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Functional integrity of freshwater forested wetlands, hydrologic alteration, and climate change

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Abstract. Climate change will challenge managers to balance the freshwater needs of humans and wetlands. The Intergovernmental Panel on Climate Change predicts that most regions of the world will be exposed to higher temperatures, CO₂, and more erratic precipitation, with some regions likely to have alternating episodes of intense flooding and mega-drought. Coastal areas will be exposed to more frequent saltwater inundation as sea levels rise. Local land managers desperately need intra-regional climate information for site-specific planning, management, and restoration activities. Managers will be challenged to deliver freshwater to floodplains during climate change-induced drought, particularly within hydrologically altered and developed landscapes. Assessment of forest health, both by field and remote sensing techniques, will be essential to signal the need for hydrologic remediation. Studies of the utility of the release of freshwater to remediate stressed forested floodplains along the Murray and Mississippi Rivers suggest that brief episodes of freshwater remediation for trees can have positive health benefits for these forests. The challenges of climate change in forests of the developing world will be considered using the Tonle Sap of Cambodia as an example. With little ecological knowledge of the impacts, managing climate change will add to environmental problems already faced in the developing world with new river engineering projects. These emerging approaches to remediate stressed trees will be of utmost importance for managing worldwide floodplain forests with predicted climate changes.

Key words: *Asia; Australia; climate change; coastal wetlands; floodplain forest; India; North America; temperate; tropical*

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Freshwater for Humans and Wetlands

The world's climate is warming rapidly, and associated changes in evaporation, precipitation, and freshwater use may cause future episodes of devastating freshwater scarcity (IPCC 2014). Climate change will challenge our ability to balance the freshwater needs of humans and natural ecosystems (Naiman et al. 1995). Even if emissions are reduced greatly, because the greenhouse gas specter already has been released from stored carbon, emerging novel climates will persist for many decades (Hansen and Cramer 2015). We will need to re-examine our approaches to wetland conservation as this new reality unfolds.

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Future human demand for freshwater will intensify as temperature and evaporation increase (CSIRO and Australian Bureau of Meteorology 2007, IPCC 2014), which will likely result in less freshwater for wetland management (Meyer et al. 1999). A conservation conflict may arise if water shortages emerge because presently, a large percentage of humanity does not have enough freshwater. Eventually water demand could overshadow loss due to global warming, especially in developing countries (Vörösmarty et al. 2000, IPCC 2007). In South Africa, excess freshwater extraction already has created water shortages for human consumption (Whitfield and Wooldridge 1994, Schlacher and Wooldridge 1996) and the Millennium drought in southeastern Australia saw drastic reductions in the amount of water available for both irrigated agriculture and wetland conservation (van Dijk et al. 2013).

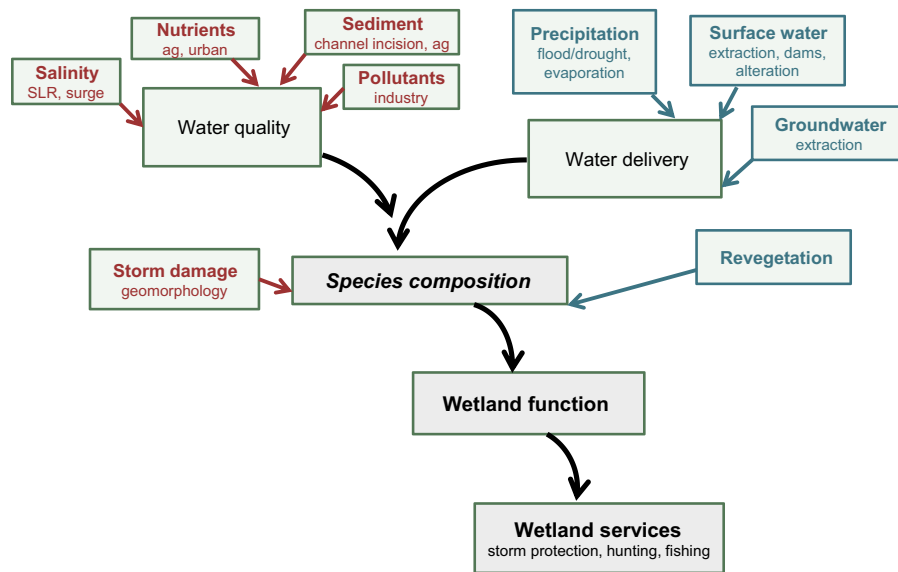


Fig. 1. Conceptual model of natural and anthropogenic driving forces, which influence water delivery and quality to freshwater ecosystems.

