioassessment Protocol [*Lazorchak et al.*, 1998] and Ohio's Qualitative Habitat Evaluation Index [*Rankin*, 2006] that have been developed to provide a snapshot of ecological condition in streams across the country. In Chapter 2 we critique the use of such metrics for restoration evaluation, and question the meaningfulness of universal metrics to quantify ecosystem conditions. In the EPA's Rapid Bioassessment Protocol, a stream receives more points for a stable streambed, for a high frequency of riffles, and for stable banks (along with seven other metrics). To protect regional and global biodiversity we must maintain a diversity of stream types, not simply engineering our preferred type. To date it is not yet possible to draw general conclusions on whether habitat heterogeneity projects are succeeding. Evaluations need more rigor and connection to project specific goals, rather than relying on generic metrics such as macroinvertebrate diversity and richness. The research in Chapter 2 was performed with contributions from Blanca Rios-Touma and Mary Power and review and insights from Bill Dietrich and Jennifer Natali.

In Chapter 3 we demonstrate a meaningful and feasible evaluation approach relying upon prey availability as an indicator. Surrogate evaluation metrics include invertebrate abundance and community diversity [*Muotka et al.*, 2002; *Pik et al.*, 2002], habitat complexity [see *Miller et al.*, 2010; *Palmer et al.*, 2010], and persistence of created habitat features [*Schmetterling and Pierce*, 1999]. Habitat construction is a common approach to restoration, though few studies have evaluated the effectiveness or lasting success of such projects. The lower Colorado River Multi Species Conservation Program established willow-cottonwood plantations to provide habitat for threatened and endangered insectivores such as the southwestern willow flycatcher (*Empidonax traillii extimus*). Therefore, insect (prey) availability has potential as a useful measure of habitat function. Riparian restoration sites have been planted more than 2 km from the river and sustained through irrigation. We used sampled insect communities in restored, control, and reference sites along the lower Colorado and Bill Williams Rivers in Arizona. Sites farther than 100m from the river's edge had: 1) fewer insects, 2) fewer aquatic insects, 3) a lower percentage of aquatic insects than sites along the river's edge, and 4) less ordinal richness. Results suggest that unless habitat construction projects consider physical and biological processes and context, essential habitat functions may not be achieved. The research in Chapter 3 was performed with contributions from Matt Kondolf, Blanca Rios-Touma, Mary Power, Parsa Safarinia, and Jennifer Natali.

Together, the chapters of this dissertation highlight different approaches and considerations for the quantification of ecosystem impacts and restoration success. The field of ecosystem quantification is still far from adopting universally appropriate indicators of change, but this dissertation seeks to highlight problems in current approaches, and to demonstrate useful models and approaches.

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Chapter 1: Mekong River Dams: Cumulative Sediment Starvation and Anticipated Geomorphic Impacts

Portions of the research included in chapter 1 have been previously published [Kondolf et al., 2014; Rubin et al., 2015]. Citations for Figures and Tables are included as appropriate.

Abstract

The Mekong River, largely undeveloped prior to 1990, is undergoing rapid dam construction. Seven dams are under construction on the mainstem in China and 133 are proposed for the Lower Mekong River and tributaries. The question is what cumulative effect will these dams have on sediment movement in the watershed. There was a lack of data on sediment yields in some portions of the basin so we delineated nine distinct geomorphic regions, for which we estimated sediment yields based on geomorphic characteristics, tectonic history, and the limited sediment transport data available. We then applied the 3W model to calculate cumulative sediment trapping by these dams, accounting for changing trap efficiency over time and multiple dams on a single river system. We build on previous work by using information on reservoir storage and location that was not previously available, by supplementing the sparse sediment transport data with information on local factors that can influence sediment supply and transport, and by applying a model that accounts for temporal effects and spatial interactions in reservoir storage. Under a ''definite future'' scenario of 38 dams (built or under construction), cumulative sediment reduction to the Delta would be 51 percent. Under full build-out of all planned dams, cumulative sediment trapping will be 96 percent. That is, once in-channel stored sediment is exhausted, only 4% of the predam sediment load would be expected to reach the Delta. We then combined geomorphic assessments of the Mekong channel and delta with the 3W model's results of sediment trapping to forecast geomorphic change. We expect the biggest changes to occur along alluvial reaches, though stripping of thin sediment deposits in bedrock reaches may also have significant consequences for benthic invertebrates, fishes, and other aquatic organisms dependent on the presence of alluvium in the channel. If all dams are built as proposed, the resulting 96% reduction in sediment supply would have profound consequences on productivity of the river and persistence of the delta landform itself and suggests that strategies to pass sediment through/around dams should be explored to reduce the magnitude and consequences of downstream sediment starvation. In this first case, we use sediment reduction as an indicator of watershed impairment. Though many

complexities (e.g. oil, gas, and groundwater withdrawals, routing of sediment through deltas) influence coastal erosion, we found sediment reduction to be a meaningful worldwide indicator. We compiled sediment data from 24 worldwide deltas and results indicate a positive relationship of sediment reductions to deltas resulting in decreased rates of aggradation. In particular, sediment reductions of more than 80% are consistent in almost complete cessation in aggradation rates. The full-build scenario of Mekong dam building would result in 96% reduction in sediment delivery and we would then expect an almost complete cessation in sediment deposition in the delta.

Introduction

Dam Impacts

Dams have multiple environmental impacts, including transient impacts of construction and reservoir filling (including noise, dust, social disruption of construction boomtowns, and displacing affected populations), and the longer-term hydrologic, water quality, and ecological changes resulting from converting flowing (lotic) to still (lentic) water environments, changes in sediment load and channel form, reservoir-induced seismicity, short and long-term, economic and social effects of displacing riparian populations, and alterations of river ecology [Petts, 1984; Williams and Wolman, 1984; World Commission on Dams (WCD), 2000]. By blocking migration of fish, dams have led to extinction (or large population reductions) of migratory fish species in many rivers [Dudgeon, 2000], and waters released from reservoirs often suffer water quality problems resulting from the interaction of nutrients, chemicals, and sunlight in standing water [WCD, 2000].

Reservoirs trap all the bedload (the coarse sand and gravel moved along the river bed) and a percentage of the suspended load (the sand and finer sediment carried in the water column, held aloft by turbulence). The percentage of the suspended sediment trapped by a reservoir can be estimated as a function of the ratio of reservoir storage capacity to annual inflow of water [Brune, 1953]. The supply of sediment to the river downstream is thereby reduced. Either sediment surplus or sediment deficit are possible below dams, depending on the relative change in sediment supply and transport capacity [Schmidt and Wilcock, 2008], but most commonly the reach downstream of the dam is characterized by sediment-starved, or ''hungry'' water, which can erode the bed and banks to regain some of its former sediment load [Kondolf, 1997]. These erosive flows commonly induce incision, undermine bridges and other infrastructure, and coarsen the bed [Kondolf, 1997], and fundamentally alter aquatic food webs [Power et al., 1996].

Most large rivers in the world are experiencing decreased sediment loads due to dam-induced sediment starvation. In the two millennia prior to widespread dam construction, human activities such as forest clearing and cultivation increased erosion and sediment delivery to the oceans (Leopold 1921, Leopold 1923, Wolman and Schick 1967, Milliman and Syvitski 1992, Syvitski 2008). Worldwide, widespread dam construction has reversed this historical trend, and substantial reductions in the delivery of sediment to the oceans are now occurring in many of the world's rivers (Milliman and Syvitski 1992). The Mekong, however, differs from other large Asian rivers, having produced a relatively consistent sediment yield over the past three thousand years (Ta et al. 2002), reflecting relatively modest levels of development that prevailed until very recently.

The consequence of sediment-load reduction in combination with delta subsidence, both natural and accelerated, discharge control, and channelization, is to accelerate shoreline erosion, threaten the health and extent of mangrove swamps and wetlands, increase salinization of cultivated land, and put human populations at risk of costly disasters (Syvitski 2008). Whereas eustatic sea-level rise associated with global warming has received much focus and interest in recent years, in many deltas, the land surface that meets the water has been subsiding more rapidly in recent years, as dam building reduces sediment supply needed for deposition on the delta plain, distributary channels are stabilized and dyked so that sediment-laden floodwaters can no longer disperse over the floodplain, and petroleum and groundwater extractions induce subsidence. Deltas that develop dense cities and industrial infrastructure become less resilient to tsunamis and storm-induced coastal surges. Lives and wetlands at risk today in coastal regions will be even more at risk in the future (Syvitski 2008). The cumulative impacts of sea-level rise, sediment starvation from reservoir trapping and instream mining of construction aggregate, channelization of delta distributary channels, and groundwater extraction are common to many of the world's major rivers (Bucx et al. 2010), and the subsequent coastal erosion is globally consistent (Table 1).

The Mekong River

The Mekong River is unique among the world's great rivers in the size of the human population supported by its ecosystem. Approximately 60 million people (mostly in Cambodia and Vietnam) use fish from the Mekong as the primary source of protein in their diet [Baran and Myschowoda, 2009]. The river remained largely unregulated through most of the 20th century because of wars in Indochina and lack of development in remote provinces of China. With peace in the Lower Mekong River Basin (LMRB) countries and economic development in China, this is rapidly changing, and the Mekong River system is undergoing extensive dam construction throughout the basin for hydroelectric generation [Grumbine et al., 2012]. While there are numerous diversions for irrigated agriculture throughout the basin, and some of these involve storage impoundments that would trap sediment, we found no comprehensive inventory of irrigation impoundments. However, most are small diversions directly from river channels, and most are concentrated in the relatively low relief Khorat Plateau of Thailand [Hoanh et al. 2009]. In any event, the impact on the Mekong River system of the projected hydroelectric dams will vastly exceed that of the existing irrigation infrastructure. In the upper Mekong in China (where it is known as the Lancang), seven hydro- electric dams have been built or are under construction on the mainstem. In the Lower Mekong and its tributaries, 133 hydroelectric dams are built, under construction, or planned, including 11 on the Lower Mekong mainstem, based on data compiled by the Mekong River Commission (MRC).

Draining a narrow catchment originating on the Tibetan Plateau, the Mekong flows through bedrock canyons in Yunnan Province of southwest China and along the border with Burma. Downstream of the Chinese border, the lower Mekong flows through Laos, Thailand, Cambodia, and Vietnam, debouching in the Mekong Delta (Figure 1). The basin drained by the Mekong River has a complex geologic history resulting from the Tertiary collision of the Indian and Eurasian plates, consequent deformation and opening of large strike-slip fault-controlled basins, and subsequent volcanism [Carling, 2009a; Gupta, 2009]. The Mekong River drains a total of about 800,000 km^2 and has an average discharge (at its mouth) of about 15,000 m^3s^{-1}, with predictable 20-fold seasonal fluctuation from dry season (November-June) to wet (July-October) [Gupta et al., 2002; Adamson et al., 2009], with the monsoon-driven high flow accounting for 75% of the annual flow (Piman et al. 2013). Gupta et al. (2002), Gupta and Liew (2007), Carling (2006), Gupta (2008), and Carling (2009a) described the geomorphic framework of the Mekong River (Table 2), noted differences in geomorphic characteristics of reaches of the Lower Mekong, and to some extent, explored how reduced sediment supplies might affect different reaches.

The predam sediment flux of the Mekong River into the South China Sea has been estimated at approximately 160 million tonnes per year (Mt yr^{-1}), of which about half was produced by the upper 20% of the basin area, the Lancang drainage in China [Milliman and Meade, 1992; Milliman and Syvitski, 1992; Gupta and Liew, 2007; Walling, 2008]. However, it is worth noting that this widely used estimate has been challenged as too high based on calculated sediment flux at Khong Chiam [Wang et al., 2011; Liu et al., 2013], and as too low based on detailed studies of the sand fraction, which suggest that sand has been systematically undersampled and imply that the true transport rate is larger [Bravard et al., 2013a, 2014]. In part, this may reflect the fact that sediment sampling has, until 2012, been focused almost exclusively on suspended sediment, so the values discussed here are values for suspended sediment, neglecting bedload [Walling, 2009], which would be preferentially trapped by dams.

Flow alteration from existing and proposed dams is expected to be more modest than the sediment trapping. Under the 41-dam 'definite future' scenario, and a full-build scenario of 136 dams in the lower Mekong, Piman et al. (2013) and Mekong River Commission (2010) predicted a dry season flow increase of 22% and 29% for definite-future and full-build scenarios, respectively, at the Kratie station. Wet season flows were predicted to decrease 4% and 13% for definite-future and full-build scenarios, respectively. Based on independent modeling, Lauri et al. (2012) also predicted similarly modest wet season flow reductions. These changes in flow regime may have significant impacts for the aquatic ecosystem and especially the fishery of Tonle Sap (Lamberts and Koponen 2008, Baran and Myschowoda 2009), but the small reduction in wet season flow is unlikely to substantially change the sediment transport capacity of the Mekong.

The Mekong Delta

One of the largest in the world is the Mekong Delta, built out over the past 8000 years by deposition of river sediments (Stattegger et al. 2013), with a subaqueous extent of about 94,000 km^2, of which 49,100 km^2 is exposed subaerially (Ericson et al. 2006; Hori and Saito 2008). On the delta floodplains of Vietnam, Hung and colleagues found that deposition rates averaged 6.9 kg m^{-2} or approximately 6 mm y^{-1} with both turbidity and deposition dropping substantially with distance from the channels (Hung et al. 2014a, 2014b). Lower in the delta, Szczuciński et al. (2013) identified three zones of contemporary subaqueous sediment deposition: (1) The subaqueous prodelta west and south from the Ca Mau Peninsula, which is mud-dominated, organic-rich, and rapidly accumulating sediment (up to 1.5 cm y^{-1}). (2) South of the river mouths is the main sink for Mekong River bedload. (3) Farther offshore is a finer grained depositional zone with accumulation rates of 0.3 – 0.4 cm y^{-1}, dominated by muddy sands rich in biogenic carbonate. Szczuciński's sediment budget proposes that the subaqueous delta front stores ~50% of the Mekong River's fine-grained sediments while ~25% are retained in the subaerial region of the delta (including the Tonle Sap Lake), and ~25% accumulates on the shelf around the Ca Mau Peninsula.

Anthony et al. (2012) used sequential satellite images to analyze coastal retreat in the Mekong delta from 2003 and 2011, finding an average of 4.4 m y^{-1} of coastline retreat across the entire delta, with higher rates of 12 m y^{-1} on the Ca Mau peninsula. Given the stable sediment supply and the growth of the delta during the last ~6000 years, Anthony et al. (2012) attributed this recent coastal retreat to the reduced sediment supply caused by massive extraction of sand and gravel from the river channel for construction aggregate (Bravard et al. 2013, 2014), and by levees and channel straightening in the delta, which increase flow velocity and loss of sediment to deeper waters. The Ca Mau peninsula appears to be particularly sensitive to these changes, as it is fed by longshore transport of sediment from the river mouth (Anthony et al. 2012). To date, there has been a lack of analysis (and for that matter, a lack of data upon which to base analysis) to understand how the Mekong delta is likely to respond to future sediment starvation. With better predictions of sediment starvation now available, it is clear that sediment starvation effects are likely to be severe, and thus there is an urgent need to draw upon available information for the Mekong delta and analogous systems to make initial projections of likely impacts and identify critical data needs.

Methods

Assessing Cumulative Sediment Starvation

How will the many dams planned and being constructed alter the sediment load of the Mekong? What will be the likely cumulative reduction in sediment load? Using 160 Mt yr^{-1} as the average annual suspended load of the entire Mekong and assuming that about half of this load is derived from the Lancang basin [Walling, 2008], the ongoing construction of seven dams on the Lancang (with cumulative trap efficiencies of about 83%) means that over 40% of the natural sediment load of the Mekong will be lost in the reservoirs of the Lancang [Walling, 2011]. Thus, the sediment load of the Lower Mekong River will consist mostly of sediment derived from sources within the LMRB itself. To predict how dams in the LMRB will likely affect sediment loads requires an understanding of the relative contributions of sediment from individual subbasins and how future dams will affect these contributions.

We build on previous work by using information on reservoir storage and location that was not previously available, by supplementing the sparse sediment transport data with information on local factors that can influence sediment supply and transport, and by applying a model that accounts for temporal effects and spatial interactions in reservoir storage. Our study provides more accurate estimates than prior studies because (1) we utilized an updated database for locations of the planned reservoirs, (2) we used total storage estimates for the reservoirs (data not previously available), (3) we treated every dam in the network individually and calculated the sediment deficit for each channel segment, (4) we used the limited sediment transport data as only one factor in estimating sediment yields, relying also on the geologic and topographic characteristics of the regions to derive geomorphically based estimates of sediment yield, and (5) we estimated reservoir trapping under multiple dam building scenarios and accounted for changes in trap efficiency over time as reservoirs accumulate sediment.

To develop a detailed depiction of reservoir sedimentation over time, we applied the 3W model [Minear and Kondolf, 2009], a network model that accounts for multiple reservoirs on a given river and changing trap efficiencies as reservoirs fill, to estimate the sediment trapping by various combinations of dams. This first required estimates of sediment yields from various tributary drainages in the LMRB, then application of the 3W model for dams within the context of these estimated sediment yields. To assess potential effects of dams in the LMRB, whose tributaries historically contributed about 80 Mt yr^{-1} (i.e., the downstream half of the 160 Mt yr^{-1} total basin load), we first sought to allocate the 80 Mt yr^{-1} to different parts of the LMRB. We conducted our analysis in three stages: (1) delineation of geomorphic regions in the lower Mekong basin, (2) determination of sediment yield by geomorphic region, (3) application of the 3W model with estimated sediment trapping for reservoirs based on Brune's [1953] empirical relationship.

Delineating Geomorphic Regions and Estimating Sediment Yields

As a basis for estimating sediment contributions, we delineated distinct geomorphic regions based on geologic history and geomorphic characteristics. Sediment yields are fundamentally controlled by tectonic uplift, climate, lithology, and land use [Syvitski and Milliman, 2007]. The underlying structural fabric of the basin controls the landscape of the Mekong Basin and the elevation of the highlands that form major sediment provenances can be related to distinct episodes of "plate-scale" tectonic activity that occurred from the late Triassic (~200 million years BP) onwards. In addition to the Lancang basin upstream, we delineated eight distinct geomorphic regions in the Lower Mekong River Basin, for nine regions in total (Table 3, Figure 1).

Existing sediment transport data (compiled by the MRC) are insufficient in and of themselves as a basis for estimating sediment yields, because the number of data points and measuring period are insufficient and there are significant questions with data reliability for many stations [Walling, 2008]. Moreover, some important regions (such as the basins of the Sre Pok, Se San, and Se Kong, the so-called "3-S" basins) have had no sediment data available. Thus, we used the geomorphic-region approach, as it offered a consistent and defensible framework. We first assessed the likely relative sediment yield of each geomorphic region based strictly on geologic and geomorphic characteristics, such as uplift history [Clift et al., 2004] and land-form relief, as well as precipitation. We also reviewed previous studies of Mekong River channel geomorphology and sediment transport [including Gupta et al., 2002; Carling, 2005; Gupta and Liew, 2007; Walling, 2008; and Sarkkula et al., 2010], along with sediment data available from the MRC, to provide further insights into likely sediment yields from distinct geomorphic regions. We then assigned relative sediment yields to each geomorphic region, such that the predam sediment yields would sum to 80 Mt yr^{-1}, the total annual average sediment yield produced by the LMRB predam.

Uncertainty in Predicting Sediment Yieds

This method of predicting sediment yield has some underlying sources of uncertainty. First, because a single estimate of sediment yield is applied to an entire geomorphic region, local variability is missed. Second, without detailed sediment transport data, estimates of sediment yield are, at best, rough estimates only, based on an assumed total sediment contribution from the LMRB of 80 Mt yr^{-1}. Third, this approach ignores potential conveyance losses as sediment is transported down the drainage network, a disadvantage partially offset by the fact that the sediment load apportioned to the contributing catchment is based on the downstream sediment load, so it already reflects conveyance losses.

Estimating Sediment Trapping in Existing and Proposed Reservoirs

We applied the 3W model of Minear and Kondolf [2009] to calculate how sediment trapping in individual reservoirs will change the sediment transport along tributaries and the mainstem Mekong throughout the entire LMRB. This allowed us to assess how sediment loads in different reaches of the LMRB will change from predam conditions under different reservoir development scenarios. We used a spatial database of existing and proposed dam sites provided by the MRC to locate each project and query data detailing mean annual discharge, contributing watershed area, full supply level and bottom elevation, and expected dam completion date. We identified and, in collaboration with MRC staff, corrected seven problematic entries in the database, such as incorrect coordinates for Nam Kong 2 and Xe Kong 3d. For projects without data on the year constructed/planned, we assumed the projects would be completed in year 2020.

The Brune [1953] curve predicts trap efficiency from the ratio of total reservoir storage to annual average inflow. While actual trap efficiency is influenced by reservoir geometry, seasonal patterns of runoff and reservoir storage, dam design and operation, and other factors, information on these factors may be difficult to obtain for many reservoirs. Thus, the Brune curve is widely used to provide first-cut trap efficiency estimates from the more readily available reservoir storage capacity and annual runoff data [Morris and Fan, 1998]. However, it is important to use total storage estimates for the reservoir volume instead of active storage, because it is total storage that influences the processes of sediment deposition within the reservoir. In fact, ''dead storage,'' the portion of the reservoir volume below the active storage layer, is commonly used in the design context as a ''buffer'' against reservoir sedimentation affecting dam operations. The difference between total and active storage can be significant, especially in dams with high-level intakes.
To quantify the total storage capacity of each reservoir, we used project reports when available [e.g., Mekong River Commission (MRC), 2011], drew upon MRC staff estimates of total storage calculated from overlaying the inundated areas (at the full reservoir level) onto topography, and consulted extensively with current and former MRC staff. Ultimately, we could not obtain total storage estimates for nine small reservoirs. For these, we used our data set of reservoirs with both active and total storage estimates, calculated a best-fit line relating total storage as a function of active storage, and used this relation to estimate total storage for the missing nine reservoirs. The nine reservoirs for which we used the best-fit method were all smaller than 0.5 km^3 in capacity, so they would have a limited impact on total basin sediment trapping in any event.

Using ESRI ArcMap 10.1 software, a geographic information system (GIS), we overlaid dam coordinates on a stream channel layer obtained from the International Water Management Institute website (http://www. iwmi.cgiar.org/) and the U.S. Geological Survey's 1 km resolution GTOPO30 Global Digital Elevation Model (DEM). With these GIS layers, we constructed a dendritic network diagram to identify, for each reservoir, which other reservoirs were planned or constructed upstream. We calculated predam sediment load (Qs) for each reservoir based on the contributing watershed area for each project and apportioning that area among geomorphic units with defined sediment yields.

The 3W model [Minear and Kondolf, 2009] is an iterative tool that simultaneously calculates reservoir sedimentation, trap efficiency, and reservoir storage volume, for each individual reservoir for each year. The trap efficiency will decrease as the reservoirs fill with sediment. Moreover, as additional reservoirs are built in a drainage basin, upstream reservoirs will trap sediment that otherwise would have been delivered to downstream reservoirs, so in multiple-reservoir systems, the upstream reservoirs slow the rate at which downstream reservoirs fill. We conducted three runs of the 3W model to estimate reservoir sedimentation for: (1) the entire set of 133 existing and planned reservoirs, (2) for a set of 38 high likelihood reservoirs designated by the Mekong River Commission as the 'Definite-Future' scenario, and (3) for the entire set of 133 dams without the eleven mainstem dams proposed for the Lower Mekong River (Figure 1). Following Brune [1953], we estimated theoretical trap efficiencies for suspended load from total storage capacity and mean annual runoff. We used the following algebraic approximation to the equation based on Brune's median curve of trap efficiency for each reservoir.

$$TE = 1 - \frac{0.05}{\sqrt{CI}}$$

Where *TE* is trap efficiency (expressed as a decimal percent) of a reservoir; and *CI* is the capacity-inflow ratio change calculated as:

$$CI = \frac{Vr}{Q}$$

Where *Vr* is the total storage volume of the reservoir (km^3) and Q is the mean annual discharge at the reservoir site ($km^3\ y^{-1}$).

Following Minear and Kondolf [2009], we constructed a coupled worksheet model to calculate annual values for trap efficiency, reservoir sediment deposition, and reservoir volume. Each year, the trap efficiency decreases as the volume of deposited sediment reduces the storage volume of the reservoir. To account for sediment trapping in upstream reservoirs, the inflowing sediment load *S* was calculated based on upstream reservoir trapping, if upstream reservoirs were present

$$S = Qs - \sum Vs$$

Where *Qs* is the predam annual sediment discharge (km^3), and $\sum Vs$ is the sum of sediment trapped in all upstream reservoirs calculated as

$$Vs = TE \times S$$

The model then calculated a new reservoir storage volume *Vr*, and used that to calculate a new trap efficiency *TE* for the following year. This procedure was done through the year 2420 (i.e., about 400 years). To convert estimates of sediment yield (Mt yr^{-1}) into volumes (km^3 yr^{-1}) of reservoir sedimentation, we assumed a reservoir sediment density of 960 kg m^{-3}, the average value from Dendy and Bolton [1976]. In addition to calculating trap efficiency, we conducted a sensitivity analysis to determine the effect of the Brune curve selected on the results. We used the same alpha values assumed by Kummu et al. [2010], i.e., alpha = 0.76 for the upper curve and alpha = 1.24 for the lower curve. (The middle curve, reflected in the model's results already reported, reflects an alpha = 1.0.).