Land use by six billion people in recent decades has put a strain on the biosphere, particularly freshwater and forest resources (Foley et al. 2005). While we are a part of the Earth system (Falkenmark 2003) along with countless other organisms, humans act as a mindful forcing factor of wetland change.

Beyond water scarcity, future attempts to manage wetland environments will be hampered by damaged water delivery systems both in the developed and developing world (Fig. 1). Landscapes engineered for urban and agricultural development have modified flow pathways to natural ecosystems, so that water from precipitation, channels, water bodies, and groundwater aquifers is redirected. Changes in the passage of water through rivers, lakes, wetlands, and groundwa-

ter will have implications for wetland function, integrity, and structure (Fig. 1; Middleton 1999a,b, Baron et al. 2002).

While landscape alteration has affected water delivery systems to wetlands in similar ways worldwide, climate change is likely to have more regionally specific effects on ecosystem function (Jump et al. 2009). Based on projections of climate change related to levels of greenhouse gases in the atmosphere, future climate environments will vary in temperature, CO₂, storm frequency, and water availability (Intergovernmental Panel on Climate Change (IPCC) 2014), but intra-regional climate patterns may differ, especially with respect to flooding (Mallakpour and Villarini 2015; Fig. 2).

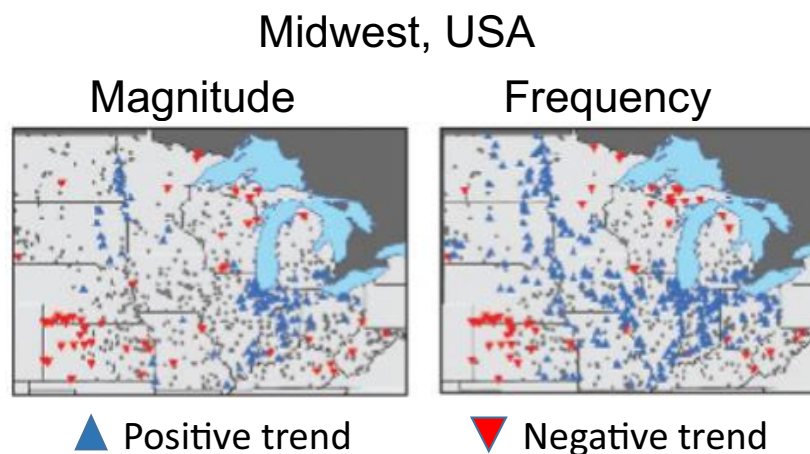


Fig. 2. Trends in the magnitude and frequency of annual flood events from 1962–2011 in the midwestern United States with maps showing trends in the magnitude (A), and frequency (B) of flood events. Triangles show gage station locations with increasing and decreasing flooding indicated by blue red triangles, respectively ($p < 0.05$). Gray circles indicate stations that did not change ($p > 0.05$) (from Mallakpour and Villarini 2015; with permission of Nature Geosciences).

In this review, we consider land-use changes that are likely to exacerbate climate-change stresses on floodplain ecosystems including water extraction, hydrologic alteration to wetlands, and human-induced alterations to natural landscapes. The objectives of this paper are to describe: climate and associated land-use changes affecting freshwater floodplain forests, vegetation health and assessment, and emerging technologies to remediate stressed vegetation. We will describe the predicted impact of global climate and land-use change on the world's major rivers. We focus on forested floodplain forests as highly productive and biodiverse ecosystems, which provide important ecosystem services to humans. Among the examples given to demonstrate potential management approaches during climate change include hydrologic remediation studies in mixed swamp forests of the Mississippi River Alluvial Valley of the United States, river red gum black box and coobah forests of the River Murray in South Australia, and freshwater forests of the Mekong River and its Tonle Sap Great Lake in Indochina. This review will bring together vital information to inform scientists, managers, and policy makers as they begin to frame their planning for future floodplain management.

Freshwater scarcity on already damaged floodplains will pose serious problems for managers of nature conservation areas. In the future, the U.S. National Park Service Advisory Board state that managers should prepare for conditions of "continuous change" because the fires, hurricanes, and droughts of future decades may be beyond our historical experiences (NPSAB 2012). If climates change as projected in the future, conservation managers will be tasked with an increasingly difficult job to protect species and ecosystems in freshwater floodplain forests.

Land Use, Ecosystem Alteration, and Climate Change

Both human population and demand for freshwater have doubled since the 1940s, so that extraction of water overshadows most other human impacts on wetlands (Naiman et al. 1995). While wetlands have resilience to natural disturbances, extreme changes in the water environment outside of the adaptive capabilities of the plant species can break the ability of natural ecosystems to rebound after disturbance (Middleton 1999a). Ecosystems can be impaired to the point that they no longer function in a way similar to undisturbed ecosystems (Carpenter et al. 1992, Baron et al. 2002). As humans demand more freshwater as climates become hotter, this imbalance of water allocation to humans and ecosystems will increase. Beyond the water itself, climate change impacts to wetlands will be amplified by associated changes in temperature, drought, windstorm, and saltwater intrusion (Naiman et al. 1995).

While water extraction for human and industrial use impairs water delivery systems to floodplain forests, other problems emerge for floodplains after changes in land-use including deforestation, soil tillage for agriculture (both irrigated and dryland), and re-engineering for flood protection. Landscape-level changes in the water delivery system affect the nature of the flood pulse delivered to the floodplain (Junk et al. 1989, Middleton 1999a, Tockner et al. 2000, Jolly et al. 2008), and any change in the seasonality, timing, quantity, and quality of water delivery can impair function (Baron et al. 2002). Virtually all rivers in the United States have been hydrologically altered (Dynesius and Nilsson 1994) with 90% of discharge from U.S. rivers affected by dams, reservoirs, water transfers, levees, channelization, straightening, and/or irrigation (Jackson et al. 2001). Damage to river systems in the developing world is catching up with a recent frenzy of dam planning and building. As development accelerates, new major dams are planned or under construction along world rivers including the Mekong River, Amazon, and Arunachal Pradesh in India (14, 60, and 135, respectively; International Rivers 2015).

In addition to landscape-level damage to water delivery systems, coastal wetlands face direct threats from sea-level rise, which will infuse salinity into fresh surface and groundwater (Schneider 1993). Accelerated sea-level rise is almost inevitable in the future because of the rapid melting of glacial and polar ice (IPCC 2014). Coastal flooding is already occurring in North America, with very high flooding unrelated to storm activity on the East Coast of the United States (McCoy 2015). Coastal cities with growing human populations are flooding because of the double whammy of sea-level rise and coastal subsidence (Kennish 2002).

Arguably, the majority of damage to wetlands to date may have had less to do with the direct impact of climate change than with modification of water delivery systems. Regardless of which of these factors is more of a threat to wetlands, the combined impact of climate change and landscape modification will create a myriad of novel problems for future wetland managers. As the unknown challenges of future environments unfold, we may find ourselves unprepared to protect wetlands both scientifically and legally. Laws to protect natural ecosystems are usually based on lowest acceptable water flow or quality standards, but the environments engendered by these laws may not adequately support either rare species or sustainable natural ecosystems as climates change (Baron et al. 2002).

Repairing Hydrology of Floodplain Forests and Climate Change

Fixing the water delivery system to floodplain forests could be of great value for managers as climates change, but the path to repair depends on how the delivery system was broken (Middleton 1999a), the physical

environment of the damaged floodplain, and the desired outcome of forest ecosystem composition and function (Bunn and Arthington 2002). After the water delivery system is damaged, a shift in vegetation composition can alter wetland functions such as decomposition, nutrient dynamics, energy flow, and carbon sequestration (Ellison et al. 2005). Also, species interactions, e.g., plant–herbivore and predator–prey, are likely to vary by habitat (McCluney et al. 2012) and climate change (Parmesan et al. 2005). Therefore, this review considers the specific effects of particular types of hydrologic alteration, and how these alterations may affect the future management of floodplain forests stressed by climate change.

Floodplain reengineering and associated structures

Rivers and their floodplains have been altered by water extraction, dams, reservoirs, water transfers, levees, and irrigation (Middleton 1999a, Jackson et al. 2001). These river alterations change the flow of water from river channels to floodplain forests with respect to the seasonality of flooding depth, duration, and frequency. Floodplain forests cut off from flood pulsing may be stressed by drought, and decreased nutrient supply (Jolly 1996). After freshwater extraction, floodplain forests may change in species composition or even die (Horner et al. 2009).