Geomorphic Impacts of sediment trapping on the channel and delta

To assess likely downstream channel response to reduced sediment loads, we drew upon prior geomorphic work on the Mekong River basin by Adamson (2001), Gupta (2004), Gupta and Liew (2007), Gupta (2008), and Carling (2009a) to characterize channel reaches in terms of their likely response to sediment starvation. To assess likely response of the delta to reduced sediment loads, we compiled available data for other deltas as reported in the literature, and systematically analyzed data such as degree of hydrologic alteration, percentage reduction in sediment supply, documented historical subsidence rates, and wave energy, as well as reported responses such as accelerated coastal retreat, land loss, and changes in aggradation rate. Based on these analogous case studies, we made initial predictions for probable response of the Mekong delta to the reduction of its sediment supply. Because of difficulties in scaling reported changes in coastal accretion/erosion (reported in length) and rates of land loss (reported in area) among deltas, we focused on rates of vertical accretion (aggradation), as this variable would arguably be more comparable among deltas.

Results

Sediment Yields by Geomorphic Region in the Mekong River Basin

As noted by Clift et al. [2004, p. 20], competing controls on erosion rates include ''topography, modern tectonic rock uplift rates and climate, especially precipitation.'' They found ''a relatively good correlation between rates of tectonic deformation and erosion, but no strong link with seismicity,'' with the highest erosion rates in the ''steep margins of the Tibetan Plateau in regions of active tectonic strain'' [Clift et al., 2004, p. 22]. Yields from the upper Mekong River Basin (Lancang) are clearly the highest in the basin, with predam sediment yields of about 450 t km^{-2} yr^{-1}, based on long-term suspended sediment records at Chiang Sean. Within the LMRB, heavy precipitation in the Kontum Massif and Central Highlands of Vietnam, combined with the region's recent and ongoing uplift documented by apatite fission track analysis [Carter et al., 2000] results in the next-highest erosion rates, which we estimated to be 280 and 290 t km^{-2} yr^{-1} for the Kon Tum Massif and the Tertiary Volcanic Plateau, respectively (Table 3), and as reflected in the active incision of river channels. These two regions are drained primarily by the Sre Pok, Se San, and Se Kong rivers, which are known informally within the Basin as the ''3-S'' rivers, and which have been identified as important sediment contributors to the mainstem [e.g., P. T. Adamson, An exploratory assessment of the potential rates of reservoir sediment in five Mekong mainstream reservoirs proposed in Lao PDR, Unpublished report, Mekong River Commission, Vientiane, 2009], although no sediment data for them have been available. The Tonle Sap basin receives substantial sediment in backwater flooding upstream from the Mekong River mainstem at flood stage and is actually net depositional [Tsukawaki, 1997; Kummu et al., 2005], so we assigned a zero sediment yield. The Delta is also (by its nature as a delta) a sediment sink, so also has a zero sediment yield.

Recall that these sediment yields are based on apportioning a total of 80 Mt yr^{-1} contributed to the river from the Lower Mekong River Basin among geomorphic provinces, and they do not account for the well- known inverse relationship between drainage area and sediment yield [Walling, 1983], nor conveyance losses downstream. Thus, actual sediment yields by subbasin may have been higher than implied by the exercise of apportioning 80 Mt yr^{-1} amongst the potential source areas. The model could be viewed as overcoming the need to incorporate conveyance losses because the sediment load apportioned to the contributing catchment is based on downstream sediment load, which already reflects conveyance losses. However, because conveyance losses are not taken into account, the sediment loads estimated for upstream stations will underestimate the true loads passing these points. An important corollary is that amount of sediment deposited in the various reservoirs will likely be underestimated, reservoir life overestimated, and trap efficiencies decreased more rapidly than predicted.

Sediment Trapping in Reservoirs

As trap efficiency is a function of the capacity/inflow ratio, the largest trap efficiencies were found for tributaries with relatively large reservoirs. Twenty-four reservoirs had initial trap efficiencies greater than 95%, with many more above 90%. At the other extreme, reservoirs that are small relative to the annual inflow will have negligible trapping. Our algebraic approximation of the Brune curve has an x-intercept for the capacity/inflow ratio of around 0.0025, which means that we account for no sediment trapping when $CI < 0.0025$. In our data set, ten of the dams fall below this cutoff, with negligible trap efficiency and thus were ignored in our model. Even dams with trap efficiencies far less than 90% can have a significant effect on basin sediment yield depending on the location within the channel network. Sambor Dam would have an initial trap efficiency of 48%. Under the ''Definite Future'' scenario, about 77 Mt yr^{-1} of sediment would be delivered to Sambor from upstream of which it would trap about 38 Mt yr^{-1}, significantly affecting sediment delivery to downstream reaches. We modeled the cascade of dams in the upper Mekong (Lancang) in a separate 3W model, resulting in a collective trap efficiency of 83% (of the upper Mekong's 80 Mt yr^{-1}). Because the cascade of reservoirs has such a large storage volume, the trap efficiency of the

Lancang cascade will remain at 83% for many decades and thus we treated it as a constant. As upstream reservoirs fill and then trap less sediment, the downstream reservoirs of the Lancang cascade will simply capture that sediment.

3W Model of Basin-Wide Reservoir Trapping

Under the 38 dam ''Definite-Future'' scenario, the cumulative sediment trapping by reservoirs will be 51%, implying that sediment load reaching the Delta will be 49% of its pre-1990 level, after sediment stored in-channel is exhausted (Figure 2). This result indicates surprisingly modest impacts given that this scenario includes the Lancang cascade and some dams on high-sediment-yield tributaries, such as the ''3-S'' basins. Eight of the ''Definite-Future'' dams are small reservoirs and with limited trap efficiencies (less than 25%) and many are high in the catchments, offering the tributary rivers some opportunity to partially recover their sediment loads downstream.

However, with full build-out of dams in the Lower Mekong River basin, including mainstem dams, about 96% the sediment load will be trapped (as of year 2020, the year by which we assume all dams are to be completed) (Figure 3). This is not to say that the sediment load reaching the Delta will immediately drop to only 4% of its pre-1990 load, because the model does not account for the potential of sediment-starved flows downstream of dams to erode sediment from the bed and banks to compensate for lack of sediment supply. The 3W model simply assumes that decreased supply from trapping sediment in reservoirs results in a comparable decrease in downstream sediment loads. Given that most of the Mekong River is bedrock controlled with very limited sediment storage [Carling, 2009a; Gupta, 2009], this assumption could be expected to hold for bedrock-controlled reaches, at least once sediment deposits are stripped out. A number of studies have examined sediment loads after closure of Manwan Dam with various results, in part because of a data gap in the records at Chiang Saen from the mid-1970s to early 1990s [Kummu and Varis, 2007; Fu et al., 2008; Walling, 2011; Liu et al., 2013]. Walling's [2011] analysis of suspended sediment data for the Lancang River at Jinghong from 1963 to 2003 provided clear evidence for increased sediment loads from the 1970s to early 1990s attributable to human disturbance (nicely shown on a double-mass curve), followed by a reduction in sediment loads (since 1993, post-Manwan Dam).

Because of the importance of the mainstem dams to sediment trapping, as well as their profound impacts as barriers to fish migration and conversion of formerly lotic habitats to lentic water bodies, we also modeled a scenario for Full Buildout in tributaries (by year 2020) but without building the mainstem dams. Under this scenario, the cumulative sediment trapped would be 68%, so that once in-channel sediment deposits had been stripped out (and not accounting for other factors such as sand mining), about 32% of the historical sediment load would reach the Delta (Figure 4).

Our sensitivity analysis showed relatively little effect on results through using the upper or lower Brune curves. For the Full Buildout scenario, cumulative sediment trapping below Sambor Dam (the lowest in the system) was 93% using the lower curve and 98% using the upper curve, compared to the 96% calculated using the middle curve. For the Definite Future scenario, cumulative trapping was 50% using the lower curve and rounds to 51% using either the upper curve, or the middle curve.

Geomorphic Impacts of sediment trapping on the channel and delta

Although the influence of reservoir-induced sediment starvation on downstream channel change will clearly be complex and varied, fundamental principles such as Lane's Balance (Lane 1955) and the presence or absence of geological controls can be used as preliminary predictive tools. Cumulative sediment trapping by dams will be substantial while reductions in high flows will be minimal (Mekong River Commission 2010, Piman et al. 2013). Therefore, the Mekong River will continue to have the capacity to transport sediment in large quantities, but the supply of sediment for transport will be reduced with future hydropower development. Thus, sediment trapping by reservoirs is arguably the most important consequence of dams for the downstream channel. Channel adjustment will be limited primarily by geological controls.

Delineation of reaches by geomorphic characteristics

The Upper Bedrock reach extends from the Chinese border downstream to about 5 km upstream of Vientiane, Laos. In this reach, the Mekong River channel is bedrock controlled, with limited, and presumably transient, sediment storage (Figure 5). The channel gradient averages 0.0003, and channel width ranges from 200 to 2000 m. This reach includes many wide, bedrock-floored reaches where bedrock is discontinuously overlain by a thin (ca 1 – 2 m) veneer of sand (Figure 6). The Middle Alluvial Reach extends downstream from Vientiane to Savannakhet, Laos. It is alluvial, with both single-channel and island-bar sections. Channel gradient averages 0.0001, and the channel is 800 – 1300 m wide. From Savannakhet downstream to Kratie, Cambodia, the Middle Bedrock Reach is again bedrock controlled. This reach includes a wide range of channel forms, as reflected in Gupta and Liew (2007) having broken this section of river into four reaches (their reaches 3, 4, 5, and part of 6). For our analysis, the key attribute of all these reaches is bedrock control (and thus we consider it a single reach), though a variety of sedimentary forms are present including sections with alluvial banks and anastomosed channels with rock-core islands covered with a relatively thin veneer of sand and silt (Meshkova and Carling 2012; Van et al. 2012). For example, from Sambor to Kratie, the bedrock control is largely buried, so the river here displays many alluvial features. However, the underlying bedrock limits its potential response to sediment starvation. Channel gradient in the upper portion of Reach 3 is approximately 0.00006 and decreases downstream. Channel width ranges from 750 to 5000 m.

The Cambodian Alluvial Reach extends from Kratie downstream to Phnom Penh. Here, the Mekong is again alluvial, crossing the wide floodplain of Cambodia to enter the depositional reaches of the delta. Channel gradient is 0.000005 and widths range from 3000 to 4000 m. Some large-scale structural control is provided by bedrock, but channel planform and position are primarily set by channel migration through alluvium. Downstream of Phnom Penh is the Mekong delta, by definition a reach of net deposition. The delta occupies an area of ~94,000 km^2 including its subaqueous extent, about 49,100 km^2 exposed subaerially (Ericson et al. 2006), making it the third largest delta in the world (Coleman and Wright 1975). The delta begins ~330 km from the sea where the Bassac, the first deltaic distributary, separates from the mainstem. Ultimately, there are four main channels that reach the sea. As a result of groundwater extraction and limited sediment starvation, the Mekong is categorized as a 'Delta in Peril' with late twentieth-century aggradation at less than 0.5 mm yr^{-1} and relative sea-level rise occurring at ~ 6 mm yr^{-1} (Syvitski et al. 2009). Sediment trapping under future dam building scenarios will further limit sediment delivery and distribution in the delta.

Potential effects on channel reaches

Definite-Future scenario

Under the Definite-Future scenario, the Upper Bedrock Reach will have an 83% reduction in sediment at the upstream end of the reach, though as less regulated tributaries enter the reach, the cumulative trapping decreases to 64% at the downstream end of Reach 1. The relative reduction in sediment supply in this reach is the greatest of any reach in the Definite- Future scenario, yet because the reach is bedrock controlled, we anticipate only modest channel adjustment. Loose sediment deposits over bedrock as described by Carling (2009a) (including slack-water deposits on bars, islands, inset floodplains and banks) (Figure 6) will likely be swept away in the first competent floods post-dam. Changes in bed level will likely be confined to accelerated scouring of pools (Carling 2009b).

In the Middle Alluvial Reach, the sediment reduction decreases to 52% at the downstream end of the reach while sediment reductions in the Middle Bedrock and Cambodian floodplain Reaches fluctuate between 51% and 56%. Under current conditions, bank erosion is not excessive (Darby et al. 2010), but we anticipate the most substantial post-dam erosion and channel adjustment in the Middle Alluvial reach and Cambodian Alluvial reaches, where bank erosion is currently occurring and where coarse bed sediment is exposed, suggesting Holocene incision (Carling 2009a). Large island features are believed to result from chute cut-offs and suggest a dynamic river system (Carling 2009a). Because upstream reservoirs will have a limited influence on flow regime, but will trap more than half of the total sediment load, we expect channel widening in alluvial reaches as the river seeks to recover its sediment load by eroding the channel margin, as commonly observed in sediment-starved rivers (Kondolf 1997). Incision is also likely except where the bed elevation is controlled by bedrock. The Middle Bedrock reach has single-thread, bedrock-confined reaches, anastomosed reaches of bedrock islands, and also includes the base-level control of Khone´ Falls (Carling 2009a). Future high flows of sediment-starved water may erode alluvium on bars, banks, and islands without replacement. Erosion into bedrock is not expected on a timescale of decades. The Cambodian Alluvial reach is a floodplain river with active meandering in anastomosed sections. Individual islands are transient features, though the island complexes are relatively stable (Carling 2009a). Without replenishment, new islands will be less likely to develop and loss of the island features will likely occur. Erosion of the main channel bed and banks is also expected. The Mekong delta will receive about half of its natural sediment load, and can be expected to experience accelerated subsidence and coastal erosion. Further research is needed on the size distribution of sediment transported by the river, and the size fractions most affected by the dams, but we expect the dams to disproportionately affect bed material load, notably sand, which is most important for building beaches and nourishing the coast, as discussed below.

Full-build scenario

With all proposed dams constructed (full-build scenario) and cumulative sediment reduction ranging from 83% below the Chinese boarder to 96% in Vietnam, we expect the most dramatic response in the alluvial reaches from Vientiane to Savannakhet and from Kratie downstream, where channel bed and banks will be susceptible to erosion. While the bedrock-controlled reaches above Vientiane and from Savannakhet to Kratie will not downcut (except to remove any layers of erodible alluvium overlying bedrock, and/or to deepen pools) and will not have dramatic occurrences of erosion or channel instability, the extensive existing sediment deposits (bars, islands, inset floodplains and banks) will be stripped away, and bed material size will coarsen, all with potentially important ecological consequences. If the thin veneer of sediment in the bedrock reaches is removed, it can substantially alter the substrate, baseflow channel roughness, and water velocity that influence fundamental elements of habitat availability for the benthic macroinvertebrates, fishes, and other aquatic biota. Since little is known about many Mekong species, it is difficult to predict their response to channel change and consequent loss of habitat.

If the mainstem dams are constructed, they will inundate long reaches of the river and their backwater effects will extend further upstream. Within these reservoirs and backwater areas, rather than experiencing erosion from energetic flows, the channel will become a depositional zone. An important ecological feature of the river are the deep pools that provide essential habitat for native fishes and river dolphins (Poulsen and Valbo-Jorgensen 2001, Baird and Flaherty 2005), and which are maintained by scour created by local hydraulics. Sediment starvation below dams is unlikely to negatively affect these pools through increased erosion. However, within the extensive zones of reservoir inundation and backwater, local hydraulics will change, likely eliminating the scouring currents that have maintained these features, and they will begin to fill with sediment and debris.

Potential effects on Mekong River delta from analogous Cases

At present, approximately 21,000 km^2 of land in the Mekong delta is less than 2 m above sea level and 37,000 km^2 is regularly flooded (Syvitski et al. 2009). While sediment delivery to the Mekong delta remained relatively constant over most of the twentieth century, recent decades have seen accelerated rates of sea-level rise, more rapid compaction due to groundwater extraction, loss of sediment to offshore waters by channelization, and in the late twentieth century, reduced sediment delivery resulting from in-channel mining of sand and gravel. Thus, even under the pre-dam sediment regime, the delta was submerging and flood-prone areas expanding. Anticipated dam-induced reductions in sediment supply can only exacerbate the rate of land loss.

The pre-dam sediment accumulation rate across the Mekong delta was ~0.5 mm y^{-1}, while overall relative sea-level rise was ~6 mm y^{-1} (Syvitski et al. 2009). Mekong delta data are limited and predictions are complicated by uncertainty in subsidence rates, sediment delivery, and eustatic sea-level rise. Using the Mississippi River as an analogue, the US Geological Survey assumed subsidence rates up to 10 mm y^{-1} and future increases in eustatic sea-level rise rates of 2 – 6 mm y^{-1} (based on IPCC AR4 scenarios of best case (B1) and worst case (A1F1) emissions (Doyle et al. 2010)), more than the 2 mm y^{-1} increase of recent years.

Similar to full-build predictions for the Mekong River, the Colorado, Ebro, Indus, Krishna, Nile, and Yellow River deltas have all experienced sediment reductions of 90% or more (Table 1). Since those deltas have comparable or lower rates of relative sea-level rise than the Mekong and similar intensities of wave energy, they provide a reasonable framework for understanding the likely impacts of unmitigated dam construction. The Indus, Nile, and Yellow River deltas were all prograding prior to dam construction and subsequently were net erosional. For example, the Indus coast was prograding ~100 m y^{-1} before dam construction and retreating ~50 m y^{-1} in recent decades. Rates of pre-dam delta growth were not reported in the literature for the Colorado, Ebro, and Krishna, but all are actively eroding in the post-dam period (see Table 1 for citations). Rates range from 1 to 90 km^2 y^{-1} of area lost per year and from 10 to 70 m y^{-1} of coastline retreat. Detailed modelling of the Mekong delta is required to make quantitative predictions of erosion, but experience from around the world suggests a high likelihood of widespread erosion unless sediment management practices are implemented for proposed Mekong dams. Although there is considerable variability, our compiled global data set (Table 1) shows a strong relationship (r2 = 0.53) between reduction in sediment supply to deltas and the subsequent reduction in aggradation rates (Figure 7), as might be expected from geomorphic principles. The six deltas with sediment reductions of 80% or more (Indus, Chao Phraya, Krishna, Ebro, Nile, and Colorado), all show reductions in aggradation rates of more than 88%.

Discussion

Original Methodological Contribution

We build on previous work, using data not previously available, accounting for differences among geomorphic provinces, and accounting for time and space effects in sediment trapping, to develop the best possible estimate of cumulative sediment trapping. Our compilation of total storage values for the Mekong reservoirs allowed our analysis to avoid systematic underestimates of trap efficiencies that could result from using the more widely available active storage values as input to the Brune curve. For example, for the proposed Xayaburi Dam in Laos, the active storage listed in the MRC database is 0.225 km^3, but the total storage is 1.3 km^3 [MRC, 2011]. Using these different values in the Brune curve yield very different trap efficiencies: negligible versus 51%, respectively. Thus, although Xayaburi has been called a ‘‘run of the river’’ dam because it will not significantly alter the flow regime, the Brune curve suggests it has the potential to trap half the river’s sediment. The 3W model allows the calculation of each reservoir individually, rather than lumped calculations by basin as done in an earlier study. Consider a simple case: a basin with three principal tributaries. If three reservoirs are built, one each on the three tributaries, then the reservoir storage could be combined and used to estimate trap efficiency with the Brune curve, probably without introducing great errors. However, if the reservoirs are built all on one tributary in series (a cascade of reservoirs), the theoretical trap efficiency of the lower dam is meaningless because there may not be any sediment left to trap in that reach, although it may be abundant in other tributaries.

Model Uncertainty

The uncertainty in sediment yields (discussed above in Methods) is likely the main source of uncertainty. In addition, the 3W model is clearly a simplification of real river processes, as it ignores conveyance losses downstream, scaling effects of reduced sediment yield with increasing drainage area, and potential ‘‘buffering’’ effects of sediment stored in and adjacent to the channel, the erosion of which can partially compensate for sediment sequestering behind dams. The general problem is illustrated by the fact that worldwide, the amount of sediment impounded behind dams is estimated to be nearly an order of magnitude greater than the amount by which downstream sediment loads have been reduced [Walling, 2012].

Effect of Dams on Sediment Supply

Unlike many river basins with high sediment loads, the LMRB does not contain large areas of weak, easily-eroded rocks. Most rock types are relatively hard, so the range of sediment yields (and the high yields from some regions) reflect very active tectonic settings and differences in geologically-recent tectonic shearing and uplift history. Besides the rapidly eroding catchment of the Lancang, which includes the Tibetan Plateau and deeply incised valleys downstream, the highest sediment producing regions in the basin are the Northern Highlands, Kon Tum Massif, and Tertiary Volcanic Plateau, which we estimated to produce 200–290 t y^{-1} km^{-2}.

The already built and certain future dams in the MRC's ''Definite-Future'' scenario are distributed in such a way that their impact of sediment loads is relatively modest, leaving nearly half of the natural sediment load in the river when it passes into the downstream alluvial reach and Delta. This result is somewhat surprising, but encouraging, as it implies that some rethinking of dam-building plans, along with incorporation of sediment management strategies such as sediment pass-through or sediment bypass, could mitigate the magnitude of the sediment starvation from dams in the LMRB.

The full build-out scenario without reservoir sediment management measures would trap 96% of the river's sediment load, and eventually result in nearly complete sediment starvation, with only 4% of the natural load reaching the Delta. This predicted sediment starvation is significantly greater than previous estimates [Kummu et al., 2010], and implies that the downstream impacts of full buildout of all dams would be greater than previously recognized. Our results indicate that if all planned dams except the mainstem dams were built, the cumulative sediment trapping would be 68%, i.e., allowing 32% of the sediment load to pass downstream to the Delta. While still a large impact, a 68% reduction in sediment load is not as severe as a 96% reduction in sediment load, suggesting that some combinations of tributary-only dams might be worth evaluating, especially because the mainstem dams have other profound ecosystem effects, especially on migratory fish [Baran and Myschowoda, 2009].

Our analysis estimates sediment starvation only from hydroelectric dams, and does not address sediment starvation resulting from mining of sand and gravel from the river channel, which was estimated by Bravard et al. [2013] at about 27 Mt y^{-1} upstream of Vietnam, of which 20.7 Mt y^{-1} was mined in Cambodia. In addition, we do not take into account sediment trapping by irrigation dams on tributaries, which are most concentrated on the Khorat Plateau of Thailand [Hoanh et al., 2009]; this region has low natural sediment yields, so it is unclear how significant an impact these impoundments would have on sediment load of the river. As discussed above, our analysis does not account for buffering of sediment starvation by erosion of sediment deposits, which is likely to be limited and short-term over most of the river's course over bedrock, but would be greater in the lower alluvial reach in Cambodia. Nor does our analysis address factors such as the construction of extensive dykes along the river in Cambodia, which will prevent frequent inundation of large areas of the floodplain and thereby may result in less floodplain sedimentation in the alluvial reach in Cambodia. Other land-use changes and climate change can also affect sediment load. Clearly, assessment of sediment starvation effects of planned dams must consider other factors and trends, whose effects on the sediment budget can be significant.

Our calculations of reduced sediment supply to the Mekong from reservoir trapping assume no sediment bypass or pass-through strategies are implemented in these dams, but there are many proven techniques to pass sediment through or around reservoirs [Morris and Fan, 1998; Kondolf et al., 2014b], and implementing these measures on Mekong dams could significantly reduce the sediment trapping and resulting sediment starvation. Planning and modeling efforts are now underway on several proposed dams in Cambodia and Laos to assess potential benefits of implementing sediment passage to prolong reservoir life and reduce downstream sediment starvation impacts.

Effects of Sediment Starvation on Downstream Channels

The reduction in sediment supply predicted by our analysis would likely have profound implications for the productivity of agriculture and the fishery within the lower Mekong River and Delta, as well as the offshore fishery and the sustainability of the Delta landform itself. While the objective of this study was primarily to estimate the likely magnitude of sediment reduction from planned dams, some mention of the likely effects of this reservoir-induced sediment supply reductions may be in order. These impacts should depend largely on the magnitude of the reduction and the nature of the river channel affected. Except for the 300 km alluvial reach from Vientiane to Savannakhet, the river upstream of Kratie is bedrock controlled [Gupta and Liew, 2007]. These bedrock controlled reaches display considerable diversity in form [Meshkova and Carling, 2012] and contain a variety of alluvial forms within the larger bedrock channel context (Figure 6). However, the main response to reduced sediment load in bedrock reaches will be to strip out alluvial deposits, with- out affecting the overall structure of the channel. In alluvial reaches, however, the potential for channel change is much greater: incision from sediment starvation is likely to occur, and with it, some bank collapse and retreat [Carling, 2010]. Reduced suspended load in overbank flows over the floodplains of Cambodia will reduce the natural renewal of soil fertility, and reduced suspended sediment and nutrients flowing into Tonle Sap threaten the productivity of this extraordinary system [Sarkkula et al., 2003]. At the downstream end of the fluvial system, the delta is vulnerable to reduced sediment supply, especially in light of its relatively rapid, recent formation [Sarkkula et al., 2010].

Effects of Reservoir Sedimentation on Storage Capacity

Sediment trapping in reservoirs affects not only downstream reaches through sediment starvation, but also reduces storage capacity of reservoirs and can interfere with functioning of the dam and hydroelectric power plant. With full build of all 133 dams proposed in the LMRB, our model results indicate that by 2100 (after about 80 years), seven will be more than 70% full and 13 will be more than 50% full. By 2420 (after about 400 years), 23 will be more than 70% full and 39 will be more than 50% full. In some cases, the relatively slow rates of filling result from upstream dams trapping sediment that would otherwise have deposited in the reservoir.

Conclusions

We applied the 3W model of Minear and Kondolf [2009] to predict sediment starvation from existing and proposed dams in the Mekong basin. The unprecedented rate of dam construction in the Mekong River Basin is likely to result in greater sediment starvation than recognized by previous studies. By developing systematic estimates of sediment yield by geomorphic province within the basin (constrained by historical measured transport rates), using total storage instead of active storage in calculating trap efficiency, calculating trapping by individual reservoirs instead of lumping by tributary basin, and accounting for trapping effects of upstream reservoirs and changes in trap efficiency over time, we developed refined estimates of sediment trapping. Our results indicate that full build-out of proposed dams would trap the equivalent of 96% of the river's historical sediment supply to the lower alluvial Mekong River and Delta. Dams already built and deemed virtually certain in the near future would reduce the sediment supply only to 49% of its pre-dam level. While our model is transparently simple (ignoring effects such as buffering of sediment starvation by bank and bar erosion), our results indicate significant sediment starvation is likely downstream, though after a lag time during which the river would ''cannabilize'' its limited supply of sediment stored in channel deposits and accessible bank deposits, probably much less than two decades for bedrock-dominated reaches. Precise predictions of the consequent impacts on specific reaches of the river and delta was somewhat beyond the scope of this broad study, but we need only look at analogous cases [Syvitski et al., 2009] to recognize the potential severity of impacts on the river channel and delta landforms, floodplain fertility, and productivity of the ecosystem, including the extraordinary Mekong River fishery, which provides essential protein to 60 million people [Hortle, 2009]. In light of the magnitude of the potential sediment starvation on the river system, riparian countries, international agencies, and donor countries should arguably prioritize efforts to require new dams to be designed to pass sediment (and retrofit existing ones where possible), with the added benefit of more sustainable hydropower production into the future if reservoir sedimentation can be reduced.

Deltas evolve through a complex interplay of river, tides, waves, and biological and human factors. For example, mangrove communities slow wave velocities, thereby efficiently trapping sediment, and improving water quality and preventing coastal erosion. The extent of mangrove ecosystems in the Mekong delta has remained stable in recent decades (Shearman et al. 2013). Elsewhere, mangroves are at risk from rapid sediment deposition (Ellison 1999) (not likely in the Mekong), as well as sea-level rise and increased storm intensity, aquaculture and water quality impacts, and sediment reductions from dams (Thampanya et al. 2006). Such interactions exemplify the challenges posed to modelers of deltaic systems. These processes are important to understand, yet challenging to accurately parameterize, given the unpredictable nature of storm events and other stochastic processes. While detailed models of delta morphodynamics would no doubt be helpful in predicting the response to reduced sediment supply, for management decisions that need to be taken soon, we see no need to wait for such detailed studies. Rather, our compilation of experience from other deltas can provide a reasonable basis upon which to anticipate the consequences of sediment reduction. Given that the deltas with sediment reductions of 80% or more all had reductions in aggradation rates of more than 88%, under a full- build scenario in the Mekong basin, we expect an almost complete cessation of sediment deposition. As such, land subsidence and eustatic sea-level rise will be essentially uncontrolled, and few options will be available to mitigate coastal retreat. An expert team assembled by the Natural Heritage Institute is currently working with the governments of Laos and Cambodia to explore alternative placement and design of some key dams such that sediment can be passed through or around the dams, in hopes of partially mitigating the sediment trapping effects of these structures (Wild and Loucks 2014).

Given the magnitude of sediment starvation likely to occur in the near future, our review of experiences elsewhere, combined with fundamental geomorphic principles, can provide an initial prediction of likely effects. In data-limited, poorly understood systems such as the Mekong, implementation of detailed models may be unrealistic due to the lack of long-term and/or reliable data for calibration. By relying on the global data set, we hope to provide decision-makers and stakeholders in the Mekong River basin with some useful information on likely trends, until more accurate modeling and forecasting tools become available.

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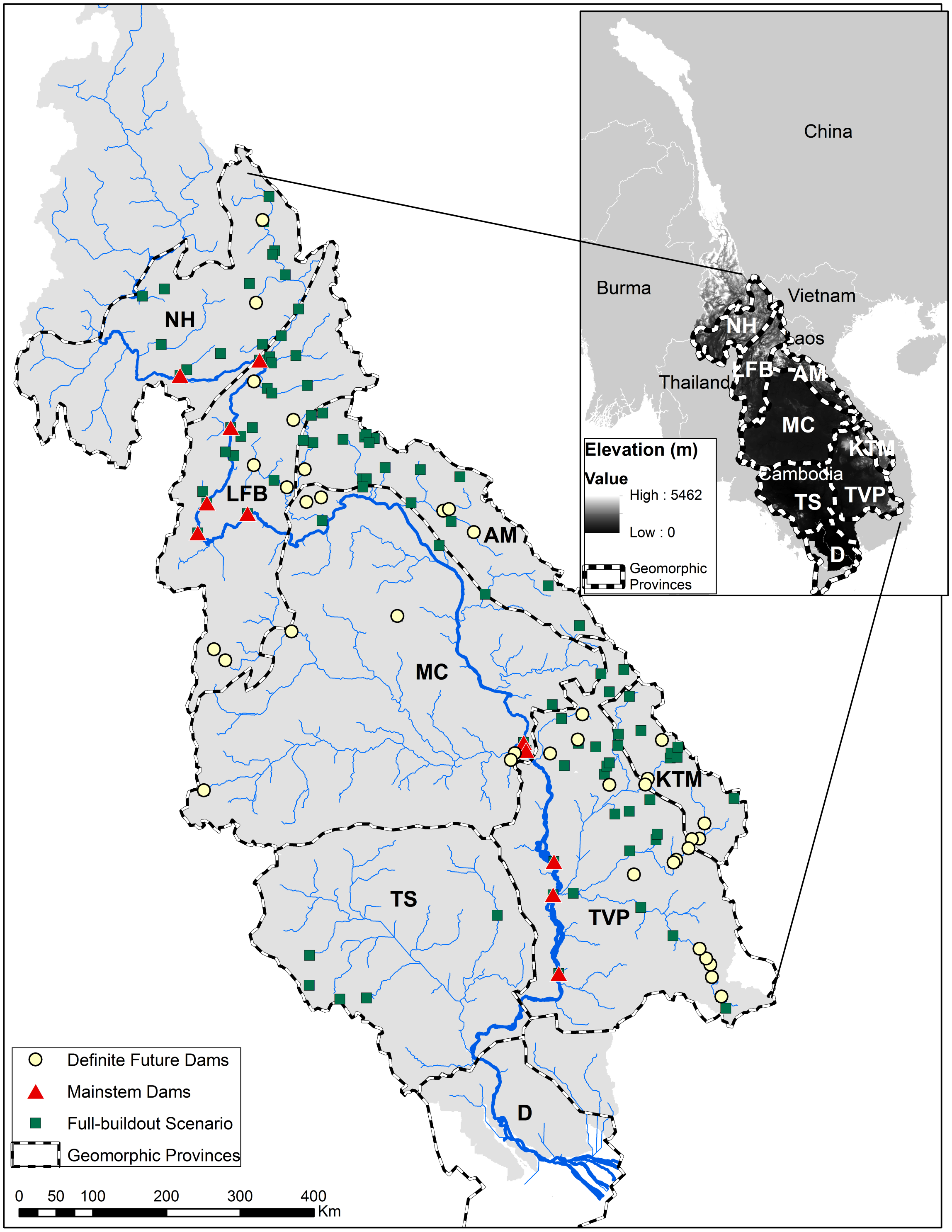

Figure 1. Lower Mekong River Basin. Based on rock type, uplift, land-use, and available sediment transport data, we delineated nine geomorphic provinces: NH (Northern Highlands,) LFB (Loei Fold Belt,) MC (Mun-Chi Basin,) AM (Annamite Mountains,) KTM (Kon Tum Massif,) TVP (Tertiary Volcanic Plateau,) TS (Tonle Sap,) D (Delta). Dam locations are indicated for three scenarios: definite future, full-buildout, and full buildout without the mainstem dams. Inset: Entire Mekong River basin, with generalized elevations indicated in black-gray-white shading. Figure from Kondolf et al., 2014.

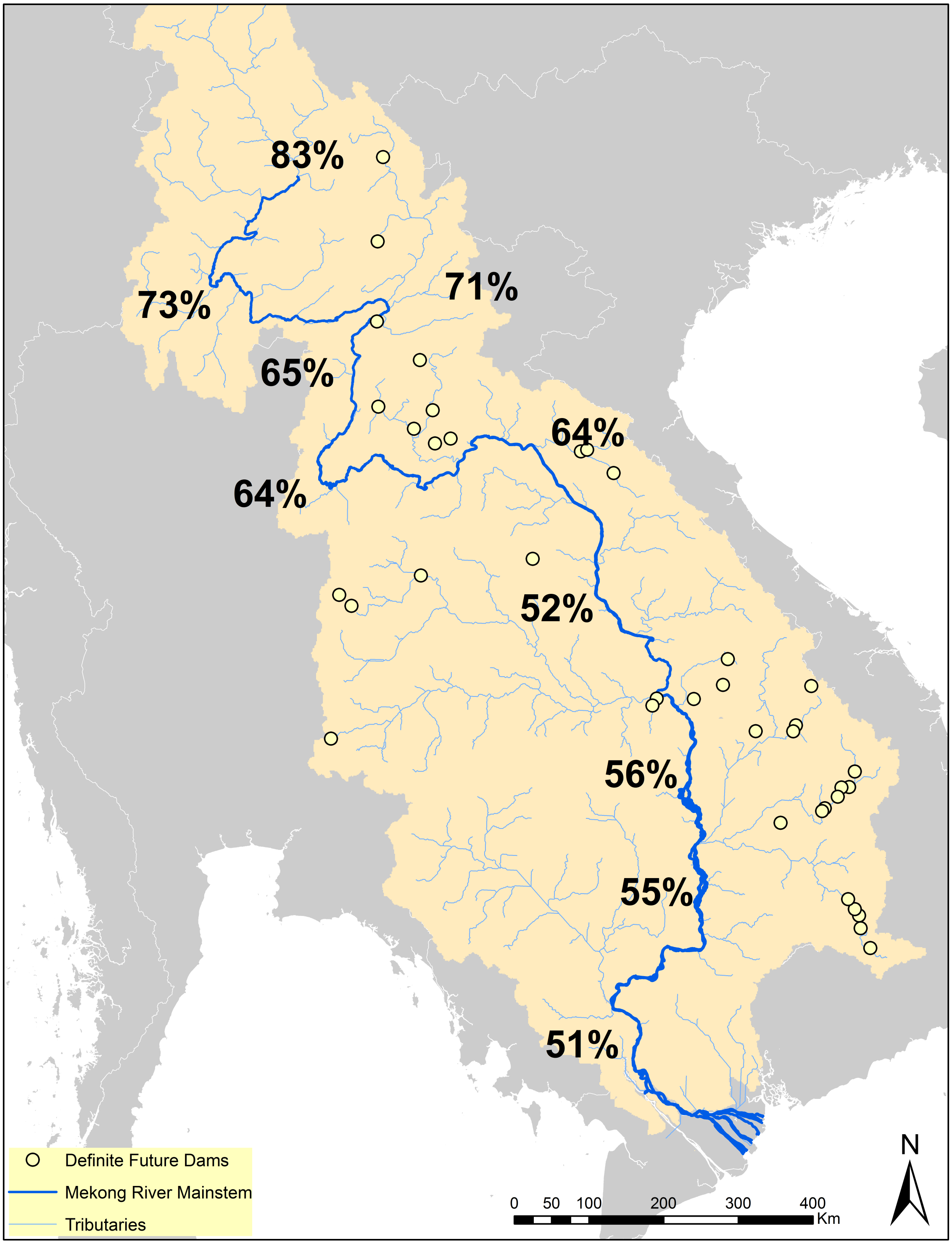

Figure 2. Cumulative Sediment Starvation in year 2020 under the 'definite future' scenario. For the whole basin, a surprisingly modest 51% of the basin's 160 Mt y^{-1} would be trapped under definite future scenario despite 83% trapping in the Lancang and 38 LMRB dams. Figure from Kondolf et al., 2014.

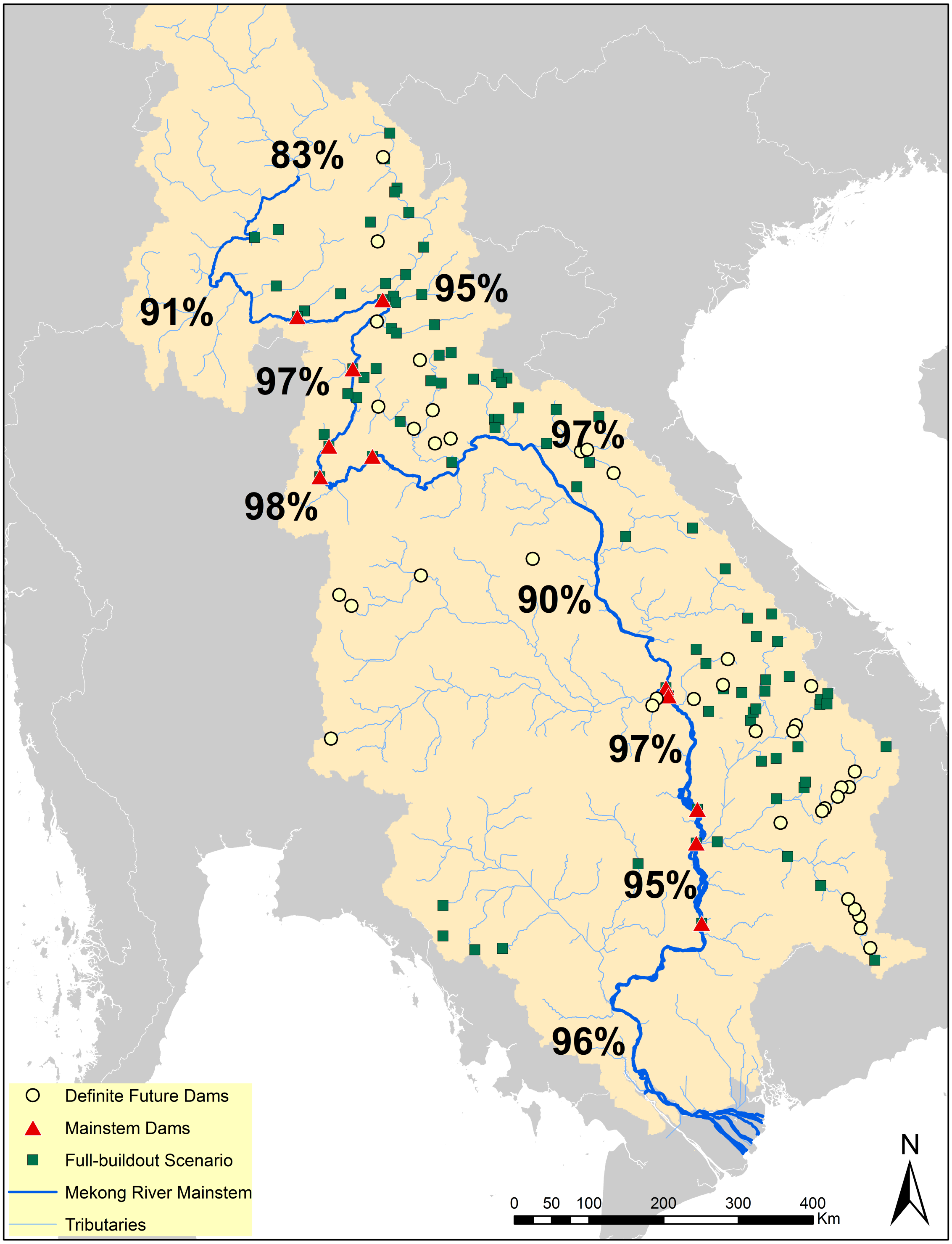

Figure 3 Cumulative Sediment Starvation in year 2020 under the 'full buildout' scenario. Less than 5% of the basin's unregulated 160 Mt y^{-1} would arrive to the delta if all 133 LMRB dams are constructed as currently proposed, and without implementing sediment management measures. Figure from Kondolf et al., 2014.

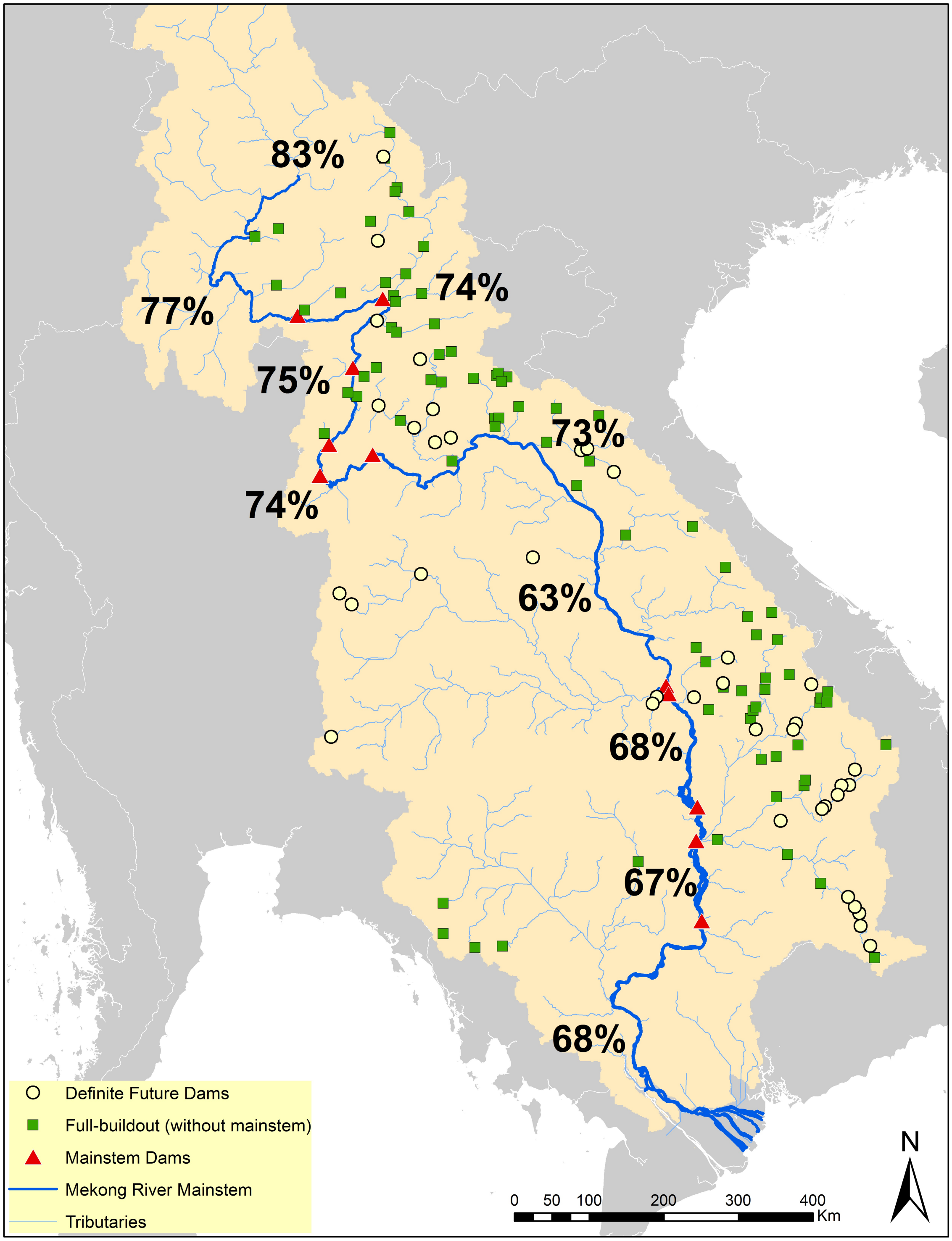

Figure 4. Cumulative sediment starvation for full buildout without the mainstem dams. 68% of the total sediment load of the river would be trapped before reaching the delta.

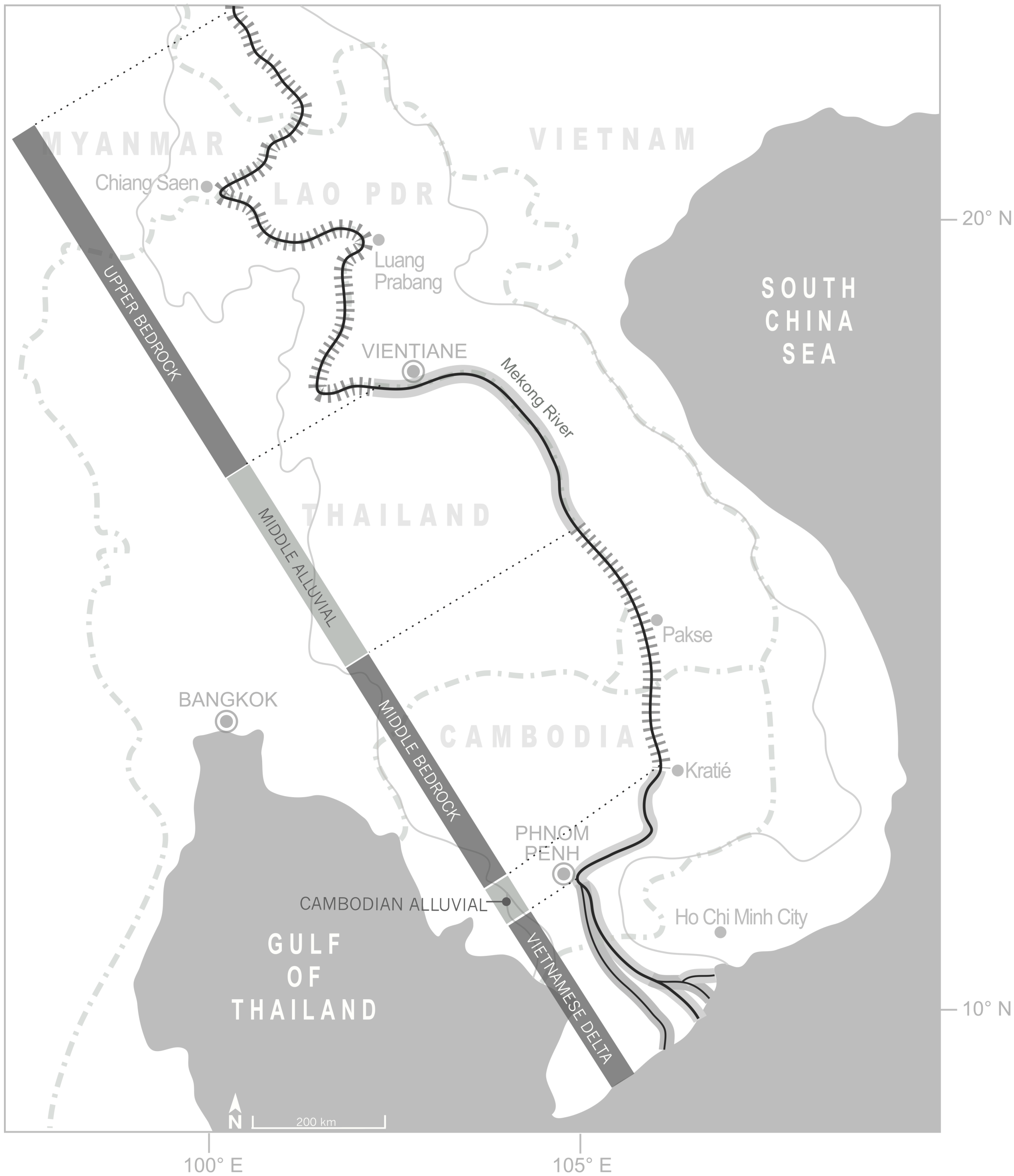

Figure 5. Reaches of predominantly bedrock vs. alluvial channel, as generalized from Adamson (2001), Gupta (2004), and Carling (2009a). Bedrock-controlled channel reaches are likely to experience rapid loss of surficial sediment deposits, but will not manifest large channel changes in response to reduced sediment loads, whereas alluvial reaches will likely incise and/or widen as they erode to compensate for reduced sediment supply. Figure from Rubin et al., 2014.

Figure 6. Thin veneer of sand over bedrock channel, mainstem Mekong River near Xayaburi, Laos. (photo by Kondolf, from Kondolf et al., 2014).