Dams

Dams alter floodplains both upstream and downstream of the structure and their effects depend on their purpose and position on the river. According to the serial discontinuity concept, the extent to which a dam alters the river floodplain depends on the position of the dam with respect to the floodplain (Ward and Stanford 1995). Upstream of dams, wetland tree production declines over time if permanently flooded, so that tree regeneration may be relegated to an elevation-specific ring in the driest part of the impoundment, e.g., *Taxodium distichum* forests in Buttonland Swamp, Cache River, Illinois (Middleton 2000 and Middleton and McKee 2005, respectively). Freshwater tree species have different tolerances to flooding, so that species compositions may “adjust” to flooding over time depending on flooding depth and seasonality. Along the impounded Wisconsin River floodplain, *Betula nigra* and *Fraxinus pennsylvanica* grow in flooded areas, whereas less flood-tolerant trees (*Quercus velutina* and *Q. ellipsoidalis*) have become less common over time (Predick et al. 2009). These patterns of flood tolerance are not uniform geographically, e.g., trees succumb more quickly in hotter climates because of the relationship of temperature and physiological constraints of the carbon balance during their respiration process (Middleton and McKee 2004), so that raised

temperatures under climate change may exacerbate this problem. Downstream of dams, incision and erratic in channel flow may deter water from entering the floodplain. If the elevation of water in the channel is always below the river bank level, the floodplain will dewater (Middleton 1999a).

Beyond obstructing water flow to floodplain forests, dam operations can interfere with tree growth, production, and regeneration by altering either the amount of water or timing of flooding or drying or both. For example, increased summer water flow from dam release along the Bill Williams River in western Arizona has stressed riparian trees, which could cause associated narrowing of stream channels and an overall reduction in vegetation extent (e.g., *Populus fremontii* and *Salix gooddingii*; Shafroth et al. 2002).

Dam operation also can have negative effects on forest regeneration. After 60 years of water regulation on the Missouri River, the floods of 2011 did not lead to forest recovery because subsequent flow from the dams did not resemble historical flow and sediment dynamics. Furthermore, the *Populus deltoides* saplings that had germinated in 1997 had a high level of mortality during the 2011 flood (Dixon et al. 2015). *Populus deltoides* regeneration is stifled by a shift in seasonal flood regime, specifically, the lack of high pulse of spring/summer flood flow followed by a drop in flow (Dixon et al. 2015). In northern floodplains, *Populus* seedling establishment near the river channel can be limited further by ice or water scouring (Scott et al. 1996). On dewatered floodplains adjacent to incised channels, lack of proper moisture conditions limits germination (Middleton 1999a, Dixon et al. 2015). Such dewatering of floodplains reduces sediment deposition (Coker et al. 2009). For these reasons, unregulated rivers generally have more tree regeneration than regulated rivers, e.g., the unregulated Santa Maria River had more seedling establishment than the downcut Bill Williams River in western Arizona (Shafroth et al. 2002).

Levees

Levees, canals, and water control structures alter the delivery of water and sediment to floodplains, depending on the relative positions of the embankment, channel, and floodplain (Sklar and Browder 1998). The flow of water from the channel to the floodplain is cut off on the backside of the levee, and impounded on the channel-facing side of the levee (Middleton 1999a). Over the last 100 years, the Mississippi River floodplain has become riddled with levees and water control structures to maintain the channel for transportation and control flooding. These structures deepen channel water for navigation when the river levels would otherwise be low in the dry season (summer and autumn). Levees prevent water from flowing from the channel onto the floodplain and have changed the seasonal hydrology and ecosystem function of forested floodplains of the Mississippi River (Fig. 3; DuBowy