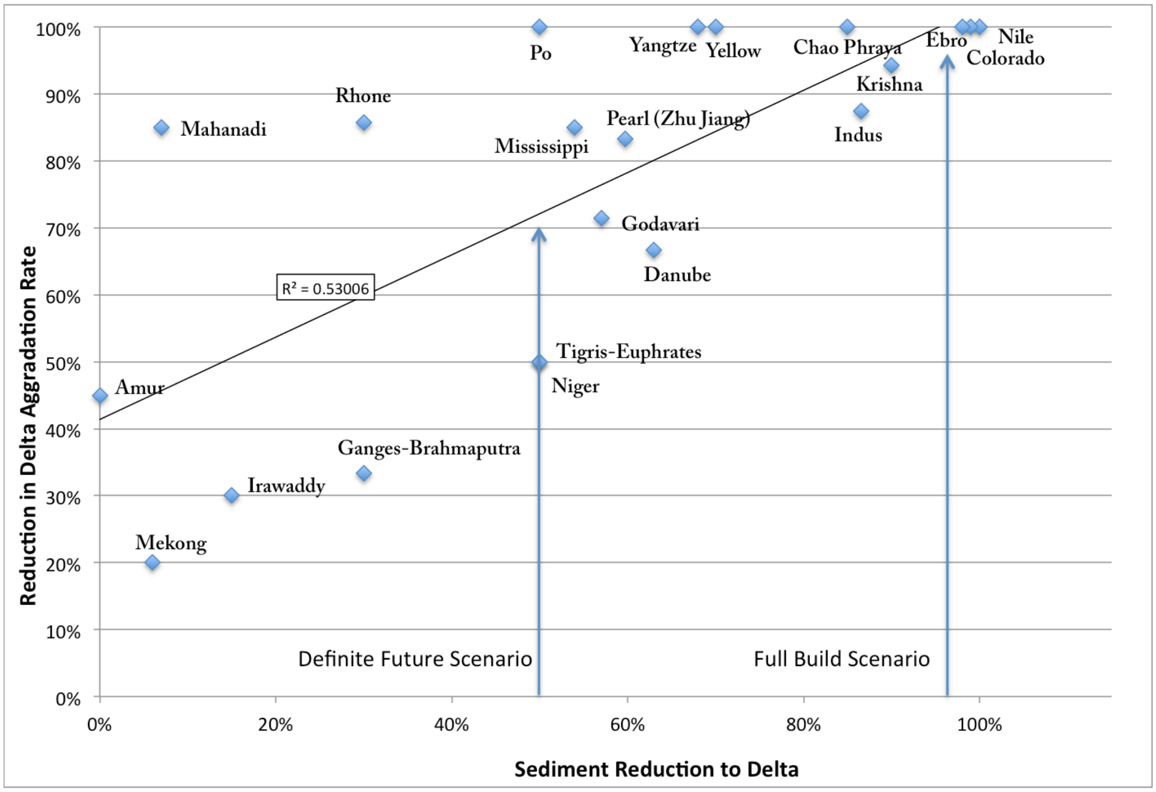

Figure 7 Data set of 24 worldwide deltas shows positive relationship of sediment reductions to deltas resulting in decreased rates of aggradation. In particular, sediment reductions of more than 80% are consistent in almost complete cessation in aggradation rates. The full-build scenario of Mekong dam building would result in 96% reduction in sediment delivery and we would then expect an almost complete cessation in sediment deposition in the delta. Figure from Rubin et al., 2014.

Table 1. Deltas of the World

River	Drain-age Area [qq] (1000's km^2)	Relief (km) [qq]	Total Delta Area [b] (1000's km^2)	Delta Area <2m above sea level [a] (1000's km^2)	Flow Reg. [k] (%)	Early 20th Century Aggradat-ion Rate [a] (Mm y^{-1})	Recent Aggra-dation Rate [a] (Mm y^{-1})	Relative Sea Level Rise [a] (Mm y^{-1})	Reduction in Sediment Delivery (%)	Wave Energy [r] w_a : max monthly wave height (m)	Notes
Amur, Russia	1,755	2.51		1.2	9%	2	1.1	1	0% [a]		
Chao Phraya, Thailand	142	1.9	11	1.8	76%	0.2	0	13-150	~ 85%[a,l]	1.5	~50% reduction in average annual maximum flow after flow regulation [s]
Colorado, Mexico	638	3.7	0.6 [c]	0.7	280%	34	0	2-5	100% [a]	0.5	Total annual flow reduced ~90% [t]. Since 1930, ~90 km^2 y^{-1} of delta area lost [u].
Danube River, Romania	779	4.1	4 [f]	3.7 [jj]	5%	3	1	1.2	63% [a]	1.5	Prograding during last 2,800 years. Mean rate of coastal retreat from 3 to 5 m y^{-1} during recent decades [kk]. Eroded land includes ~ 6 km^2 y^{-1} of agricultural and industrial land and 83 km^2 y^{-1} wetlands from 1987-2001 [nn].
Ebro River, Spain	85	3.34	0.3 [d]	0.1 [i, j]	23%	5-7 [i]	0 [i]	3-4 [i]	99% [d]	1.5	Flow regulation reduced max monthly discharge and annual average discharge by ~70% [r]. 10-60 m y^{-1} coastline retreat [v] with 45% of the emerged delta expected to be submerged by 2100 [d]

River	Drain-age Area [qq] (1000's km^2)	Relief (km) [qq]	Total Delta Area [b] (1000's km^2)	Delta Area <2m above sea level [a] (1000's km^2)	Flow Reg. [k] (%)	Early 20th Century Aggradat-ion Rate [a] ($Mm\ y^{-1}$)	Recent Aggra-dation Rate [a] ($Mm\ y^{-1}$)	Relative Sea Level Rise [a] ($Mm\ y^{-1}$)	Reduction in Sediment Delivery (%)	Wave Energy [r] w_a : max monthly wave height (m)	Notes
Ganges-Brahmaputra, Bangladesh	1,628	6.09	106	6.1	8%	3	2	8-18	30% [a,l]	1.0	~20 m y^{-1} of coastal retreat in recent decades along the coast of Bangladesh [ll]. Eroded land includes ~ 292 km^2 y^{-1} of agricultural and industrial land and 358 km^2 y^{-1} wetlands from 1989-2001 [nn].
Godavari River, India	313	1.06	5 [e]	0.2	37%	7	2	3	40% [a] 74% [l]	2.0	0.74 km^2 y^{-1} loss of delta land area since the 1970's [w]. Saltwater intrusion exacerbated by surface and groundwater withdrawals [c].
Indus River, Pakistan	941	5.18	30	4.8	13%	8	1	>1.1	80% [a] 93% [l]	3.5	Flow regulation reduced max monthly discharge (~40%) and annual average discharge (~50%) [r]. Prior to dam construction, coastline prograding ~100 m y^{-1}. Recent decades average 50 m $^{y-1}$ of coastline retreat [x]. The Indus Delta has the highest wave energy of any major delta [y]. Eroded land includes ~ 79 km^2 y^{-1} of agricultural and industrial land and 199 km^2 y^{-1} wetlands from 1992-2000 [nn].
Irawaddy, Burma	406	4.8	21	1.1	1%	2	1.4	3.4-6	30% [a] 0% [l]	1.5	From 1925 to 1989 the Irawaddy delta grew 8.7 km^2 y^{-1}, then eroded at a rate of 13 km^2 y^{-1} from 1989-2006, probably a result of sediment trapping in small-medium sized tributary dams[mm].

River	Drainage Area [qq] (1000's km^2)	Relief (km) [qq]	Total Delta Area [b] (1000's km^2)	Delta Area <2m above sea level [a] (1000's km^2)	Flow Reg. [k] (%)	Early 20th Century Aggradation Rate [a] ($Mm\ y^{-1}$)	Recent Aggradation Rate [a] ($Mm\ y^{-1}$)	Relative Sea Level Rise [a] ($Mm\ y^{-1}$)	Reduction in Sediment Delivery (%)	Wave Energy [r] w_a : max monthly wave height (m)	Notes
Krishna River, India	259	1.33	5 [e]	.3	37%	7	0.4	3	~90% [a,l]	2.0	Flow regulation reduced max monthly discharge (~20%) and annual average discharge (~40%) [r]. 0.78 $km^2\ y^{-1}$ loss of delta land area since the 1970's [c].
Mahanadi, India	142	1.14	11 [f]	.2	17%	2	0.3	1.3	~ 7% [a,l]	2.0	Eroded land includes ~ 7 $km^2\ y^{-1}$ of agricultural and industrial land and 31 $km^2\ y^{-1}$ wetlands from 1989-2002[nn].
Mekong, Vietnam	759	5.47	94	21	3%	0.5	0.4	6	12% [a] 0% [l]	1.5	Negligible change in annual discharge and monthly maximum discharge [r]. Delta area stable in recent decades. 4.4 m $^{y-1}$ of coastline retreat with higher rates (12.2 m $^{y-1}$) along Cau Mau Peninsula [rr]
Mississippi River, USA	3,203	4.4	29	7.1	16%	2	0.3	5-25	48% [a] >60% [m]	0.5	Negligible change in annual discharge and annual monthly maximum discharge [r]. 4,900 km^2 lost since early 20th century [m] with land-loss rates from ~ 40 $km^2\ y^{-1}$ to 100 $km^2\ y^{-1}$ from 1940's to 2000 and wetland losses of 43 $km^2\ y^{-1}$ from 1985-2010 [z].
Niger, Nigeria	1,240	2.1	19	0.4	15%	0.6	0.3	3.2	50% [a]	1.4	Eroded land includes ~ 0.5 $km^2\ y^{-1}$ of agricultural and industrial land and 6 $km^2\ y^{-1}$ wetlands from 1987-2002 [nn].

River	Drainage Area [qq] (1000's km^2)	Relief (km) [qq]	Total Delta Area [b] (1000's km^2)	Delta Area <2m above sea level [a] (1000's km^2)	Flow Reg. [k] (%)	Early 20th Century Aggradation Rate [a] (Mm y^{-1})	Recent Aggradation Rate [a] (Mm y^{-1})	Relative Sea Level Rise [a] (Mm y^{-1})	Reduction in Sediment Delivery (%)	Wave Energy [r] w_a : max monthly wave height (m)	Notes
Nile River, Egypt	2,026	3.78	13	9.4	95%	1.3	0	4.8	98% [a]	1.5	Dams and diversions reduced max monthly discharge (~80%) and annual average discharge (~60%) [r]. The delta was prograding prior to construction of Aswan Dam in 1970. Coastal erosion rates of 38-71 m y^{-1} [aa] and crash of the abundant sardine fishery of the Mediterranean after Aswan. Eroded land includes ~ 0.7 km^2 y^{-1} of agricultural and industrial land and 0.8 $km^2 y^{-1}$ wetlands from 1984-2001 [nn].
Pearl (Zhu Jiang), China	370	3.5	8 [g]	3.7	31%	3	0.5	7.5	67% [a] 22% [l] >90% [n]	1.5	Dam construction began in the 1950s and expanded to reach over 9,000 dams. Rapid and widespread development obscure changes in the coastline, but river regulation, combined with incised channels (from reservoir sediment trapping and extraction of sand for construction) and sea level rise has led to saltwater intrusion and increased erosion potential due to the larger tidal prism [bb].
Po, Italy	72	4.8	13	0.6	4%	3	0	4-60	50% [a]	1.5	Flow regulation reduced max monthly discharge (~30%) and annual average discharge (~20%) [r].

River	Drainage Area [qq] (1000's km^2)	Relief (km) [qq]	Total Delta Area [b] (1000's km^2)	Delta Area <2m above sea level [a] (1000's km^2)	Flow Reg. [k] (%)	Early 20th Century Aggradation Rate [a] (Mm y^{-1})	Recent Aggradation Rate [a] (Mm y^{-1})	Relative Sea Level Rise [a] (Mm y^{-1})	Reduction in Sediment Delivery (%)	Wave Energy [r] w_a : max monthly wave height (m)	Notes
Red (Hong Ha), Vietnam	150	3.14	12		3%				76% [l] 50% [o]	1.5	Flow regulation reduced annual average discharge (~10%) [r]. Notorious for extreme floods, the delta of the Red River is dynamic with large and rapidly shifting zones of erosion and deposition. In recent decades, the coastline has retreated 2 km in some areas while advancing up to 5 km in others making it difficult to discern the coastal impacts of sediment trapping [cc]
Rhone, France	99	4.81	1.5 [h]	1.1	6%	7	1	2-6	30% [a]	2.0	Coastal retreat halted through hard engineering structures, though steepening of shoreline suggests chronic erosion problems in the future [oo].
Tigris-Euphrates, Iraq	1,050	2.96	18	9.7	124%	4	2	4-5	50% [a]	1.0	Close to 10,000 km^2 of marshes destroyed in last decades of 20th Century through dam construction and diversion of water away from marsh area [pp]. Though not 'eroded', this change is one of the most dramatic of any delta system.

River	Drain-age Area [qq] (1000's km^2)	Relief (km) [qq]	Total Delta Area [b] (1000's km^2)	Delta Area <2m above sea level [a] (1000's km^2)	Flow Reg. [k] (%)	Early 20th Century Aggradat-ion Rate [a] (Mm y^{-1})	Recent Aggra-dation Rate [a] (Mm y^{-1})	Relative Sea Level Rise [a] (Mm y^{-1})	Reduction in Sediment Delivery (%)	Wave Energy [r] w_a : max monthly wave height (m)	Notes
Yangtze (Chang Jiang), China	1,794	4.38	67	6.7	12%	11	0	3-28	~ 70% [a,l,p]	1.5	Negligible change in annual discharge and annual monthly maximum discharge [l,r]. Sediment delivery decreased since the 1950's and the delta became net erosional with the filling of Three Gorges Dam in recent years [dd,ee,ff,gg].
Yellow (Huanghe) River, China	865	5.9	36	1.4	51%	49	0	8-23	30% [a] 84% [l] 90% [q]	1.5	Flow regulation reduced max monthly discharge and annual average discharge by ~20% [r]. The delta was prograding at a rate of 20–25 km^2 y^{-1} in the early 20th century, but is now net erosional [hh,ii], grain size has coarsened, and sediment dispersal patterns and the slope of the coastline have changed [q]. Eroded land includes ~ 66 km^2 y^{-1} of agricultural and industrial land and 67 km^2 y^{-1} wetlands from 1989-2000 [nn].

References:

a) Syvitski et al. 2009; b) Hori and Saito 2008; c) Luecke et al. 1999; d) Rovira and Ibàñez 2007; e) Gamage and Smakhtin 2009; f) Coleman and Huh 2003; g) Encyclopedia Britannica Online 2013; h) Pont et al., 2002; i) Ibàñez, et al. 1996; j) Less than 0.5 m above mean sea level; k) Nilsson et al. 2005- flow regulation calculated as sum of live reservoir storage relative to virgin mean annual flow; l)Gupta et al. 2012; m) Alexander et al. 2012; n) Dai et al. 2008; o) Dang et al. 2010; p) Yang et al. 2011; q) Wang et al. 2010; r) Syvitski and Saito 2007; s) Kundzewicz, et al. 2009; t) Schmidt 2008; u) Luecke et al. 1999; v) Jimenez and Sanchez-arcilla 1997; w) Malini and Rao 2004; x) Giosan el al. 2006; y) Wells and Coleman 1984; z) Couvillion el al. 2011; aa) Frihy el al. 1994; bb) Zhang el al. 2010; cc) Van Maren 2004; dd) Yang el al. 2003; ee) Yang el al. 2007; ff) Yang el al. 2005; gg) Yang el al. 2006; hh) Peng el al. 2010; ii) Saito el al. 2007; jj) Lóczy 2007; kk) Panin, 1999; ll) Sarwar and Woodroffe 2013; mm) Hedley et al. 2010; nn) Coleman et al. 2008; oo) Sabatier et al., 2009; pp) Mertes and Magadzire 2008; qq) Syvitski and Milliman 2007; rr) Anthony et al., 2012

Table 2. Reach characterization of the lower Mekong River (from Rubin et al., 2014)

Reach Location	Adamson 2001 and Carling 2009a	Gupta 2004	Representative reach characteristics	Expected Changes
China	Zone 1: China	-	Not applicable	
Upper Bedrock Chinese border to 5km upstream of Vientiane	Zone 2: Bedrock single-thread channel - Chiang Saen to Vientiane: deep pools, bedrock benches	1a, 1b, 1c, 1d	Gradient: 0.0003 Channel width: 200m to 2000m Low flow depth: 10m Seasonal stage change: 20m	Negligible downcutting. Erosion limited to stripping of alluvial deposits overlying bedrock (bars, islands, inset floodplains, banks)
Middle Alluvial Vientiane to Savannakhet	Zone 3: Alluvial single-thread or divided channel	2a, 2b	Gradient: 0.0001 Channel width: 800m to 1300m Low flow depth: 3m Seasonal stage change: 13m	Alluvial bed and banks susceptible to erosion. Both downcutting and bank erosion likely.
Middle Bedrock Savannakhet to Kratie	Zone 3 continued (Savannakhet to Pakse). Zone 4: Bedrock anastomosed channels: Pakse to Kratie i.e. Siphandone (4000 islands reach)	3, 4, 5, 6	Gradient: 0.00006-.0005 Channel width: 750 to 5000m Reach length: 400km Low flow depth: ≤ 5 to 8 m Seasonal stage change: 9-15 m	Negligible downcutting. Erosion limited to stripping of alluvial deposits overlying bedrock (bars, islands, inset floodplains, banks)
Cambodian Alluvial Kratie to Phnom Penh	Zone 5A: Alluvial meandering/ anastomosed channels - Kratie to Phnom Penh: scroll bars, backwaters, overbank flooding, i.e. upstream of confluence with Tonlé Sap River Zone 5B: Tonlé Sap Lake and River seasonally reversing flows	6, 7	Gradient: 0.000005 Channel width: ≤4km. Floodplain width: 8 to 64km Low flow depth: 5m Seasonal stage change: 18m	Alluvial bed and banks susceptible to erosion. Both downcutting and bank erosion likely.
Vietnamese Delta Phnom Penh to ocean	Zone 6: Alluvial deltaic channels- Phnom Penh to ocean: distributaries, no marine influence in upper delta	8	Gradient: 0.000005 Channel width: ≤3km Delta inundation width: ~180km Low flow depth: 25m Seasonal stage change: 15m	Reduced rates of aggradation, shrinking delta, increasing risk of flooding from river and storm surge.

Table 3. Lower Mekong River sediment yields by geomorphic province (from Kondolf et al., 2014)

Geomorphic Province	**Description**	**Estimated Sediment Yield ($t\ km^{-2}\ y^{-1}$)**
Lancang	Active tectonics, and complex geology. Mekong River follows the fault between Sibumasu Block and older block and older block from South China-Indochina merge. High altitude, steep topography.	450
Northern Highlands (NH)	Hard sandstones and limestones (Paleozoic), granites and metamorphic rocks. Late Miocene uplift	250
Loei Fold Belt (LFB)	Hard sandstones and limestones (Paleozoic), granites and metamorphic rocks. Late Miocene uplift	160
Mun-Chi Basin (MC)	Sandstones of early Cretaceous Khorat Group: almost exclusively quartz sandstones. This has the lowest relief and appears to be the oldest landscape, may be a relict of older, pre-Miocene drainage system, with little recent uplift. This area has been extensively modified for agriculture and other development, so erosion and sediment yields may have been anthropically increased in recent years, but these sediments would probably be dominantly fine grained	40
Annamite Mountains (AM)	Hard sandstones and limestones (Paleozoic), granites and metamorphic rocks. Late Miocene uplift	200
Kon Tum Massif (KTM)	Heterogeneous geology of Paleozoic sedimentary rocks and igneous intrusive rocks, along with Khorat Group and younger Cenozoic basalts. Significant late Miocene uplift as reflected in deeply incised channels.	280
Tertiary Volcanic Plateau (TVP)	Heterogeneous geology of igneous intrusive rocks, younger Cenozoic basalts, and underlying Paleozoic sedimentary rocks. Significant late Miocene uplift as reflected in deeply incised channels.	290
Tonle Sap (TS)	The Tonle Sap basin consists mostly of lowland floodplain and small, short tributary drainage basins in the surrounding mountains. Net deposition (from Mekong River backwater) exceeds net sediment export.	0
Delta (D)	Net deposition	0

Chapter 2. Improving monitoring: identifying shortcomings in stream restoration evaluation

Abstract

In a search for accountability, the effectiveness of many large restoration programs has been evaluated using standards such as acres or length of stream restored per dollar, but this is inadequate. Another common restoration metric is based on the common goal of enhancing ecosystems by creating more complex habitats. Although widely implemented, there is little understanding of the success to date of such projects. There is also little agreement on the best approaches and metrics for quantifying success. We reviewed the methods of 26 peer-reviewed evaluation studies and investigated the influence of study design on evaluation results. Of the 26 studies, many did not implement rigorous study designs. For example, only 46% of the studies used quantitative measures of habitat, 62% included only one year of post-project monitoring, 46% used zero or one control (unrestored) sites, and 62% did not include reference (best potential ecological condition) sites. Studies that used more rigorous designs (e.g. sampled more years, measured habitat quantitatively) were more likely to find increased taxonomic diversity and richness in response to heterogeneity enhancement. More fundamentally, all studies used macroinvertebrate diversity and/or richness as the measure of ecological success. We question the logic of assuming that reach-scale diversity or richness is useful as a universal measure of ecosystem integrity. Monitoring and evaluation should first establish hypotheses and conceptual models based on watershed perturbations and set specific milestones towards a sustainable, dynamic, and healthy ecosystem. Restoration targets should be defined based on regional, historical, and analytical reference conditions and by conducting manipulative experiments that can help predict ecosystem responses to restoration actions. It is important to understand if habitat heterogeneity projects are succeeding, but fundamental questions persist regarding what indicators actually define success. Metrics to evaluate performance of stream restoration projects need more rigor and should be tied to project specific goals. Generic metrics may yield misleading results.

Introduction

In contrast to sediment starvation, the easily interpreted indicator for quantifying ecosystem impacts discussed in Chapter 1, the theoretical and practical considerations for quantifying restoration are much more complex. One fundamental problem is that the metric that is used to quantify success can be achieved through many pathways. In practice, this problem of equifinality means that one might "restore" the riparian trees of the Colorado River (as discussed in Chapter 3) in several ways and not all ways will be functionally equivalent. Essentially, while the question of impacts can often be assessed with simple and obvious metrics, quantifying restoration requires assessment of ecosystem functions, much more difficult problems. In Chapter 2 we highlight current shortcomings with some common methods of restoration evaluation.

Instream habitat improvement through rock and log additions and channel reconfiguration is among the most common restoration goals in the US [*Bernhardt et al.*, 2005]. This approach to restoration is based on a large body of research since the 1960's linking species diversity to habitat heterogeneity [e.g., Simpson, 1949; MacArthur and Wilson, 1967; Ricklefs and Schluter, 1993]. Fundamentally, habitat heterogeneity enhancement is assumed to provide more ecological niches for members of a community [*Warfe et al.*, 2008], provide refugia that stabilize predator prey and host-pathogen dynamics, and generally support greater diversity and a more resilient ecosystem [Bell et al., 1991, *Palmer et al.*, 2010].

Habitat heterogeneity is usually not explicitly defined or measured in restoration projects, which makes it challenging to determine whether streams were truly degraded with respect to channel heterogeneity [*Laub et al.*, 2012] and whether restorations actually increase heterogeneity. Many habitat heterogeneity evaluations have used metrics based on macroinvertebrate composition to evaluate the biological success of habitat heterogeneity enhancement projects. Both Miller et al. (2010) and Palmer et al. [2010] independently reviewed studies using macroinvertebrates to measure the success of heterogeneity enhancement projects, but reached divergent conclusions. In their review of 24 published studies, Miller et al. [2010] used quantitative meta-analysis and differentiated replicated from unreplicated studies and concluded that heterogeneity enhancement projects had increased the richness, but not diversity of macroinvertebrates. Large woody debris additions yielded larger and more consistent increases in richness than boulder additions or channel reconfiguration. The parallel review by Palmer et al. [2010] compiled 18 studies but did not combine the studies to test for statistical significance. Evaluating each study independently, Palmer and colleagues found no evidence that increasing habitat heterogeneity increased stream invertebrate diversity.

Both Miller et al [2010] and Palmer et al [2010] cited the lack of robustness in the studies as a potentially important limitation. In particular, Miller et al [2010] noted the "(1) low quantity and poor quality of published biotic and abiotic data; (2) lack of rigorous study designs; (3) a dearth of replicated restoration efforts within physiographically similar areas". However, neither study conducted in-depth assessments of the quality and methods of the studies they reviewed, nor evaluated the influence of study quality on the evaluation results. We were intrigued that these two studies, published in the same year, and both reviewing multiple prior published studies, reached such divergent conclusions about the effectiveness of restoration projects designed to increase habitat heterogeneity. To better understand why, we carefully analyzed all the individual studies in the two reviews.

In light of the widespread implementation of habitat heterogeneity enhancement projects and the substantial public investment they represent, the question of their effectiveness is an important one. Many restoration failures documented in the literature [see discussion in *Palmer et al.*, 2010] have led to calls for better project evaluation and accountability in restoration. But rather than simply calling for "monitoring", the question can be cast as "how can monitoring most effectively produce meaningful results?" To address this question it is necessary to first establish how monitoring is currently being performed and then to recommend best practices and the usefulness/limitations of success criteria. We question the assumption that macroinvertebrate abundance, diversity, and richness are always appropriate as success criteria. Instead of searching for universal success criteria we recommend evaluations be designed as ecosystem experiments with targets and testable hypotheses specific to the system of interest. At present, some studies use generic metrics without explaining the suitability of that metric.

Methods

We sought to determine how the different methods and metrics used might have influenced determination of "success". Previous studies have not formally questioned the universality of metrics such as macroinvertebrate diversity and richness. With these goals, we conducted a systematic analysis of methods and metrics in all the studies used by Palmer et al. [2010] and/or Miller et al. [2010]. We also reviewed observational and theoretical studies of food webs and species interactions. Of the 26 habitat heterogeneity enhancement studies reviewed by either Palmer et al. [2010], or Miller et al. [2010], 14 were included in both reviews (Table 1). We categorized each of the 26 studies according to twelve criteria representing a range of important considerations for restoration evaluation as presented in publications such as [*Palmer et al.*, 2005; *Wohl et al.*, 2005; and Kondolf et al., in prep]. These criteria included location, the extent of pre- and post-project monitoring (to control for temporal variability), sampling frequency (to control for seasonal variability), and whether the studies sampled different habitats (e.g. pool, riffle, banks) separately, in aggregate, or were restricted to only certain habitat types (Table 2). We also noted whether control (degraded, unrestored) sites were included to control for temporal variability and provide basis for comparison, whether regional reference (e.g. nearby, best potential ecological condition) sites were included to control for spatial variability and provide a regionally appropriate standard of success, what standards of success were stated and employed, and whether studies measured habitat heterogeneity directly, visually, or not at all. Other considerations were whether potential construction impacts from the restoration project were considered, and whether the regional reference sites had comparable drainage areas and slopes (Table 2). We scrutinized the methods and designs of each individual study, to assess the likelihood that the study designs employed would be adequate to detect biological change.

Results

Of the 26 studies reviewed by Miller et al. (2010) and Palmer et al. [2010], many did not implement rigorous study designs. Interestingly, the more rigorous studies were more likely to detect increased ecosystem diversity and richness in response to heterogeneity enhancement.

Habitat and biological metrics used

Although all studies sought to evaluate the biotic response to habitat heterogeneity enhancement projects, only 12 studies (46%) quantified habitat alteration. The other studies either did not report any data about habitats (six studies, 23%) or used visual estimates of grain size or subjectively judged 'habitat quality' (eight studies, 31%) with protocols such as EPA's Rapid Bioassessment Protocol (EPA-RBP) and Ohio's Qualitative Habitat Evaluation Index (QHEI). To accurately measure bed material size and variation requires use of a scientifically credible approach such as facies mapping and sampling sediment sizes in distinct habitats by pebble count (Wolman, 1954; Kondolf and Li, 1992). Among all metrics used to describe habitat heterogeneity (Table 1), depth variability was included in most studies that included direct habitat heterogeneity measurements. Other common metrics were velocity (and velocity variability) and substrate (either by visual estimates or direct measurements of average grain size and/or spatial distribution of grain sizes). Other habitat heterogeneity measures such as wood and organic matter retention, log or pool spacing and bank erosion were used less frequently (Table 1).

Most studies presented more than one biological metric to test improvement after restoration (Table 1). Taxonomic richness of macroinvertebrates was the most common biological metric used to test the effects of habitat heterogeneity enhancements (21 studies, 81%). Abundance and density of macroinvertebrates were also highly used (17 studies, 65%). Other diversity measures such as Shannon index or evenness indexes were used by 13 studies (50%). Composition of macroinvertebrate community or assemblage was used in nine studies (35%). Functional measures, such as functional feeding groups or trait composition were also used in nine studies (35%). Biological indices (such as the Benthic Index of Biotic Integrity B-IBI) were used in six studies (23%).

Controlling for time and variability

Of the 26 studies we reviewed, 16 (62%) included only one year of post-project monitoring (Table 2). Fourteen studies (54%) sampled only once per year, a frequency that is inadequate for seasonal environments. Sixteen studies (62%) had no pre-project monitoring and only one study (4%) included more than one year of pre-project monitoring.

Controlling for Space and variability

Although the projects were designed specifically to create habitat heterogeneity (e.g., pools and riffles, velocity and depth heterogeneity, etc.) eight studies (31%) either sampled only riffles or did not specify what habitats were sampled. Ten studies (38%) sampled and analyzed biota from different habitats separately, and eight (31%) used either pooled or random designs that integrated samples from all habitat types.

Twelve studies (46%) had either no control or only one control site, eight (31%) used two or three controls and (23%) included seven to thirteen control sites. Sixteen studies (62%) did not include a regional reference site, three (12%) used one reference site, and six (23%) used 2-5 reference sites. Therefore, only nine studies (35%) used any reference ecosystems as the standard of success. The other studies compared the restored reach to pre-restoration conditions or to a control site where legacy impacts persisted.

Of the nine studies that included regional reference sites, only four presented watershed attributes for the reference sites (we were able to find basic attributes for one additional reference site using the coordinates and USGS StreamStats). Of the five studies for which attribute data were available, three were substantially smaller and steeper than the treatment stream, though one study used a reference site whose drainage area was five times larger than the treatment site and about 30% urbanized (USGS Streamstats).

Relationship between study design and biological improvement

Seventy-eight percent of the studies that monitored for more than one year found increased diversity or richness (Figure 1). By contrast, only 44% of the studies that sampled for only one year found increases. This effort-diversity relationship suggests that either studies did not sample enough to detect statistically significant differences or perhaps the studies that sampled for several years were also designed rigorously in other ways that helped detect changes. Similarly, studies that evaluated habitat directly were more likely to find increases (66%) than those using visual estimates or not presenting any habitat measures (50% in both cases) perhaps reflecting another relationship between study effort and the significance of results. Again, studies using multihabitat sampling for macroinvertebrates either by pooled samples or differentiated habitats were also most likely to find improvement (75% and 60 % respectively, Figure 1). By contrast, only 20% of studies that sampled only one habitat (riffles) found increased diversity/richness. In cases where log or rock structures were designed to create pools, macroinvertebrate communities could be expected to shift towards more pool-dwelling taxa. In cases where riffles were constructed, communities may shift towards more riffle-dwelling taxa. Regardless of the restoration approach, it is imperative to measure different habitats separately. In sum, we found that the studies that conducted longer duration and/or more rigorous evaluations were more likely to detect increased richness or diversity.

Discussion and Conclusions

Based on this review, we challenge the assumption inherent in all the reviewed habitat enhancement assessments that increased diversity and/or richness are universally appropriate standards for restoration success. The meta-analyses of Palmer et al. [2010] and Miller et al. [2010] sought to answer questions about the effectiveness of habitat heterogeneity enhancement projects, a common style of restoration. Results of our systematic analysis of the 26 studies reviewed by Palmer and/or Miller suggest that many studies did not utilize the best practices in evaluation. Thus, the basis for drawing conclusions about the success to date of habitat heterogeneity enhancement efforts is limited at best. Further, most studies adopted increased diversity/richness as the sole standard of success without considering functional feeding groups, species targets for restoration, or other specific hypotheses related to the ecosystem of interest.

Metrics of assessment

The need for rigorous evaluation of habitat heterogeneity projects has been noted for many years [Reeves et al., 1991; Kondolf and Micheli, 1995; Chapman, 1996; Kauffman et al., 1997]. Early evaluations of instream structures focused primarily on longevity of constructed features or pool area [Roni et al., 2005]. As restoration science matures and funding opportunities become more competitive, complex questions about the objectives of restoration are increasingly raised. Recent studies question the mentality that drives restoration [*Katz*, 1992], when the river can "heal itself" [*Kondolf*, 2011], how restoration interacts with carbon cycling [*Madej*, 2010], sustainability of restoration projects [*Palmer et al.*, 2005], and the societal benefits of restoration [*Dufour and Piégay*, 2009]. Technical advancements in methods such as how to remeander streams [*Shields et al.*, 2003], create riffles and pools [*Newbury*, 1995], and stabilize banks [*Li and Eddleman*, 2002] have occurred rapidly in recent decades. Nevertheless, there are ongoing challenges to clearly articulate goals and to establish meaningful and feasible metrics for quantifying success.

Efforts to clearly distinguish and define terms such as "restoration", "rehabilitation", "enhancement", and "success" have been ongoing for several decades [e.g. *Lewis*, 1990]. Nonetheless, despite the inherent complexity in the topic, only 9 of the 26 papers (35%) stated the rationale for their selection of evaluation metrics. Instead, studies adopted standard metrics and monitoring protocols for habitat quality (e.g. EPA's Rapid Bioassessment Protocol (RBP) [*Lazorchak et al.*, 1998], Ohio's Qualitative Habitat Evaluation Index (QHEI) [*Rankin*, 1989; 2006], Bank Erosion

Hazard Index (BEHI) [*Rosgen*, 2001] or easy-to-measure habitat parameters such as pool area, and flow depth and velocity. These habitat quality metrics were primarily developed for quick implementation at a large number of sites and may not be appropriate for detailed restoration evaluation. Indeed many of the parameters included in the habitat quality protocols can be engineered without ecosystem benefit. For example, a narrow, sinuous, stable-banked stream may be achieved with heavy equipment and large boulders but might not be an appropriate target for all streams. All studies used taxonomic diversity or richness of invertebrates as biologic indices without presenting arguments why those metrics were relevant measures of success. Habitat heterogeneity projects are rarely done explicitly for biodiversity goals, so the logic and meaning behind the use of this metric is vague though presumably used as a general assessment of overall ecosystem health. In a review of biodiversity monitoring, Yoccoz (2001) quotes Krebs (1991) "Monitoring of populations is politically attractive but ecologically banal unless it is coupled with experimental work to understand the mechanisms behind system changes". Krebs continues that even if we knew population sizes for every species on earth we could not help conservation managers unless we also knew the mechanisms behind system changes. Considering such complexity, it becomes less clear why macroinvertebrate diversity is appropriate as a generic assessment of ecosystem condition. Additionally, of the 26 studies, five used the abundance of EPT (Ephemeroptera, Plecoptera, Trichoptera) taxa or the Benthic Index of Biotic Integrity (B-IBI) [*Kerans and Karr*, 1994] as measures of success. Developed as measures of water quality, it is not obvious why reach-scale habitat heterogeneity projects should be expected to increase the abundance of sensitive taxa.

The visual habitat assessments that are a part of habitat quality protocols (including RBP, QHEI, and BEHI) can lead to erroneous estimates. For example, Kondolf & Li (1992) found that visual estimates of substrate overestimated median grain size when compared to the repeatable method of pebble counts (*Wolman* 1954). Whitacre and colleagues (2007) compared six rapid assessment protocols and found statistically significant differences in results for nine out of ten basic habitat attributes such as sinuosity, percent pools, and median grain size. The importance of quantitatively and precisely assessing habitat heterogeneity was further documented by Laub et al. (2012) who found that many unrestored (but non-channelized) urban streams had relatively high heterogeneity when compared to reference, forested sites and that restoration sites were often not more complex than unrestored sites. Strikingly, the findings from Laub suggest that "heterogeneity enhanced" sites may not have more heterogeneity than unrestored sites. More comprehensively, Lisle and colleagues (2014) questioned the basic premise of using rapid assessments to evaluate the sediment impairment of gravel-bed streams based on a single metric or protocol because any single metric is unlikely to reveal causative relations and channel condition can result from multiple pathways. Instead, Lisle argued that channel condition be interpreted through the context of predictive mapping, site history and human influence.

There are no universally applicable measures of habitat or biotic quality

As statistician John Tukey [1962] famously stated, "Far better an approximate answer to the right question, which is often vague, than an exact answer to the wrong question, which can always be made precise." Standard metrics cannot be an alternative to thoughtful consideration of the local conditions and processes, and hypotheses directly related to those conditions and project objectives. For example, using bank erosion as a general measure of habitat quality either assumes that excess erosion is occurring at the site or that all bank erosion is undesirable. To equate "failure" with bank erosion or the displacement of an in-channel structure, wrongly assumes that successful stream restoration should create fixed, "stable" streams [*Bennett et al.*, 2011] even though ecological theory shows many *benefits* of bank erosion including delivering spawning gravel to the channel (California Department of Water Resources, 1984) and facilitating riparian vegetation succession [*Piegay et al., 1997*; *Florsheim et al.*, 2008], and providing habitat for early successional plants or disturbance-dependent species like bank swallows and yellow-billed cuckoo [Gardali et al., 2006; Golet et al., 2013]. These examples call into question the underlying premise of habitat heterogeneity projects which seek to establish persisting forms rather than the dynamic processes that create those forms.

Reach-scale diversity is not a universal indicator of success.

At the regional or global scale, changes in biodiversity may reflect overall ecological condition. However, in the context of reach-scale restoration, the "more is better" diversity assumption may not be not appropriate. Consider a few cases as "thought experiments": a desert stream, a glacial outwash stream, and a headwater stream in old-growth forest. Human impacts could result in increased macroinvertebrate diversity (at the expense of native species) in all these cases.

Development and its resultant "urban slobber" may perennialize the desert stream and allow many new species to colonize. In such systems, richness and diversity have been strongly correlated with stream permanence, though 7% of species were found only in intermittent streams (*Feminella, 1996*). In the glacial outwash stream example, a warming climate may allow new species to establish, increasing diversity but threatening endemic specialists (Jacobsen et al. 2012). In the headwater stream example, logging of the old-growth forest could allow light penetration to the stream and increase diversity of habitats and species (Hernandez et al. 2005). In all of these examples, diversity may be increased through additional generalist species, but communities of native, endemic, and/or target species will not be supported. More specifically addressing this issue in the context of restoration, Lepori et al. (2005) found less macroinvertebrate species richness in reference streams than in either channelized or restored streams. In sum, an increase or decrease in species abundance or diversity may be a poor measure from which to infer restoration success unless site-specific and species-specific targets are defined and justified.

Not all species are equal

Food web interactions may be far more significant than diversity or abundance for influencing populations of the top predators often targeted by restoration. For example, in the Eel River, California, Power et al. [1996] found that scouring winter floods promoted trophic interactions that produced more prey for steelhead, whereas drought years (or flow regulation) favored grazing macroinvertebrates that do not feed steelhead. In this case, we can see that it's not necessarily the abundance or diversity of species that is important to top predators, but rather which species and energy pathways become dominant. In other cases the removal of a key species can affect the whole ecosystem, like the Amazonian fish *Prochilodus mariae* whose removal from the Rio Las Marias in Venezuela altered nutrient cycling, sediment structure and diatom and macroinvertebrate assemblages (Flecker, 1996, Taylor et al., 2006). These examples suggest that restoration monitoring requires knowledge of the biophysical interactions that underpin the structure and dynamics of the target ecosystem.

Controlling for variability in time

Prolonged monitoring is necessary to discern the influence of restoration activities on biota. For example, the disturbance caused by the restoration activity itself (i.e. vegetation clearing, dewatering the channel, compaction from heavy machinery in the channel, grading the bed and banks) may decrease or increase biotic metrics such as abundance and diversity, and that influence may last days or decades, depending on severity and rates of recovery processes. Many studies suggest the need to monitor at least several years in order to allow benthos to recover and to recolonize a restored reach [Miller et al., 2010]. Conversely, disturbances caused by restoration actions may actually enhance diversity over intermediate time scales [Connell 1978], so both pre-project monitoring and reference targets would be needed to interpret whether biotic differences following restoration are due to restored conditions, or to temporary effects.

Controlling for variability in space

Several decades of research have established how watershed conditions and position in the river system influence channel processes and forms [*Leopold and Maddock*, 1953] and ecosystem characteristics [*Vannote et al.*, 1980; Ward and Stanford, 1983]. However, local conditions such as channel geometry can vary greatly over small distances or short time periods [*Waters and Haynes, 2001*] because of changes in lithology and vegetation, tributary confluences, beaver activity, climate, or land use [*Montgomery*, 1999; *Polvi et al.*, 2011; *Wohl*, 2010]. Therefore, it is essential to compile as much information as possible from historical reference periods, multiple regional reference sites (using space for time substitution), and patterns, gradients, and processes that support defining analytical reference states to quantify the range of natural variability of form and process over time.

Reference states and best practices

Appropriate **regional reference** sites may be challenging or impossible to find [*White and Walker*, 1997; *Whittier et al.*, 2007], especially for larger watersheds. Watershed position, regional climate, disturbance history, tectonic uplift rates, local geology, and many other factors will all influence stream processes and forms. However, physical controls such as channel slope, sinuosity, bed-material size, and precipitation are increasingly identifiable through automated monitoring and remote sensing. That makes identification of appropriate reference sites practical using GIS and we expect analytical models will produce increasingly reliable predictions of watershed condition based these physical controls. Watershed comparisons have traditionally used a "paired basin" approach, in which "treated" are compared to "untreated" reference basins. Paired watershed experiments suffer from at least three problems (as reviewed by Reid et al. I981) similar to those outlined above for restoration evaluations. First, "treatment" variables are usually only qualitatively characterized (e.g., "Managed vs. unmanaged" or "logged vs. unlogged'. Second, "control" treatments are never pristine. Third, even if untreated and treated watersheds have been matched with respect to aspect, area, slope, forest type, drainage density, and geological parent material, they may differ in subtle but important respects (e.g., structural orientation of bedrock, undetected ancient landslides whose scars are presently filled, disease or fire history of vegetation). Comparisons of watershed outputs, such as total sediment yield at their mouths or changes in salmon escapement back to watersheds over the experimental period, are too noisy to reveal causality, particularly when observation records are short (less than decades).

Human alterations are widespread, and landscapes may still be responding to impacts from decades or centuries ago. Nonetheless, we argue that without reference sites, restoration targets will likely rely upon generic standards that may not be regionally or locally appropriate. Several reference streams are almost certainly required to adequately account for variability between streams and to test assumptions about variability in target conditions. Using an inappropriate reference stream (with different slope, drainage area, hydrologic regime, watershed position, etc.) is not useful, and having only one reference stream is unlikely to help understand the natural range of processes and communities. Of the 26 studies we reviewed, only nine used reference streams, and of those, only four presented information about the reference watershed. Of those four, two accepted considerable differences between fundamental watershed characteristics for reference and restored streams: in one study the drainage area of the reference site was five times larger than the restored stream.

The conceptual model linking restoration actions and the anticipated versus observed biological outcomes should be well defined**.** The states of ecosystems past, present, and future can be evaluated quantitatively only with reference to some baseline set of conditions or "reference state". Above, we distinguished control (unrestored) sites from "reference" sites that are nearby, with better ecological conditions towards which restorers hope to move more degraded sites. There are three other types of reference states (Table 3) that may be useful in evaluating restoration targets and success or failure (Power et al. 1998). The commonly used **historical reference state** is chosen from many past states to represent the conditions we would expect to observe in our impacted ecosystem had it not experienced degradation. The problems of deciding which historical time slice to choose as the proper baseline, let alone documenting sufficiently the conditions that prevailed at the time, have been much discussed in restoration literature. More useful than the single historical reference state is analysis of the historical range of variability (e.g. Morgan et al., 1994; Wohl 2011; Rubin et al., 2012) which considers all available historical information to understand natural fluctuations in processes and forms in a system prior to intensive human disturbance. As an alternative, Paine (1994) proposed evaluating noisy nature relative to a simplified reference state, produced by doing field experiments to remove certain players or processes to produce a **manipulated reference state** intended to be the product of simplified, well-understood interactions. Once defined, this manipulated, simplified reference state could be used to evaluate the outcomes produced by adding back natural complexity (large predators, habitat structures, etc.) whose effects over time could be studied by evaluating the deviations they produce from the simpler reference state. Finally, W.E. Dietrich (personal communication) has advocated using an **analytical reference state** for comparing ecosystems at larger (reach to watershed) spatial scales. Analytical reference states emerge as expectations of the condition of habitat from its biogeographic and landscape context. For example, the bankfull depth and water surface slope of a river reach allows us to predict from theory the expected median grain size of the bed (e.g. Snyder et al., 2013). Deviations from this expected grain size arise from effects of woody debris and bar resistance (Buffington 1995) and from perturbations of sediment supply (Dietrich et al. 1989, Buffington 1995). An over-supply of gravel (e.g. from a landslide or poorly engineered roads upstream) will reduce median sediment size on a river bed relative to analytical expectations. A deficit of gravel supply (e.g. from a dam upstream) could coarsen bed sediments. This simple example may inspire other efforts to "read the river" locally to evaluate, ahead of field examination, what should be predicted at the site from map-ascertained characteristics of sites and their known (or assumed) relationship of these characteristics with flows and accumulations of watershed "currencies": water, sediments, organic matter, solutes, momentum, heat, gases and organisms. The analytical reference state may be the most robust framework for determining what could be achieved, were the restoration to succeed. Although the applications of analytical methods are currently somewhat limited, these approaches are increasingly accurate and could benefit restoration planning, not only in setting evaluation targets, but also in determining what restoration actions, if any, are warranted. For example, before planning to treat an incised stream, a target range of width: depth ratios should be predicted based on all available types of references. How the current conditions compare to regional, historical, manipulated, and analytical reference targets can help managers decide if channel alteration is warranted (e.g., is the stream degraded at all?) and if recovery can be expected through passive means.

Designs of restoration monitoring projects

While the purpose of this paper is not to advocate specific monitoring approaches, we highlight some studies with thoughtful or exemplary designs. Lester et al. [2007] is a good example of a paired design with control sites and restored sites chosen based on a broad set of variables (median and maximum flow velocity, water depth, average substrate particle size, complexity of in-stream habitat, nutrient concentrations, catchment area and riparian vegetation composition). Studies that conduct biological sampling with a multihabitat approach are better suited to capture the enhanced heterogeneity that was the restoration goal. Sampling edges and in-stream habitats separately may be a useful sampling design for a homogenous reach (as implemented by Nakano and Nakamura [2006] and Harrison et al. [2004]). Other more complete designs are randomly stratified samples, as used by Negishi and Richardson [2003], where velocity, depth, substrate and invertebrates were sampled randomly with several replicates in space and time. Complete and thoughtful habitat measurements should also address whether restoration practices harm stream biota, as noticed by Muotka et al. [2002], where the removal of moss by heavy machinery used on the projects led to extensive biodiversity loss on sites.

Conclusion

Based on our review of literature reporting heterogeneity enhancement projects, we caution that generic metrics of habitat and biological condition should not be used to evaluate projects without careful consideration of whether those metrics are appropriate for a given case. Our ongoing research seeks to identify indicators that may be universally applied. Regardless of the indicators selected, monitoring should be driven by hypotheses based on knowledge of watershed controls, perturbation history, and biophysical interactions that could underlie a self-sustaining, dynamic, and resilient natural ecosystem. A portfolio approach combining knowledge from regional and historical reference streams, unrestored (control) sites, analytical models and manipulative experiments, is required for defining relevant target conditions, though such an approach will add to the time and effort required for evaluation. Prior to restoration, project designers and evaluators should develop conceptual models (which increasingly should include analytical reference states) of their ecosystems and consider success criteria carefully, in light of predictions generated from these models.

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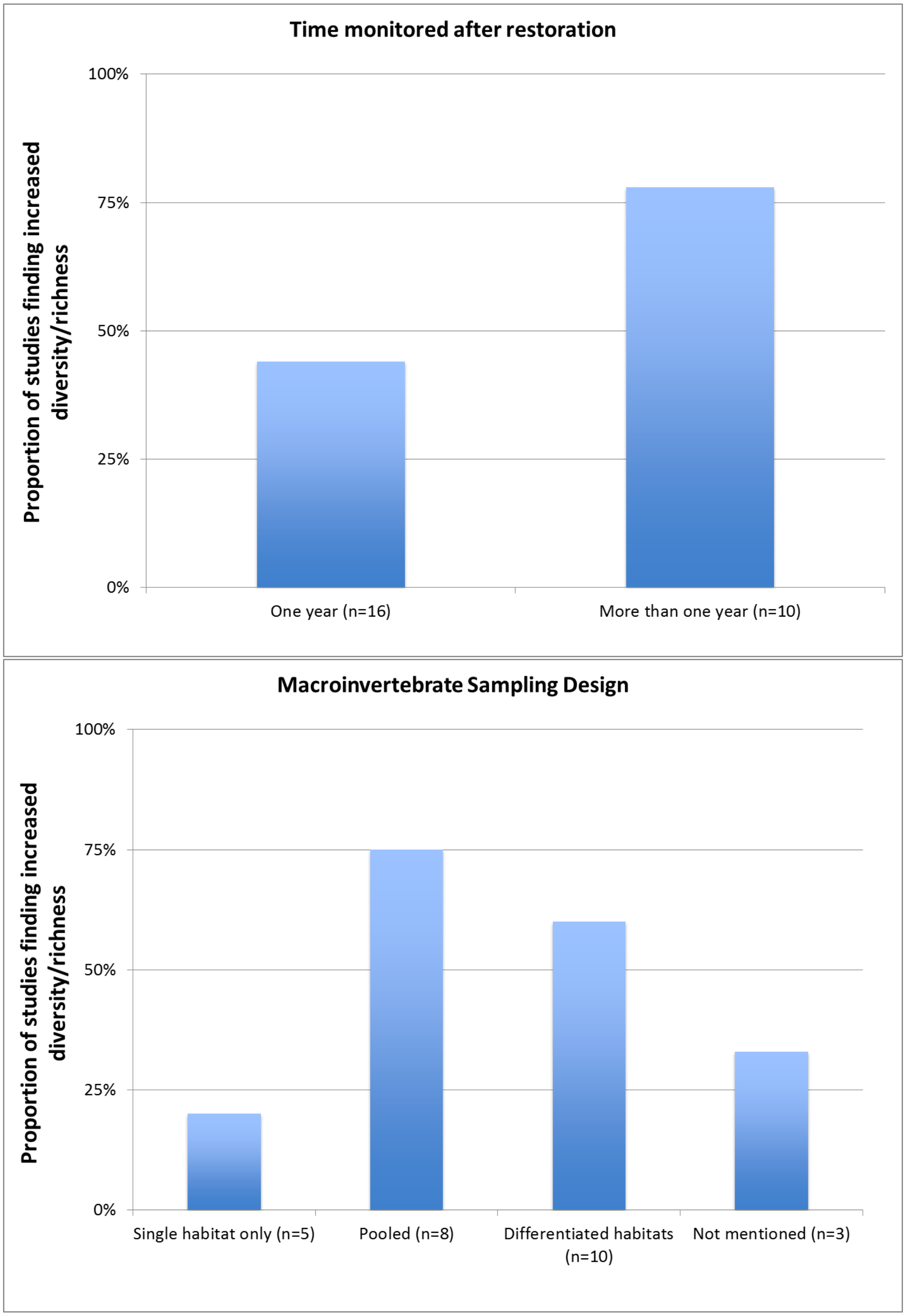

Figure 1. More rigorous study designs were more likely to detect increased macroinvertebrate diversity/richness. Of the 26 studies we reviewed, 16 monitored for one year after restoration and ten studies monitored for more than a year. The studies that monitored more than one year were almost twice as likely to find increased diversity/richness as the 16 studies that monitored for only 1 year (above). Similarly, studies that purposefully pooled habitats or sampled independently (differentiated) were more likely to detect diversity/richness (below).