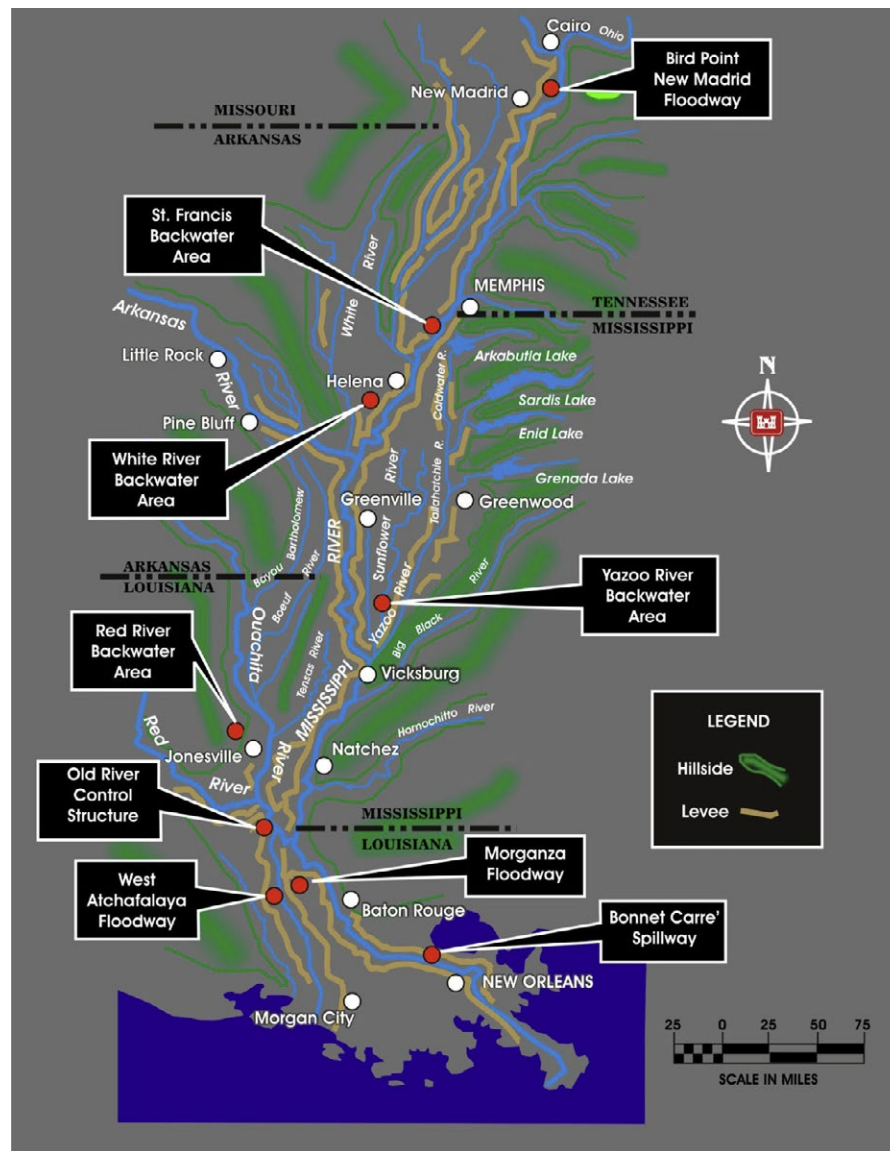


Fig. 3. Mississippi River and tributaries to illustrate historical modifications for navigation (DuBoway 2013), with permission of Elsevier.

2013). To control wet season flooding and dry season salt water intrusion, a large number of levees and other water control structures are being constructed in the Mekong River delta (Hoa et al. 2007). These dykes and weirs will increase flow velocities in rivers and canals, increasing bank erosion and both river and canal depth (Hoa et al. 2007). These hydrologic changes increase the risk of catastrophic dyke failure and flooding in unprotected areas. Reduced estuarine siltation caused by upstream dam construction and sea level rise are both predicted to enhance flooding in the Mekong River delta in Vietnam (Hoa et al. 2007).

Groundwater decline

The current rate of groundwater extraction for human use is unsustainable (Richey et al. 2015). The decline

in groundwater has had cascading negative effects for natural vegetation, especially riparian floodplain forests in arid and semi-arid regions (Dynesius and Nilsson 1994, Stromberg et al. 1996). Trees begin to die on floodplains if the groundwater falls below the rooting zone (Cunningham et al. 2011). Groundwater pumping, dams, and levees have dewatered floodplains along the Middle Rio Grande, which are normally flooded in the spring (Scott et al. 1999). *Populus* forests along the South Platte River and elsewhere also have been damaged by groundwater decline (Scott et al. 1999).