Table 1

Study	Location	Habitat metrics	Biological metrics
Biggs et al. 1998	Brede river in Denmark, Cole river in UK		Taxonomic richness, abundance/cover and species rarity (data available for the UK only).
Ebrahimnezhad & Harper 1997	Harper's Brook (Northamptonshire, England)	D, V	Diversity and taxonomical differences among sites.
Edwards et al. 1984	Olentangy River, OH,USA		Taxonomic richness, abundance, diversity, biomas. For fish (number, biomass, species)
Friberg 1998	Jutland, Denmark	D, S, V	Taxonomic Richness, density, composition. Abundance of stone-dwelling species were analyzed to specific level.
Gerhard & Reich 2000	Central Germany	D, F, L, S, V, W	Maximum and average number of species and density.
Gortz 1998	River Esrom, Denmark	C, O, S, V	Saprobic index, Danish fauna index, invertebrate diversity, count of trout spawning reds.
Harrison et al. 2004	Several locations in UK	D, S, V	Taxonomic richness, Shannon diversity (H'), total abundance and the abundances of individual taxa
Jähning & Lorenz, 2008/Jähning et al 2009/Jähning et al. 2008	Lahn, Eder, Nims and Bröl Streams in Germany.	B, D, S, V, W	Taxonomical richness, similarity on compsition among substrates and channel types (single or multiple).
Jungwirth et al. 1993	Epipotamal and Melk Rivers, Austria	D, S, V	For macroinvertebrates number of species and drifting biomass. For fish number of species and diversity.
Laasonen et al. 1998	Head waters of Rivers Iijoki, Oulankajoki and Oulujoki in Finnland	D, S, V	Taxonomical richness, density and assemblage structure. Shredders density.
Larson et al. 2001	Pudget Sound Tributaries	D, L, P, S	Index of Biological Integrity (IBI).
Lemly and Hilderbrand 2000	Stony Creek, Virginia	P	Taxonomical richness and FFG.
Lepori et al. 2005	Ume River, Sweden	D,M, Q, S, V, W	Taxonomic richness, compostion, eveness of fish and macroinvertebrate. Taxonomical density for macroinvertebrates.
Lester et al. 2007	Gippslandand, south-west Victoria, Australia	W	Number of families, abundance , number of EPT families, average SIGNAL2 sensitivity score, Shannon's Evenness index (Shannon's E), and FFG.
Moerke et al. 2004	Juday Creek, Indiana, USA	C, G, L, S	Macroinvertebrate density and diversity. Benthic algae density. Fish richness and biomass
Muotka & Laasonen 2002	Northeastern Finland restored: (Kutinjoki, Kosterjoki, Loukusajoki	D, M, O, S, V	Densities of macroinvertebrates and FFG
Muotka et al. 2002	Same as Laasonen et al 1998	Same as Laasonen et al 1998	Densities of macroinvertebrates and Functional Feeding groups (FFG)
Nakano & Nakamura 2006/ 2008?	Shibetsu River in Northern Japan	D, S, V, Y	Taxonomic richness, density and compostion.
Negishi & Richardson 2003	Southwestern Brithish Columbia, Canada	D, S, V	Macroinvertebrate abundance, rarefacted richness and community compositon
Pedersen 2007	Skjern River, Denmark	D, S, V, W	Macrohpyte number of species and coverage. Macroinvertebrates: richness, abundance, Shannon diversity, EPT and dominant species frequency
Purcell et al 2002	S.F Bay, CA, USA	G	Taxonomic richness, family richness, total number of individuals, Family biotic index, EPT richness, % EPT individuals
Roni 2006	Lower Umpqua and Coquille River basins, Oregon, USA	L, P, S	Total abundance, Taxonomical richness,relative abundance (proportion of total abundance) of FFG (shredders and collectors) orders EPT tax , and I-IBI.
Rosi-Marshall 2006	Cook's Run, MI	D, G, O	Periphyton concentration and biomass.Macroinvertebrates: density, diversity and FFG composition. Fish: Trout abundance, % Age 1+Trout; % harvestable trout.
Sarriquet 2007	Tamoute River, France	H, Q, S	Composition, density, taxonomical richness
Tullos 2009	North Carolina Piedmont, USA	E, G, O, P, S	Taxonomical and Trait composition, and Shannon Diversity. Species indicator analysis of restored and un restored sites.
Walther & Whiles, 2008	Cache River, IL (USA)		Taxonomical diversity, richness, eveness and similarity, composition of assemblages. FFG.

B is braiding index, C is canopy cover or shading, D is depth, E is bank erosion, F is facies mapping, G is generic quality assessment (EPA's RBP, or Ohio's QHEI), H is hyporheic exchange, L is count or volume of wood, M is moss cover, O is Organic Content of substrate or leaf retention, P is pool spacing or pool area, Q is water quality, S is substrate size, V is velocity, W is width, Y is shear stress

Table 2. Sampling design summary

Location	14 (54%) of projects were in Europe. 10 (38%) were in North America, 1 (4%) in Australia, and 1 (4%) in Asia
Pre-project monitoring	16 (62%) had no pre-project monitoring. 9 (35%) monitored for one year and 1 (4%) monitored two years
Post-project monitoring	16 (62%) had one year of post-project monitoring. 5 (19%), 3 (12%), 1 (4%), 1(4%) sampled 2,3,4, and 5 years respectively
Sampling frequency	14 (54%) sampled once per year. 7 (27%), 4 (15%), and 1 (4%) sampled 2, 3, and 4 times per year respectively
Habitat sampled	5 (19%) sampled a single habitat exclusively, 10 (38%) sampled multiple habitats separately, 8 (31%) sampled multiple habitats in a pooled/random design, and 3 (12%) did not mention what habitats were sampled.
Control Sites	12 (46%) used 0 or 1 control sites, 8 (31%) used 2 or 3 control sites, and 6 (23%) used from 7 to 13 control sites.
Reference Sites	17 (65%) did not include a reference site, 3 (12%) used 1 reference, 6 (23%) used 2-5 reference sites
Standard of success	9 (35%) used a reference site as the standard of success for benthic macroinvertebrates. 17 (65%) compared restored to unrestored conditions (pre-restoration or control site)- typically assuming increased diversity, richness or B-IBI score is an improvement.
Habitat/substrate assessment	6 (23%) did not assess habitat/substrate. 8 (31%) usual visual assessments. 12 (46%) used quantitative measures to assess habitat.
Construction influence assessed/discussed	9 (35%) measured construction harm (through multiple measurements per year or multiple years, emphasizing the first year after construction and including pre-construction data or control sites.
Reference site parameters presented?	Of the 9 studies using reference stream conditions as the measure of success, 5 studies (55%) did not present any watershed attributes of the reference sites (watershed area, stream gradient, width, depth, discharge).
Land use assessed or discussed	50% discussed the land use of the catchment while 50% did not.

Table 3. Approaches for identifying reference conditions

Basis	Examples	Limitations
Regional	Analog least-disturbed sites to indicate the processes and forms present in similar watersheds with less impairment. Ideally several sites identified to provide a range of target forms and processes.	Challenging to identify enough appropriate sites to characterize natural variability.
Historical	Historical air or ground photography, paintings, written descriptions, satellite imagery, maps, stratigraphy, palynology used to indicate historical processes, forms, and biota.	Limited available sources, many sources required to categorize historical range of variability, nonstationarity of climate, and other legacy impacts,
Analytical	Predictions of physical habitat and biological communities (e.g. grain size, width, depth, channel planform and pattern) based on drainage area, slope, lithology, vegetation, and other controlling conditions. May be empirical or physically based models.	Models are increasingly accurate but not well developed in many cases.
Manipulated/ Experimental	Manipulative experiments to test interactions of different species and/or habitat alterations.	Useful to understand system dynamics under experimentally simplified conditions. May be time consuming to plan and conduct experiments.
Control	Sites that have been degraded in a manner similar to the reach being restored. Deviation from the condition of such sites towards the reference condition can indicate success. Prior to restoration, if control site (i.e. unrestored) condition does not significantly differ from reference condition, then the justification for restoration may be invalid.	Monitoring of control sites can help define variability over time and space in unrestored streams. Control sites can not be used to define target conditions, and merely evaluating change from the control condition can not be used to define success.

Chapter 3. Prey Availability as an Evaluation Metric in Riparian Restoration: Lower Colorado River, USA

Abstract

Below Hoover Dam, riparian vegetation along the Colorado River was extensively cleared for agriculture. Thus, large areas of habitat were lost to clearing. Moreover, the functions of the ecosystem were compromised as the connections of the river to its floodplain were severed by levees, flow reduction by dams and diversions, channel incision, and groundwater pumping. Subsequently, native species declined, including the southwestern willow flycatcher (*Empidonax traillii extimus*) that nests along rivers in dense riparian thickets. The Lower Colorado River Multi Species Conservation Program (MSCP) was established in 2005 to re-create habitat for 26 species including the flycatcher, but the benefits of these restoration sites for target species have not been quantified. Many MSCP projects have involved extensive plantings of willow (*Salix exigua, S. gooddingii*] and cottonwood [*Populus fremontii*) on high terraces disconnected from the river by levees. MSCP projects goals are specified as acres of habitat, but to support functioning food webs, riparian ecosystems in arid regions require a subsidy of aquatic insects. We documented prey availability for the southwestern willow flycatcher in constructed habitats as an indicator of their potential to support the species. The number of aquatic insects, proportion of aquatic insects, total number of insects, and number of insect orders all decreased with distance from the river, and the decrease occurred within the first 100 meters from the river. The MSCP cottonwood-willow plantation (more than 500 m from the river) at Cibola National Wildlife Refuge had 86% fewer total insects ($p=0.055$), 97% fewer aquatic insects ($p=.032$), and only half as many insect orders ($p=0.015$) as sites adjacent to the river. In the plantation, only 16% of the insects were aquatic vs. 59% aquatic at the river's edge ($p=.063$). Our results suggest that restoration success (and the recovery of southwestern willow flycatcher) may be limited by prey availability and that future riparian plantings should be concentrated along the river or tributary channels. Southwestern willow flycatchers have not been nesting in MSCP plantations. Thus the metric of "acres restored" is inadequate to capture ecosystem function. More meaningful metrics would identify potential limitations in ecosystems (such as prey availability) so that habitat suitability and functionality can be assessed and adaptively managed.

Introduction

Managers and scholars are increasingly interested in quantifying the effectiveness of ecosystem restoration projects, yet appropriate metrics are challenging to identify. Many restoration projects are implemented to benefit species of concern, yet because target populations may take decades or longer to respond to restoration activities, and because populations of target species may be strongly affected by factors unrelated to restoration actions, population measurements are often not appropriate for promptly evaluating restoration success. Therefore, many monitoring programs use surrogate metrics to evaluate restoration project performance. Surrogate evaluation metrics include invertebrate abundance and community diversity [*Muotka et al.*, 2002; *Pik et al.*, 2002], habitat complexity [see *Miller et al.*, 2010; *Palmer et al.*, 2010], and persistence of created habitat features [*Schmetterling and Pierce*, 1999]. As discussed in Chapter 2, there is a pressing need for practical indicators that can be provide insight on short time scales. Here in Chapter 3 we use prey availability for species of concern as an indicator of habitat performance. Prey availability was selected as an indicator based on knowledge of the region and is linked to project goals.

Neither habitat assessments nor population monitoring will provide insight unless restoration actions are connected to clear hypotheses linking physical processes and biological responses. Habitat assessments and species monitoring are common methods of restoration evaluations, yet projects rarely consider the functionality of physical habitat and the interactions of habitats with the many species that comprise food webs [*Lake et al.*, 2007]. By monitoring habitats in conjunction with long-term populations *and* assessing how the habitat is functioning, hypotheses can be tested, and restoration science can advance. Hypotheses could address questions relevant to the recovery of target populations, e.g.: "Does the habitat contain food for the species of concern? Are growth or survival rates of the target species higher in restored habitats? Can riparian vegetation establish, mature, and reproduce to create new or persisting habitat? If target species require specific seral communities, will these be available over the long term?" Ideally for restoration planning and evaluation, full-scale food-web studies should be implemented to test the interactions between habitat and species [e.g. *Cross et al.*, 2013; *Naiman et al.*, 2012]. However, even in the absence of detailed food-web studies, some quantification of prey availability may be feasible and can provide insights into system function. Here we present an example from the lower Colorado River in which data on prey availability for migratory birds in constructed habitats provide critical information about the missing links between the created habitat and populations of the target species.

Riparian zones of the arid and semi-arid western US

When viewed from above, the riparian areas of the western US are instantly recognizable as bands of green vegetation crossing extensive, dry, tan or grey landscapes. The riparian corridors consist of Fremont cottonwood (*Populus fremontii*), willow (*Salix spp: S. gooddingii, S. exigua, S. lutea, S. laevigata*, and others), and other native woody species, along with invasive exotics such as *Arundo donax*, salt cedar (*Tamarix spp*), and Russian olive (*Elaeagnus angustifolia L.*) [*Mitsch and Gosselink*, 1993; *Stromberg*, 1993]. Rivers of the southwestern US and other dryland areas have characteristically variable flow regimes, which produce dynamic episodes of disturbance and subsequent recovery of instream biota [*Arthington and Balcombe*, 2011; *Bunn et al.*, 2006]. The same disturbance and recovery episodes also control the establishment, expansion, persistence, and extent of riparian vegetation [*Campbell and Green*, 1968; *Douhovnikoff et al.*, 2005; *McBride and Strahan*, 1984]. Riparian vegetation along arid and semi-arid streams is structured and maintained by flooding [*Stromberg*, 2001; *Stromberg et al.*, 1991]. Early research by Lowe [1964] and Campbell and Green [1968] suggested that the interplay of disturbance (flood, drought, scour, channel avulsion) and recovery is so important that the riparian vegetation found following disturbance should not be regarded "merely a temporary unstable seral community" but instead as "a distinctive climax biotic community" known as "disturbance climax" or "disclimax".

Historical change on the lower Colorado River

The Colorado River is one of the most regulated in the world with total reservoir capacity equaling four to seven years of the river's total flow [*Rajagopalan et al.*, 2009; *Schmidt*, 2010]. Completion of Hoover Dam in 1936 and Glen Canyon Dam in 1963 greatly reduced flood magnitudes downstream. Massive water withdrawals (surface and shallow groundwater) for irrigation and municipal uses have reduced flows in the lower Colorado River [*Blinn and Poff*, 2005], and the annual discharge debouching into the Gulf of California has been eliminated completely in most years. The reduction in flood magnitude in the lower Colorado River following construction of Hoover Dam facilitated channel straightening, levee construction and conversion of floodplains to agriculture [*Norman et al.*, 2006]. With Glen Canyon and Hoover Dams each trapping more than 95% of inflowing sediment, essentially no sediment is transported downstream in the lower Colorado other than what is eroded from channel margins or carried in by local tributaries [*Schmidt*, 2008]. Downstream of dams, the channel has incised severely—up to 5 m in several reaches along the lower Colorado River [*Schmidt and Wilcock*, 2008]. This incision further disconnects the lower Colorado River from its historical floodplain and riparian corridor.

Widespread vegetation clearing occurred on the lower Colorado River in the 19th century, as mesquite, cottonwood, and willow were collected to fuel steamboats [*Ives*, 1861]. By 1890, easily accessible riparian trees had been largely eliminated [*Grinnell*, 1914; *Ohmart et al.*, 1988]. While riparian vegetation subsequently increased along much of the Colorado River and its tributaries above Hoover Dam during the 20th century [*Webb and Leake*, 2006; *Webb et al.*, 2007], the mainstem Colorado River below Hoover Dam experienced further reduction in riparian vegetation as a result of agricultural clearing, levees that disconnected the floodplain from the river, and channel incision and groundwater pumping that lowered floodplain groundwater levels by as much as 6 m [*Nagler et al.*, 2007]. Non-native *Tamarix*, which is better-adapted to the altered hydrology and saline soils, now dominates the remaining riparian areas that still exist along the lower Colorado [Stromberg et al., 2007].

With the loss of riparian habitat along the lower Colorado River [*Webb et al.*, 2007] populations of native species that depend on this habitat have also declined. An emblematic example is the southwestern willow flycatcher (*Empidonax traillii extimus*) whose population had declined to less than 1,000 individuals when they were federally listed as endangered in 1995 [*U.S Fish and Wildlife Service*, 2002]. The flycatcher nests from May to September in dense riparian thickets adjacent to the slow or ponded water found in secondary channels, backwaters, marshes, and sloughs associated with the river. Nest sites typically have dense foliage from the ground to ~4m high [*U.S Fish and Wildlife Service*, 2002]. The flycatcher relies on aquatic prey, at least when nesting [*Bakian*, 2011], though there is some debate on the overall importance of aquatic insects for the flycatcher through its life history [*Delay et al.*, 2002; *Wiesenborn and Heydon*, 2007]. Essentially, the flycatcher requires the kinds of habitat that existed when the rivers of the Southwest were unregulated: cottonwood and willow forests adjacent to river side channels and former channels, habitats maintained and renewed by fluvial dynamics such as frequent avulsions, flood scour and deposition [*Norman et al.*, 2006; *Webb et al.*, 2007]. These dynamic fluvial processes create surfaces for riparian vegetation colonization and diverse flood-plain water bodies. Dam-induced alteration to the timing, magnitude, and recession rate of floods has reduced cottonwood and willow establishment along the lower Colorado River mainstem [*Webb et al.*, 2007]. Conversely, reduced flood scour and elevated baseflows downstream of dams can permit long-term persistence of existing vegetation and encroachment of vegetation into the former active channel [*Douhovnikoff et al.*, 2005; *Pelzman*, 1973] as occurred throughout much of the Colorado basin [*Webb et al.*, 2007] . In either case (reduced establishment or encroachment) the riparian habitat under these static conditions differs from dynamic riparian habitat sustained along unregulated rivers. The habitat value of these stands has been questioned. While Sogge [2003] found 25% of all flycatcher nesting sites were in *Tamarix,* the debate about the habitat value of *Tamarix* is rapidly evolving, as *Diorhabda* beetles (introduced as biocontrol on *Tamarix*) are now rapidly spreading through the lower Colorado River basin, defoliating large areas of *Tamarix,* and making the future of *Tamarix* uncertain [*Paxton et al.*, 2011; *Sogge et al.*, 2008].

Restoring Riparian Food Webs in Semi-Arid Environments

In riparian ecosystems along dryland rivers, the net flux of energy and nutrients is typically from the stream to the riparian zone rather than from riparian zones to stream [*Marti et al.*, 2000]. Riparian vegetation along arid region streams is usually less dense, provides less shade, and produces less leaf litter and other organic matter to the stream than humid-climate ripa. As a result, instream primary productivity is the most important source of organic carbon for both aquatic and terrestrial consumers [*Finlay*, 2001]. Further, riparian species in arid zones produce litter with relatively low nutritional quality [*Francis and Sheldon*, 2002] thus making the leaves unpalatable to invertebrates, as reflected in low shredder densities in arid and semi-arid stream systems [*Davies et al.*, 1994; *Ward et al.*, 1986]. High primary productivity and efficient nutrient cycling in dryland streams is attributed to high light intensity, low current velocity, and high temperatures [*Bunn et al.*, 2006].

The aquatic-to-terrestrial subsidy of nutrients and water is the primary driver of productivity in arid-region riparian zones, as documented in a variety of prior studies. In Sycamore Creek, Arizona, 97% of aquatic insect emergence biomass was transferred to the terrestrial habitat where the insects were prey for terrestrial consumers such as bats, birds, and ants [*Jackson and Fisher*, 1986]. The abundance, biomass, and richness of spiders was highest at the creek edge and decreased more than three-fold 25 m from the stream margin [*Sanzone et al.*, 2003]. In Key Pittman Wildlife Management Area and Pahranagat National Wildlife Refuge, Nevada, insect biomass decreased~70% within 30m of the river's edge [*Theimer and Pellegrini*, 2011]. In the Mediterranean-climate Eel River of northern California, Power [2004] and Hagen and Sabo [2011] found that insect numbers decreased exponentially with distance from the river and cursorial and orb-web spiders, lizards, and insectivore bats were all more common closer to the river.

The reliance of terrestrial biota on instream productivity implies that restoring habitat for insectivores such as the southwestern willow flycatcher would require restoring not only habitat structure, but also functional food webs [*Sarriquet et al.*, 2007]. Freshwater food web restoration requires understanding the ecology of basal trophic levels (primary producers, primary and secondary consumers), and for insectivorous birds, how insect productivities and spatial fluxes are controlled by the features and dynamics of river channels and floodplains under various flow regimes.

Lower Colorado River Restoration

In recognition of the impacts of dams and diversions on aquatic and riparian species (including species listed as endangered or threatened under the federal Endangered Species Act), the Lower Colorado River Multi Species Conservation Program (MSCP), was established in 2005 to create habitat for 26 listed species of amphibians, birds, bats, fish, plants, reptiles, rodents, and insects, including seven endangered and threatened species: razorback sucker (*Xyrauchen texanus*), bonytail chub (*Gila elegans*), humpback chub (*Gila cypha*), southwestern willow flycatcher (*Empidonax traillii extimus*), yellow-billed cuckoo (*Coccyzus americanus occidentalis*), Yuma clapper rail (*Rallus longirostris yumanensis*), and desert tortoise (*Gopherus agassizii*). Along 650 river kilometers from Lake Mead (Nevada/Arizona) to the Mexican border, the program goals were to create at least 3,291 ha of new habitat including 2,404 ha of riparian (cottonwood-willow) habitat. The MSCP is a 50-year program with an estimated total cost of $626 million, of which a total of $197 million had already been spent by fiscal year 2014 [*Lower Colorado River Multi-Species Conservation Program*, 2015].

The MSCP approach can be viewed as a "form-based" restoration, which seeks to directly construct specific habitats, without restoring the riverine processes that formerly created and maintained such habitats. Many form-based approaches require ongoing maintenance to retain desired habitats that cannot be sustained by contemporary processes [*Choi*, 2004; *Choi et al.*, 2008; *Harris et al.*, 2006]. There is little evidence that habitat construction projects improve biotic condition [*Bernhardt and Palmer*, 2011; *Palmer et al.*, 2010]. Many of the MSCP projects have involved extensive plantings of willow (*Salix exigua, S. gooddingii*] and cottonwood [*Populus fremontii*). Some cottonwood-willow plantations are located on high terraces that no longer receive recharge from overbank flood flows, and where deep water tables have been further lowered by groundwater pumping. These plantations will require irrigation in perpetuity [*Nagler et al.*, 2007]. More significantly, it is not clear that these plantations can provide the structures and functions of a self-formed riparian ecosystem that are critical for sustaining target populations [*Nelson*, 2003]. Accordingly, the efficacy of the restoration strategy of the MSCP has been questioned for its emphasis on creation of habitat, rather than restoration of processes such as flow regime and channel dynamics, which could create self-sustaining riparian habitats [*Adler*, 2007; *Graham*, 2007]. Further, restoration plantings may not benefit species of interest unless all requirements (e.g. access to surface water, habitat heterogeneity, adequate available prey) are achieved.

To date, MSCP restoration sites have not successfully supported breeding by southwestern willow flycatchers [*Lower Colorado River Multi-Species Conservation Program*, 2013a]. It is unclear why sites are not being used. Colonization of the plantations may be limited by the small size of the remnant population of southwestern willow flycatcher and their site fidelity, meaning there are relatively few birds available to take advantage of the newly created habitat. Because some plantations are hundreds or thousands of meters from the river bank, the availability of insects on which the flycatchers depend is another potentially important factor. However, availability of insects at MSCP restoration sites had not previously been systematically measured and compared to other sites. Thus we sought to measure aquatic and terrestrial insects in the MSCP plantations in comparison to reference (representing best current ecological condition), control (unrestored), and nearby agricultural locations along the lower Colorado River.

Methods

Study Area

We selected three sites for comparison (Figure 1), which include two contrasting styles of restoration, Ahakhav Tribal Preserve and Cibola National Wildlife Refuge, and a reference ecosystem with successfully nesting southwestern willow flycatchers (Bill Williams River National Wildlife Refuge). The Ahakhav site is adjacent to a restored backwater channel of the Colorado River, with low-density plantings of willow, cottonwood, and mesquite. Ahakhav is located 4 km downstream of the point where the Colorado River emerges from a narrow bedrock gorge and enters the broad, flat, Palo Verde Valley. Air photos from 1938 show this area to be a wide and dynamic channel with little vegetation (Figure 2). Now, almost the entire Palo Verde Valley is planted with alfalfa, cotton, or other crops, and the river channel has been narrowed and simplified.