During drought on dewatered floodplains, forest composition may rapidly shift as groundwater levels decline. Over the past 100 years, damage to the floodplain forests of the Hassayampa River, Arizona has been related to both the extraction of ground and surface water (Stromberg et al. 1996). During drought, trees become

very stressed and forest composition begins to change because the tolerance of invasive species is sometimes superior to that of native species (Horton et al. 2001). For example, invasive *Tamarix* recovers faster from drought than native tree species (e.g., *Populus fremontii* and *Salix gooddingii*). Also, *Tamarix* maintains a viable canopy with less dieback and higher leaf gas exchange rate during drought than native species (Horton et al. 2001). Similarly, along the Lower Colorado River, *P. fremontii* has been less resilient to water and salinity stress than *S. gooddingii* and *Tamarix ramosissima*. Under these conditions, arid *Populus* gallery forests are becoming riparian scrub thickets dominated by *Tamarix* because this species has higher water-use efficiency and salinity stress tolerance (Busch and Smith 1995). Thus, droughts can be a forcing factor in long-term vegetation change (Busch and Smith 1995).

Temperature Change, Drought, and Floodplain Forest Function

Temperature increase will alter freshwater availability for wetlands via changes in evaporation and precipitation. While models from the IPCC (2014) predict global temperature and precipitation patterns, their projections for freshwater availability for wetlands are not very specific. Nevertheless, IPCC-predicted patterns of precipitation and/or water flow vary regionally. The HadGEM2-ES model predicts that water flow will increase in parts of North America, Asia, and Africa, particularly by the end of the 21st century, because of warmer temperatures (2.0°–5.5°C; IPCC 2013, IPCC 2014, Betts et al. 2015). Warmer air may cause more

precipitation (e.g., in Asia), because of the water-holding potential of warm vs. cold air (i.e., Clausius–Clapeyron relation). Other parts of the world are predicted to have more erratic rainfall with extreme flooding interspersed with drought or even mega-drought (southern Australia and U.S. Southwest/Central Plains, respectively; IPCC 2014 and Cook et al. 2015, respectively).

Long-term flooding trends may be more nuanced within regions than these IPCC models are predicting. For example, in the U.S. Midwest, stream gage records from the last 50 years show decreased flood frequency and magnitude to the southwest, and increased flooding in the central part of the region during the spring and summer (Fig. 2; Hirsch and Archfield 2015, Mallakpour and Villarini 2015). Site-specific flooding projections are of interest to local managers attempting to balance climate change realities with restoration and management planning in their natural areas. Based on actual field trends, managers may need to temper their practices based on long-term intra-regional flooding variation and trends. Fortunately, even though this level of local discrimination is not generally available, detailed intra-regional analysis of flooding trends is beginning to emerge from new research (e.g., Mallakpour and Villarini 2015; Fig. 2).

Implications of changes in glacial and snowmelt input

Climate warming may have particularly bad effects on rivers dependent on freshwater input from glaciers (Xu et al. 2009) and snowmelt (Mantua et al. 2010).