The Cibola National Wildlife Refuge (Figure 3) represents the large, densely planted, flood irrigated, plantation style of restoration typical of several MSCP restoration sites. The Cibola plantation sites that we sampled were more than 500m from the river's edge. Several MSCP sites are considerably farther from the river, including plantings at Cibola National Wildlife Refuge that are more than 2km from the river, and plantings more than 1km from the river at Palo Verde Ecological Reserve and at Cibola Valley Conservation Area.

The Bill Williams River (Figure 4) represents reference conditions: floods, although limited by upstream dams, still occur and facilitate recruitment of both native vegetation and *Tamarix*, and southwestern willow flycatcher have reliably nested there, suggesting insect densities are likely sufficient. We also sampled unrestored sites along the river bank near the Cibola plantations, a nearby actively managed alfalfa field at Cibola, and an uncultivated field at Ahakhav.

Insect Sampling and Analysis

At Ahakhav, Cibola and Bill Williams sites, we sampled insects in transects, collecting samples 0 m, 30 m, and 100 m from the river's edge. At Ahakhav, the 0-30-and 100 m samples were inside the restoration area, and we also sampled an abandoned agricultural field 620 m from the river along the Ahakhav transect as a 'control'. At Cibola we also collected samples in the MSCP willow-cottonwood plantation at 550 m and 590 m from the river bank, as well as an actively farmed alfalfa field 505 m from the river. In all, we established 13 sampling stations at the three sites (Figure 5, Table 1)

We sampled insects on three visits during the nesting season of the southwestern willow flycatcher: May, July, and September 2013. We used sticky traps to collect adult insects, as these are the food source for the flycatcher [*Durst et al.*, 2008]. We coated a non-toxic sticky resin (Tree Tanglefoot®) onto both sides of 8 ½ by 11-inch biodegradable acetate sheets [as in *Encalada and Peckarsky*, 2007]. We set the traps at each of our study sites for exactly 48 hours so that two full daily cycles would be sampled, [*Pehrsson*, 1984; *Wyman*, 1998]. The traps, clipped by clothespins onto nylon rope, were oriented perpendicular to the river. We set two rows of traps at each location, each with four acetate sheets. The rows were set at 1m and 2m above the ground (Figure 6). When collected we removed each insect from the sheets using Goo Gone® as a solvent and placed them into vials of 75% ethanol solution.

We taxonomically identified each insect to the extent required to differentiate aquatic from terrestrially originated insects. In many cases, the identification process was challenging because some individuals were damaged during the trapping or when being removed from the sticky traps. We identified all aquatic insects (e.g. Ephemeroptera and Trichoptera) to family, genus or species level. However, we did not further classify insects of the order Diptera, which can be either aquatic or terrestrial, because the effort of classification was beyond the scope of our study. We identified some aquatic Diptera (e.g. Chironomidae, Phoridae, Psychodidae) to family or genus level and were able to differentiate aquatic from terrestrial taxa in those cases. If an insect was not identified as aquatic or terrestrial we still included it in the counts of total insects and richness, but did not include them in our count of aquatic insects and were left out of our calculation of percent aquatic. At each of the 13 sampling stations, we surveyed topography (using an automatic level, rod, and tape) and classified vegetation along cross sections by measuring canopy cover with a spherical densitometer [*Lemmon*, 1956], and each plant was identified in 10m diameter plots around each sampling location (Table 1).

We tested hypotheses that total abundance of insects, aquatic insect abundance, percentage of aquatic insects, and ordinal richness all decrease with distance from the river. First we used linear regression, without thresholds of statistical significance to understand overall spatial trends. Then, to compare insects sampled at the river's edge (0 m sites) with those collected more than 500 meters from the river in Cibola MSCP plantation sites, we conducted a repeated measures ANOVA (R version 3.2.0). We transformed data when necessary to meet requirements of normality. Nineteen (6% of the 312 deployed) acetate sheets were lost due to wind. No more than 4 of the 8 sheets were ever lost at a site. We used the average number of insects per square meter of acetate sheet at each site rather than an absolute count of insects at each site. We measured richness by counting the number of orders present at each sampling location.

A large monsoonal rainstorm occurred during our September 2013 sampling visit; ~2.5 cm of rain fell at Cibola and ~5cm fell at the Bill Williams River. That single day's rain was equivalent to roughly 25% of the total precipitation that fell over the year 2013. In response, two species of terrestrial insects showed large population increases: Homoptera: Aleyrodidae and cf. Hemiptera: Diaspididae. We excluded these insects from our analyses because both were too small (<1 mm) to be suitable flycatcher prey. Also, we expect they were present only in response to the rare rain event, and not representative of conditions more generally.

We tested four hypotheses about insect abundance that inform suitability of plantations as habitat for the southwestern willow flycatcher. We hypothesized that aquatic insect abundance (Hypothesis 1) and the percentage of insects of aquatic origin (Hypothesis 2) would decline with distance from their sites of emergence in the river. Specifically, we expected the two Cibola plantation sites (C–Plant 550, C-Plant 590) to have fewer aquatic insects, and a smaller % of aquatics in the total sample, than the three sites at the river's edge (Ahakhav-0, Bill Williams-0, and Cibola-0). To formally test these hypotheses we used a repeated measures ANOVA to test for differences in the number of aquatic insects at plantation sites vs. river's edge sites. Because aquatic insects are known to be important components of invertebrate biomass in desert ecosystems, and because terrestrial invertebrates may track water resources [*Bastow et al.*, 2002; *McCluney and Sabo*, 2009], we expected that the total abundance of insects would decrease with distance from the river (Hypothesis 3). Specifically, we expected fewer insects at the two Cibola plantation sites than the three sites at the river's edge (Ahakhav-0, Bill Williams-0, and Cibola-0). To formally test this hypothesis we used a repeated measures ANOVA to test for differences in the total number of insects at plantation sites vs. river's edge sites. Finally, because riparian areas adjacent to the river were expected to have both aquatic and terrestrial insects ,as well as more individuals, we hypothesized that richness at the ordinal level would decrease with distance from the river (Hypothesis 4). Specifically, we expected the two Cibola plantation sites to have less insect diversity than the three sites at the river's edge (Ahakhav-0, Bill Williams-0, and Cibola-0). To formally test this hypothesis we used a repeated measures ANOVA to test for differences in the number of insect orders present at plantation sites vs. river's edge sites.

Results

Aquatic insect abundance declined with distance from the river (Figure 7a). The number of aquatic insects decreased from an average of 381 per m^2 of sticky trap at river's edge sites to only 12 at sites 100 m from the river, and 11 in the MSCP Cibola plantation sites (a decrease of 97% from the river's edge sites). Fluxes of aquatic insects were smaller in the MSCP plantation locations (Cibola Plant 550 and Cibola Plant 590) than at the river's edge sites (Ahakhav-0, Bill Williams-0, and Cibola-0) (p= 0.03; repeated measures ANOVA on log-transformed data, Figure 8a). There was not a consistent trend in aquatic insects over the summer, although at 9 of the 13 sampling locations, we trapped the fewest aquatic insects during the September sampling. At the Bill Williams River aquatic insects decreased over the summer. At Cibola and Ahakhav aquatic insect abundance peaked in the July sampling. While our study was not designed to test differences between aquatic production between Ahakhav, Bill Williams, and Cibola sites, it is noteworthy that at the leveed and simplified channel of Cibola-0 we trapped even more aquatic insects (average of 901 per m^2 of sticky trap) than at either the reference Bill Williams-0 (191 per m^2) or the restored side-channel of Ahakhav-0 (50 per m^2).

The percentage of insects of aquatic origin also decreased, as expected, with distance from the river, from an average of 59% at river's edge sites to only 12% at sites 100 m from the river, and 16% in the MSCP Cibola plantation sites (Figure 7b).

The percentage of aquatic insects was lower at the MSCP plantation locations (Cibola-Plantation 10 and Cibola-Plantation 50) than at the river's edge sites (Ahakhav-0, Bill Williams-0, and Cibola-0) ($p = 0.063$; repeated measures ANOVA on arcsine transformed data, Figure 8b). There was no trend in the proportion of aquatic insects trapped over the sampling period from May to September.

The total abundance of insects decreased with distance from the river (Figure 7c), dropping from an average of 473 insects per m^2 of sticky trap at river's edge sites to only 90 at sites 100 m from the river, and 68 (a decrease of 86% from the river's edge sites) in the MSCP Cibola plantation sites ($p = 0.055$; repeated measures ANOVA, Figure 8c). We found a generally decreasing trend in insect abundance over the sampling period from May through September, with 11 of the 13 sites trapping fewer insects in September than in May.

The ordinal richness also decreased with distance from the river (Figure 7d), from an average of 10 at river's edge sites to 7 at sites 100 m from the river, and 5 in the MSCP Cibola plantation sites ($p = 0.015$; repeated measures ANOVA on log-transformed data, Figure 8d). Aquatic orders and families are only present regularly in the first 30-100 m from the river shore. Farther than 100 m aquatic insects appear sporadically and are mainly tiny Chironomidae that can be easily transported by the wind. Interestingly, at Cibola, there is a high density of terrestrial insects and high terrestrial ordinal richness found in the alfalfa field, suggesting that active agriculture may increase terrestrial insect production.

Discussion

Our study demonstrates low availability of aquatic insects, total insects, and ordinal richness at restoration sites that are distant from the Colorado River. Because insect densities in distant restoration sites are only ~10% of densities along the Colorado and Bill Williams Rivers, our results call into question the quality and overall suitability of MSCP cottonwood-willow plantations as habitat for endangered flycatchers. We found support for all four hypotheses. The number of aquatic insects, proportion of aquatic insects, total number of insects, and ordinal richness all decreased with distance from the river, and the decrease occurred within the first 100 meters, with much of the decrease occurring in the first 30 meters (Figure 7). The MSCP cottonwood-willow plantation (more than 500 m from the river) at Cibola National Wildlife Refuge had fewer aquatic insects, a lower proportion of aquatic insects, fewer total insects, and fewer insect orders than sites adjacent to the river (Figure 8).

Uncertainty

Two primary sources of uncertainty exist in this analysis. The first is that sticky traps were truly representative of the ambient insect abundance. Sticky traps are non-attracting, but traps placed in open areas may receive wind-blown insects that may not accurately represent prey availability. More research on flycatcher feeding habits would help address this uncertainty. Second, with only three study sites (two restoration sites and one reference site) any generalization to other restoration sites should be done with caution. It is possible that with other soil types or irrigation regimes some MSCP restoration sites could be producing more insects than the sites we sampled at Cibola. A broader investigation of sites along the lower Colorado River could address this question. Future research could forego taxonomic identification and a large number of sites could be sampled and analyzed relatively quickly.

Highest Insect Production Along River Banks

Insect availability, both total and aquatic, was highest at riverbank sites, and declined by approximately an order of magnitude within the first 100 meters from the river's edge. This decrease is consistent, though at the higher end, with results from a meta-analysis of aquatic-to-terrestrial insect subsidy studies on 109 streams which found that insect dispersal decreased 90% an average of 330m from the river's edge and predator abundance decreased 90% at 570m from the river's edge [*Muehlbauer et al.*, 2014]. Despite the Colorado River being straightened, leveed, and simplified, the river is still clearly the most productive habitat for insects. In fact, the riverbank near Cibola produced more total insects, more aquatic insects, and nearly as many insect orders as the Bill Williams River reference site (no statistical tests performed). The greater production at Cibola may be attributed to a wider channel, with more capacity to produce aquatic insects, than at Bill Williams. Alternately, the production at Cibola may be explained by what is likely better-oxygenated water in the flowing mainstem Colorado than in the more stagnant backwaters at Ahakhav and in the Bill Williams River reference site.

The plantations distant from the river appear to provide few insects for insectivore birds. MSCP has experimented with fertilizing cottonwood and willow plantations, and found no significant increase in the overall abundance of Arthropoda [*Lower Colorado River Multi-Species Conservation Program*, 2010] and no method to effectively increase insect production in plantations. Although there are many differences between the Ahahkav, Bill Williams, and Cibola sites that might influence aquatic and terrestrial insect production (e.g. water quality, channel morphology, substrate, flow depth and velocity, soil type, land use, etc.), the consistent trend between sites suggests that such variability is less significant than distance from the river. Future research should investigate variability in aquatic and terrestrial insect production along the lower Colorado River, and determine the threshold prey density that constitutes a stress for the southwestern willow flycatcher. Our results confirm prior studies documenting decreasing insect densities with distance from the river, and highlight the importance of considering prey availability as a possible limiting factor in restoration sites.

Attempts to Create Artificial Water Bodies Adjacent to Vegetation

The Habitat Conservation Plan guiding the MSCP acknowledged the importance of nesting habitat near water for aquatic insect production as prey for southwestern willow flycatcher, stating, "Created cottonwood-willow designed to provide southwestern willow flycatcher habitat will be specifically managed to ensure that moist surface soil, slow-moving water, or ponded water conditions are present during the breeding season to ensure the production of the flycatcher's flying insect prey base…Designs of created habitats will emphasize creation of nesting habitat within 200 feet of standing or slow-moving water or moist surface soils" [*Lower Colorado River Multi-Species Conservation Program*, 2004 p. 5-13, 5-38]. Looking at avian communities in general, Hinojosa-Huerta [2008] found that the presence of surface water was the greatest habitat predictor of avian density and richness along the lower Colorado River in Mexico.

In recognition of the importance of adjacent water bodies for southwestern willow flycatcher habitat, the MSCP spent more than $450,000 on lab, nursery, and field studies between 2010 and 2014 investigating the feasibility of using soil amendments to pond water and increase soil moisture in sites distant from the river with sandy soils and with deep water tables [*Lower Colorado River Multi-Species Conservation Program*, 2015]. However, it remains uncertain whether juxtaposition of artificial water features with willow and cottonwood plantations can achieve the same habitat benefits as a riparian forest next to the river. Although it is possible to construct artificial water features in the riparian plantations, it is unclear if the complex set of processes and habitat attributes associated with water can be effectively replicated.

Moreover, Glenn [2008] found the less-managed and more heterogeneous US-Mexico border (limitrophe) reach of the Colorado River had higher habitat complexity and supported richer avian populations than the more canalized reaches upstream. Similarly, in a longitudinal study of sites along the Colorado River in Grand Canyon, Cross [2013] demonstrated that sites below tributary junctions had food webs more supportive of native species, which can be attributed to more natural flow and sediment regimes, more complex geomorphology and intact populations of native biota that are present in the tributaries. One implication of these studies is that restoration potential varies across the landscape, so regional restoration planning on the lower Colorado could likely benefit from a system-wide analysis of insect communities and habitat potential along the Colorado River and tributaries.

Restoration Potential Adjacent to the River

Our findings show that riparian restoration sites that are over 100m from the river have only ~4% of the aquatic insects and ~20% of the total insects than sites adjacent to the river. Thus, the overall habitat quality of restoration sites that are distant from the river is likely considerably lower for insectivorous birds such as southwestern willow flycatcher. Despite a history of degradation from channel straightening, incision, and reduced water quality, the river channel itself is the habitat most productive of insects needed by the target bird species, suggesting that restoration projects along the river bank may have greater potential to support the birds. Conditions along the riverbank are limited by the highly regulated flow regime, and would likely benefit from increased dynamism, such as periodic high flows, which have successfully driven ecological improvements in other rivers [*Cross et al.*, 2011; *Rood et al.*, 2005]. The last high flows in the lower Colorado River occurred in the El Niño year 1982-1983. These flows re-established native vegetation and improved riparian and wetland ecosystems along the lower Colorado River and in the Colorado River delta [*Luecke et al.*, 1999; *Zamora-Arroyo et al.*, 2005]. However, such wet-year reservoir releases are increasingly unlikely in light of increased water demand, which keeps reservoir levels low. Also, without high flows in recent decades, new developments have been constructed adjacent to the river in some reaches, adding a political and economic obstacle to dam releases that might flood these buildings. The experimental pulse flow of 2014, which transported water in the lower Colorado River into Mexico (under the Minute 319 amendment to the 1944 treaty between the United States and Mexico regulating the Colorado River) was much smaller than the 1982-1983 flows, though it exemplifies a promising strategy to restore ecosystem processes in the lower Colorado [*Buono and Eckstein*, 2014]. Similarly, the integrated dam releases and vegetation management on the Bill Williams River demonstrate the possibilities that are available on the tributaries of the lower Colorado [*Shafroth and Beauchamp*, 2006].

Why has MSCP not located restoration projects along the banks of the lower Colorado River?

In our discussions, MSCP restoration planners have cited three factors. First, because of the irregular margin along the river, the available parcels tend to be small and irregular in shape, making them difficult to vegetate and irrigate using the highly mechanized approach used on the large plantations. Thus, the river-edge sites would need to be planted mostly by hand labor, resulting in a higher cost per unit area. Also there are dredge spoils along parts of the river bank, which may limit plant establishment. Second, trees planted along the river risk being eroded by the river in the (now unlikely) event of a large flow. If they were eroded, plantings may not count as credit towards the area requirements of the Habitat Conservation Plan. Third, riparian trees planted along the river banks would presumably draw upon shallow alluvial groundwater recharged by the river (at least once the tree roots extend deeply enough to reach the water table). This could create complications with water rights, as essentially all the water flowing down the Colorado River in this reach is claimed by some party and destined for diversion downstream. However, we are not aware of any systematic, comprehensive analysis of the potential ecological benefits of restoring habitat along the river and possible ways to address the factors that have to date prevented restoration adjacent to the river.

Maintaining Young Vegetation Without Floods?

As noted above, historical riparian habitats included large areas of pioneer and relatively young vegetation, the result of reworking by periodic floods. Not surprisingly, native birds are adapted to the habitat provided by these young trees. The MSCP Palo Verde Ecological Reserve hosts a growing population of yellow-billed cuckoo (*Coccyzus americanus occidentalis*), which nest almost exclusively in willow and cottonwood trees that are 2-3 years old [*Lower Colorado River Multi-Species Conservation Program*, 2013b]. Presumably the young, densely planted trees at the Palo Verde site mimic the kind of dense, young stands of trees that formerly existed along the Colorado River, and thus provide preferred conditions (such as cover) for the cuckoos.

As these plantation trees mature, however, their foliage will become limited to the upper meter or two of the canopy (Figure 9), and likely would not provide the cover from predators afforded by the young trees. A MSCP study tested the effects of patch size and canopy cover on nest predation by placing artificial nests in different types of vegetation, but the study did not investigate the MSCP plantations or how cover in the plantations evolves over time [*Theimer and Pellegrini*, 2011]. MSCP habitat is intended to provide habitat for 50 years, but it is unclear how the target habitat can be feasibly maintained in the absence of the dynamic river disturbance-succession processes of erosion, deposition, and vegetation recruitment. In theory, the MSCP could remove older trees and replant on a frequent, staggered schedule so that at any given time, stands of 2-3 year-old trees would exist. However, we have not encountered any such long-term management plan, nor any analysis of the feasibility and costs of such perpetual maintenance.

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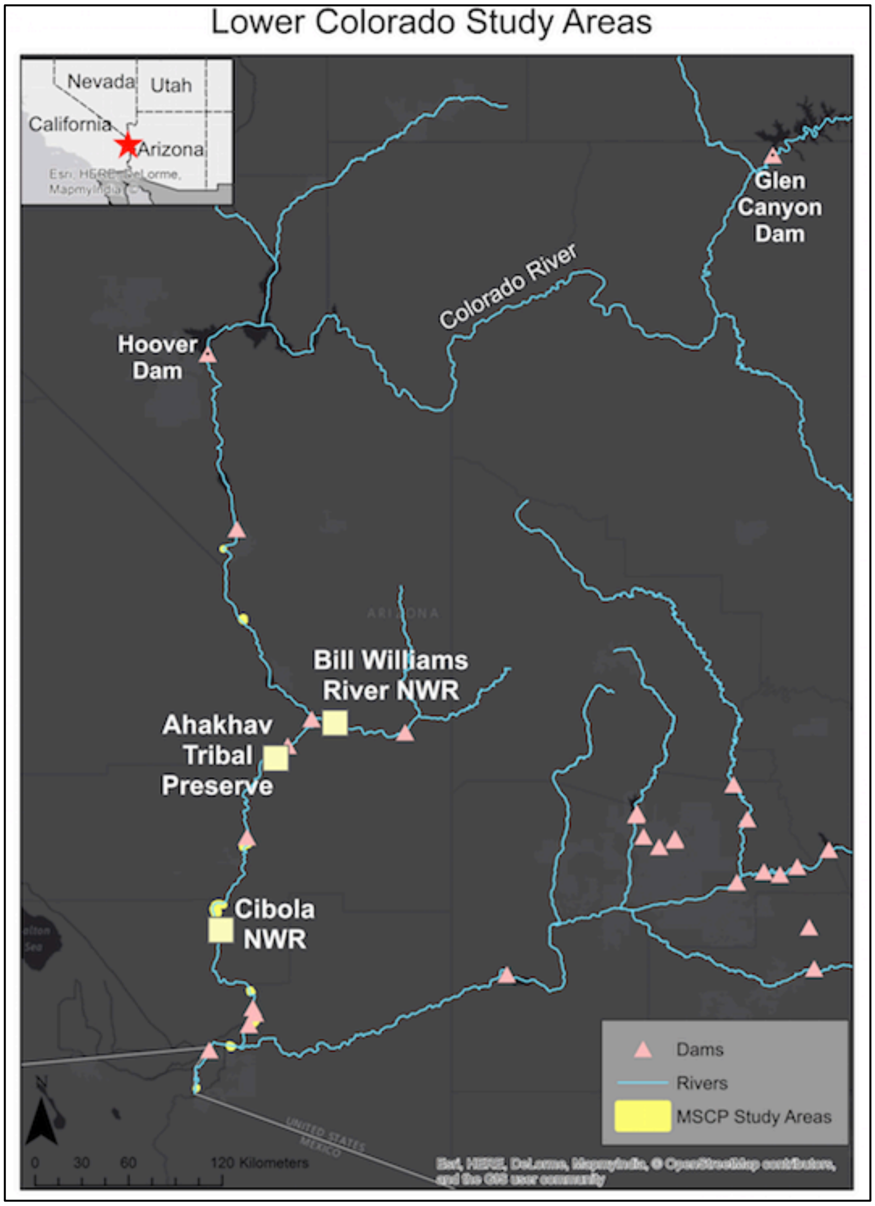
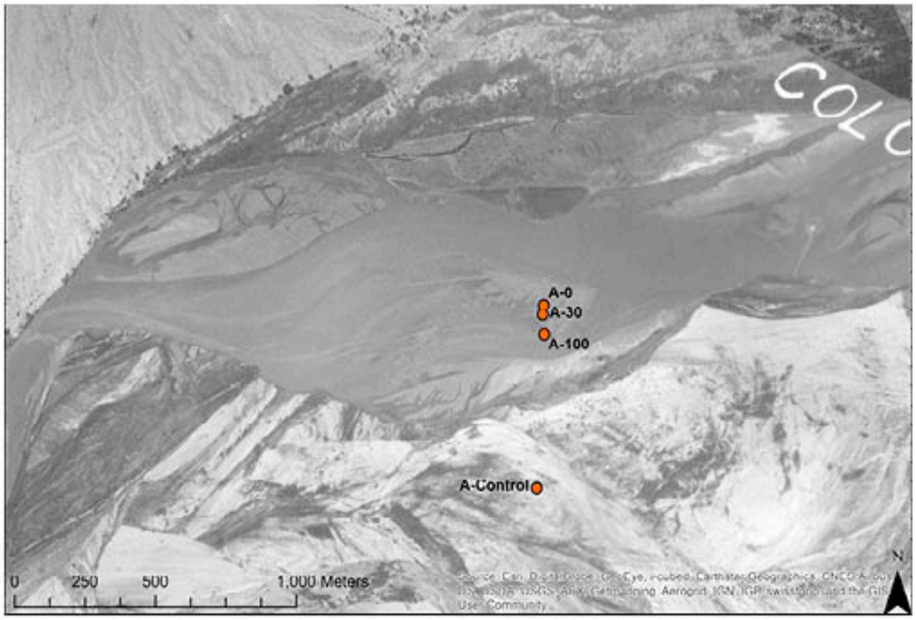


Figure 1. The MSCP program area extends along the Colorado River from Lake Mead to the Mexican border. We sampled at the Ahakhav Tribal Preserve (a non-MSCP restoration site), the Bill Williams River National Wildlife Refuge (a river formed riparian area), and the Cibola National Wildlife Refuge (a MSCP cottonwood-willow plantation).

A)
1938



B)
2013

Figure 2. The wide (~1000 m wide) and dynamic channel of 1938 (a) had greatly simplified and narrowed (~200 m wide) by 2013 (b), though it was still complex relative to other reaches of the lower Colorado (a side channel is present) including Cibola (Figure 3). The control site was a fallow alfalfa farm that hadn't been cultivated for several years. Due to restoration activities and the elimination of scouring flows, substantially more vegetation was present in 2013 than in 1938. Sources: 1938 [*Norman et al.*, 2006]; 2013 [http://goto.arcgisonline.com/maps/World_Imagery]

A)
1938

B)
2013

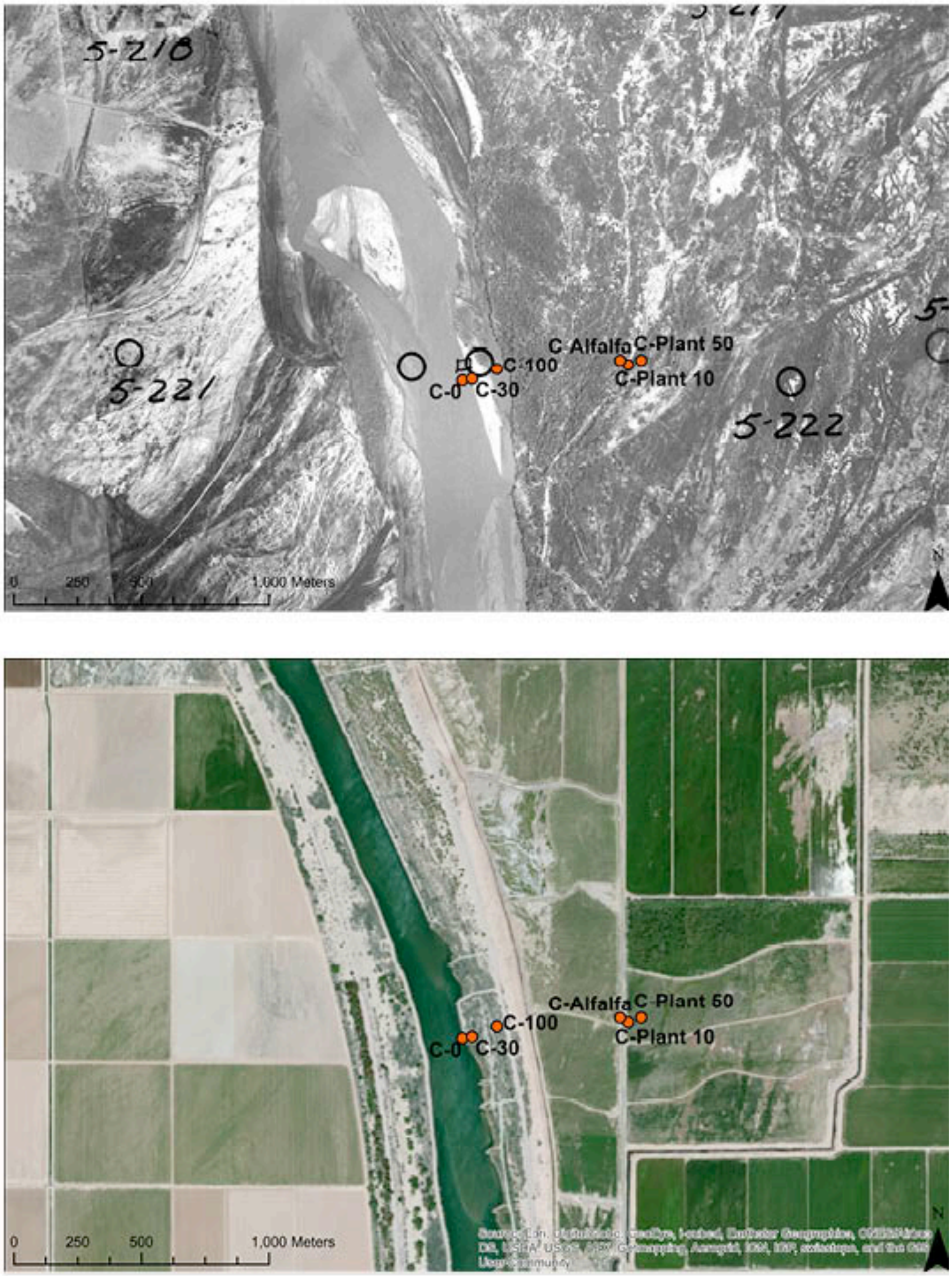


Figure 3. In 1938 (a) the Colorado River at Cibola was ~500 m wide including a large mid-channel bar, and large patches of both vegetated and unvegetated portions of the floodplain. The channel in 2013 (b) at Cibola was fixed in place with "J dykes" and a levee (just east of point C-100) and narrowed to ~100 m wide. The alfalfa field and MSCP plantation sites were disconnected from the river by the levee, though an irrigation canal was present between the alfalfa field and plantation. Sources: 1938 [*Norman et al.*, 2006]; 2013 [http://goto.arcgisonline.com/maps/World_Imagery]

A)
1947

B)
2013

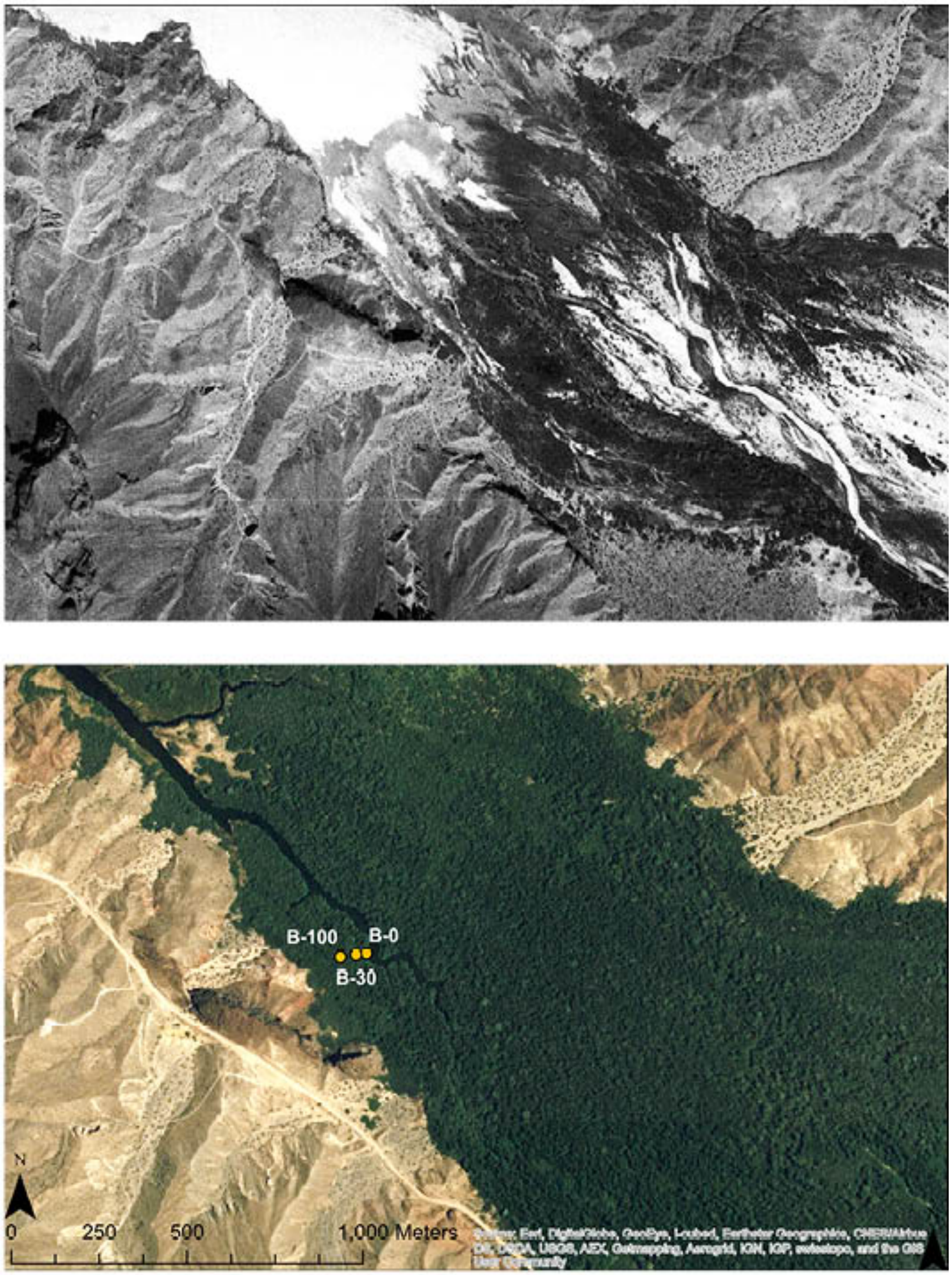


Figure 4. The lower Bill Williams River in the Bill Williams River National Wildlife Refuge is an extremely dense forest of willow, cottonwood and tamarisk, as visible in 2013 (b). Oblique imagery from 1947 shows the same area as an actively prograding delta as the Bill Williams River flows (left) into the backwater of the Colorado River at Lake Havasu. Lake Havasu was formed in 1938 when Parker Dam was completed. Thus, even though the flow of the Bill Williams River is regulated by Alamo Dam upstream, the downstream Bill Williams maintains an active channel and has had several episodes of vegetation recruitment and erosion over the past eight decades. Sources: 1947 [ARB000384820057 from USGS Earth Explorer]; 2013[http://goto.arcgisonline.com/maps/World_Imagery].

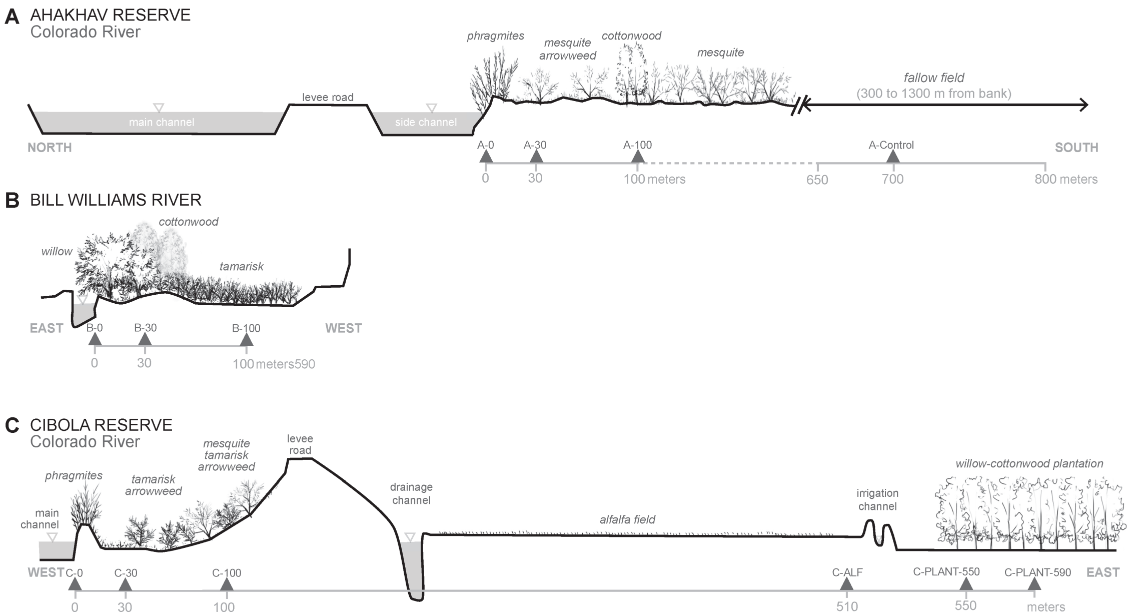


Figure 5- Cross sections (looking upstream) of A) Ahakhav Tribal Preserve restoration site implemented by the Colorado River Indian Tribes with constructed side channel, low-density riparian plantings, and a fallow alfalfa field which was sampled as a control site, B) the Bill Williams River in the Bill Williams River National Wildlife Refuge which is a river-formed riparian area with a mix of cottonwood, willow, and tamarisk vegetation and one of the few sites in the region with consistently nesting southwestern willow flycatcher, and C) the MSCP plantation sites in the Cibola National Wildlife Refuge.

Figure 6. Example of sticky trap setup at Ahakhav Preserve. Each setup included four acetate sheets at 1m above ground and four sheets 2m above ground level. Traps were oriented perpendicular to the orientation of the river.

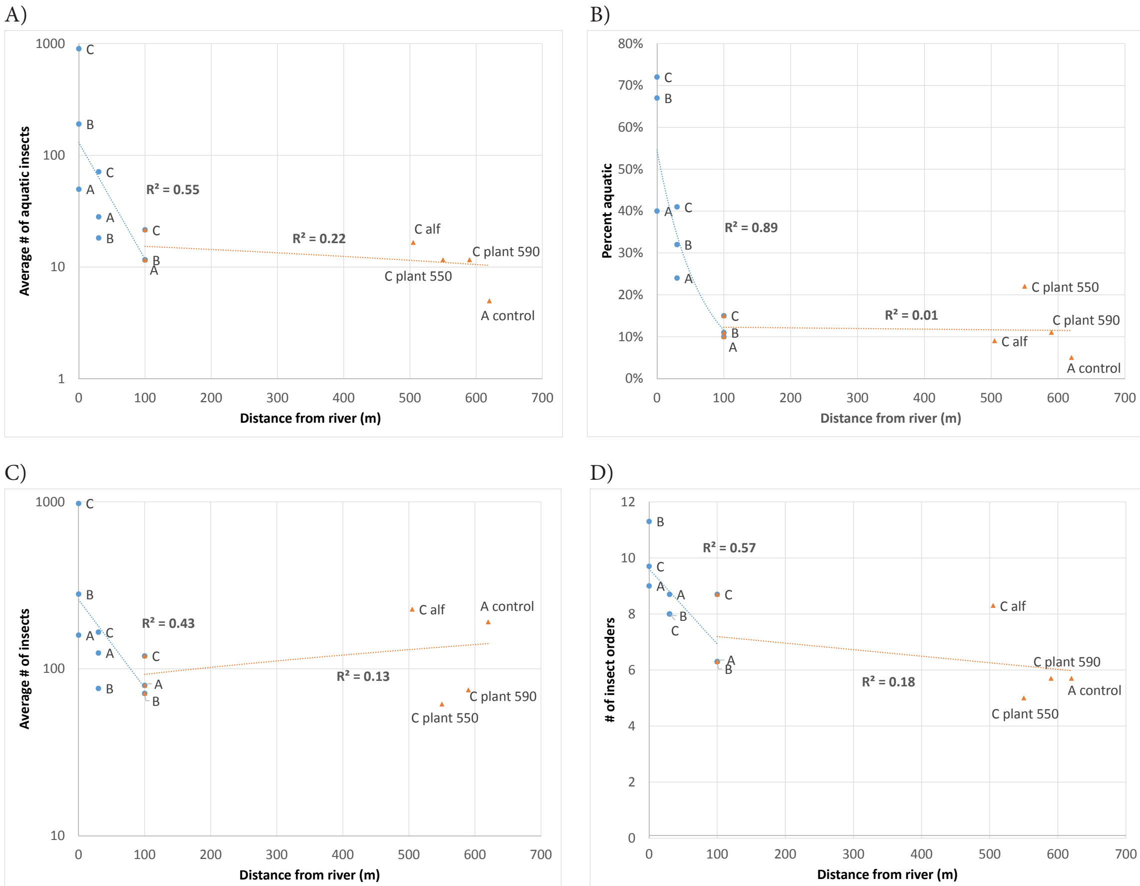
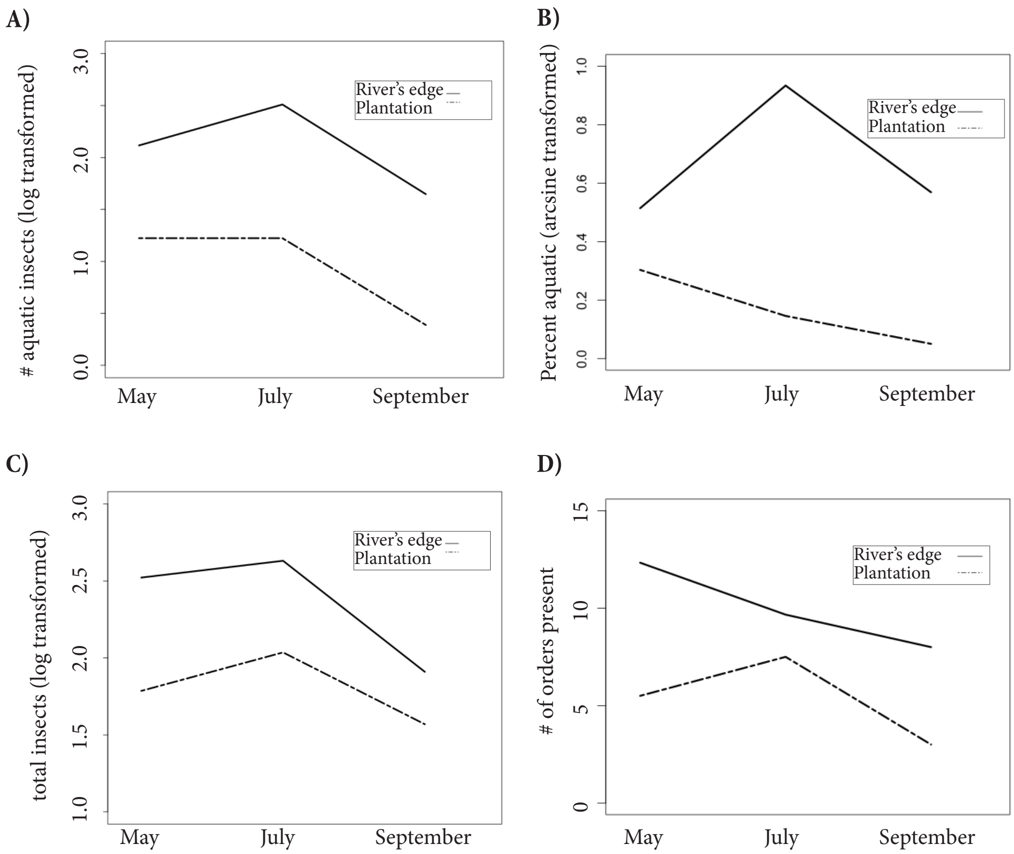


Figure 7. Sites are designated as Ahakhav (A), Bill Williams River (B), and Cibola (C). Each point represents the average of three sampling periods, (May, July, and September). Upper Left) The average number of aquatic insects per square meter decreased from 381 at river's edge sites to only 15 at sites 100 m from the river, and 12 in the Cibola plantation sites (over 500 m from the river). Upper right) 59% of insects were aquatic in origin at the river's edge sites, but aquatic insects comprised only 12% of insects at 100 m and 16% in the plantation sites. Lower left) The mean number of total insects averaged 473 per square meter at the river's edge sites, but only 90 insects at 100 m and 68 in the plantation sites. Lower right) At the river's edge sites, samples included an average of 10 orders of insects; 100m from the river samples included 7 orders, and an average of 5 orders were trapped in the plantation sites.



	Untransformed mean (river)	Untransformed mean (plantation)	p value (groups)
# aquatic insects	381	12	0.032**
% aquatic insects	59.4%	16.3%	0.063*
# of insects	473	68	0.055*
# of orders	10.0	5.3	0.015**

Figure 8. The average number of aquatic insects (upper left), the percentage of aquatic insects (upper right), the average number of total insects (lower left), and the number of orders trapped at each site (lower right), were all lower at the plantation than at the river's edge sites. The number of aquatic insects and the number of orders were significant at the 0.05 level (**) all tests were significant at the 0.10 level (*).

Figure 9. Whereas the historical riparian vegetation existed in short-lived patches between episodes of disturbance, contemporary vegetation planted by the MSCP is expected to be long lived. Although not systematically evaluated in our study, MSCP plantations such as Cibola NWR, Palo Verde Ecological Reserve (shown above), and Cibola Valley Conservation Area are typically planted in dense rows and have little understory foliage. By contrast, the vegetation of the lower Bill Williams River at the delta in Lake Havasu (below) is extremely dense, with a more complex structure including an overstory of cottonwood and willow, with an understory of dominated by tamarisk.

Table 1. Site descriptions

					Vegetation Species Count										
SITE	Station	Distance from River (m)	Type	*% Canopy Cover*	Alfalfa	Herbaceous Perennial	Pluchea sericea	Tule	Phragmites australis	Tamarix ramosissima	Prosopis glandulosa	Salix spp.	Populus fremontii	Unknown	Total Plants per 10 m circle
Ahakhav	A-0	0	Restored	*54*	0	0	15	0	280	1	1	0	0	0	594
	A-30	30	Restored	*5*	0	0	72	0	0	0	1	0	0	0	73
	A-100	100	Restored	*14*	0	0	100	0	0	0	2	0	1	0	103
	A-CTRL	620	Fallow ag	*0*	0	0	0	0	0	0	0	0	0	0	0
Bill Williams	B-0	0	Reference	*80*	0	110	0	0	0	1	0	3	0	0	228
	B-30	30	Reference	*68*	0	0	0	0	0	5	0	5	0	0	10
	B-100	100	Reference	*97*	0	0	0	0	0	9	0	0	0	0	9
Cibola	C-0	0	Unrestored	*7*	0	0	36	50	40	3	0	0	0	0	258
	C-30	30	Unrestored	*0*	0	0	200	0	0	3	0	0	0	0	203
	C-100	100	Unrestored	*0*	0	0	75	0	0	8	4	0	0	5	87
	C-ALF	505	Ag	*0*	100s	0	0	0	0	0	0	0	0	0	100s
	C-PLANT 550	550	Restored	*93*	0	0	0	0	0	1	0	280	4	0	285
	C-PLANT 590	590	Restored	*82*	0	0	0	0	0	8	0	250	7	0	265

Conclusion

In Chapter 1 I used sediment reduction as an indicator of the downstream impact of dams in the Mekong basin. However, in contrast to the clear basis for using sediment reduction as an indicator of the impacts of dams, the theoretical and practical considerations for quantifying restoration are much more complex. One fundamental problem is that the metric that is used to quantify success can often be achieved through many pathways. In practice, this problem of equifinality means that one might “restore” the sediment load of the Mekong River in several ways and not all ways will be functionally equivalent. Essentially, while the question of impacts could be assessed with the simple metric of total annual sediment load, quantifying restoration of sediment may require consideration of the size, timing, mobility, sorting, and chemistry, and habitat suitability of that sediment, a much more difficult problem.

Many restoration projects are implemented to benefit specific species, yet because populations may take decades or longer to respond to restoration activities, and because populations of target species may be strongly affected by factors unrelated to restoration actions, measures of populations are often not appropriate ways to promptly evaluate restoration. Therefore, many monitoring programs use surrogate, and potentially irrelevant, metrics to evaluate restoration project performance. In Chapter 2 we critiqued the use of such metrics for restoration evaluation, and question the meaningfulness of universal metrics to quantify ecosystem conditions. To protect regional and global biodiversity we must maintain a diversity of stream types, not simply engineering our preferred type. To date it is not yet possible to draw general conclusions on whether habitat heterogeneity projects are succeeding. Evaluations need more rigor and connection to project specific goals, rather than relying on generic metrics such as macroinvertebrate diversity and richness. More fundamentally, all studies reviewed in Chapter 2 used macroinvertebrate diversity and/or richness as the measure of ecological success, though the meaningfulness of reach-scale diversity/richness as an indicator of ecosystem condition is not clear. Monitoring and evaluation should first establish hypotheses and conceptual models based on watershed perturbations and set specific milestones towards a sustainable, dynamic, and healthy ecosystem.

In Chapter 3 we demonstrated a meaningful and feasible evaluation approach relying upon prey availability as an indicator. The lower Colorado River Multi Species Conservation Program established willow-cottonwood plantations to provide habitat for threatened and endangered insectivores such as the southwestern willow flycatcher (*Empidonax traillii extimus*). Therefore, insect (prey) availability has potential as a useful measure of habitat function. Riparian restoration sites have been planted more than 2 km from the river and sustained through irrigation. We used sampled insect communities in restored, control, and reference sites along the lower Colorado and Bill Williams Rivers in Arizona. Sites farther than 100m from the river's edge had: 1) fewer insects, 2) fewer aquatic insects, 3) a lower percentage of aquatic insects than sites along the river's edge, and 4) less ordinal richness. Results suggest that unless habitat construction projects consider physical and biological processes and context, essential habitat functions may not be achieved.

The selection of metrics for quantifying ecosystem impacts is still a complex endeavor, but in many cases meaningful metrics (e.g. alteration of sediment and flow regime) can be applied across many different systems and the same metrics are appropriate for a variety of causes. By contrast, the mechanics of ecosystem recovery and the appropriate indicators for quantifying restoration success are far from clear. With more than a billion dollars spent per year on river restoration [*Bernhardt et al.*, 2005] there is a growing urgency to quantify benefits of restoration and prioritize strategies. As an example, in an effort to ensure taxpayer resources are allocated responsibly, the US Army Corps of Engineers is working with the Office of Management and Budget to quantify restoration effectiveness by evaluating acres restored, initial costs, and ongoing maintenance costs [*U.S. Army Corps of Engineers*, 2012]. This of dollars per acre is an important step in addressing restoration effectiveness, though minimizes the importance of project performance. However, even more detailed studies of habitat and biotic change may not yield meaningful results unless connected to clear hypotheses about the system. Recent advancements in techniques such as remote sensing and data processing have allowed researchers to quantify the world in ways that were unimaginable only a few years ago, many questions still persist regarding what are meaningful indicators and how to interpret observations to effectively manage watersheds. The future of restoration evaluation is summarized nicely by John Tukey "Far better an approximate answer to the right question, which is often vague, than an exact answer to the wrong question, which can always be made precise" [*Tukey*, 1962].

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