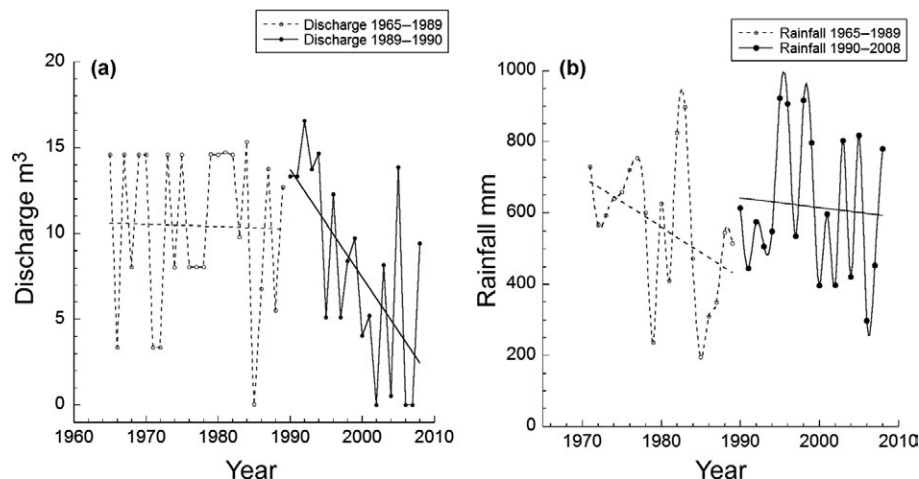


Fig. 4. Water in the Keoladeo National Park in comparison to the onset of major river diversion (~1990) including discharge in mean water discharge m³ per season (A), and regional annual mean precipitation in mm (B). Water is discharged into the park annually from Ajunbund (temporary reservoir) after the monsoon (September–February) via Ghana Canal on the Gambhir River. Linear regression lines indicate time trends in discharge: 1965–1989: $y = 34.3 - 0.01x$, $R^2 = <0.01$, $P = 0.9289$; 1990–2008: $y = 1255.4 - 0.62x$, $R^2 = 0.40$, $P = 0.0035$; and in precipitation: 1971–1989: $y = 28428.7 - 14.1x$, $R^2 = 0.17$, $P = 0.0822$; 1990–2008: $y = 6083.4 - 2.73x$, $R^2 = <0.01$, $P = 0.7542$. Discharge data are given by monsoonal year (July–September) for Bharatpur, Rajasthan, which is adjacent to the Park (Irrigation Department, Bharatpur, Rajasthan, India) (Middleton et al. 2015a; with permission of Elsevier).

The Himalayas are the water tower of Asia and the meltwater from these mountains sustains 22% of the earth's population (Xu et al. 2009). Glacial and snowmelt contribute between <5 and >45% of average flow to the region's major rivers (Xu et al. 2007). Melting and snow and glaciers provide about 70% of shoulder season (before and after monsoon rainfall) summer flow in the Ganges, Indus, Tarim, and Kabul Rivers (Xu et al. 2009). Snowmelt from the Himalayas feed rivers in Pakistan (Archer et al. 2010). Currently, the Himalayan glaciers are rapidly reducing in size due to rising temperatures (Xu et al. 2009). While increased melting will produce increased runoff, in the long term, snowmelt will decrease so that meltwater will become scarce as glaciers completely disappear or reach a lower equilibrium flow during the dry season (Xu et al. 2009).

Following the A1B model, as temperatures and precipitation increase in the western Himalayas, snowmelt will increase, so that summer water availability will decrease in downstream receiving rivers (Jeelani et al. 2012). Similarly, snowmelt from the Upper Mekong Basin is predicted to occur earlier (March) in the future, which will change the magnitude and timing of flooding in the Lower Mekong Delta (Taylor and Bach 2011). Climate simulations suggest that by 2080, there will be no snowmelt from the north Cascades (Washington State) to feed streams, so that channel water will be warmer, with more intense winter flooding and lower summer flow in the Columbia River (Mantua et al. 2010). Because salmon require seasonally appropriate water within certain temperature boundaries to complete their life cycle, climate change is likely to have severe effects on salmon populations (Mantua et al. 2010). This seasonal shift in water availability in the summer will cause some salmon populations to lack necessary freshwater for rearing young in the summer (Mantua et al. 2010). Increased winter runoff will decrease the survival of the egg to fry stage, so that these water flow changes will hamper salmon populations (Mantua et al. 2010). The proportion of available water demanded by humans in the summer/fall will increase if flow is low, leaving even less freshwater for salmon during a critical season (Mantua et al. 2010).

Abnormal drying in temporary wetlands

Freshwater diversions for cities, agriculture, and industry are an important factor in the drying temporary floodplain wetlands, even though the seasonal availability of freshwater is important in the annual cycle of vegetation regeneration (Middleton 1999a, An et al. 2007). For example, monsoonal wetlands in the Keoladeo National Park, India dried after inflowing river water was extracted for development (Fig. 4) and aquatic species became more isolated (Middleton 2009b). After these diversions, this park's only source of flooding

was rainwater, until recent restorative diversions were created (Middleton et al. 2015a). The extraction of freshwater in arid regions can have particularly serious effects on ecosystems, and there are many examples salinity levels rising in arid lakes after regional water extraction (e.g., Aral Sea [Central Asia], and Lakes Corangamite, Alexandrina, and Albert, Australia; Williams 2001).

Salinization with higher evapotranspiration and freshwater depletion

Freshwater flow into estuaries may be depleted by extraction and climate warming, but is critical to maintain salinity levels and related biogeochemical and trophic interactions (Sklar and Browder 1998), nutrient and organic matter cycling, and physical environments (Wetz and Yoskowitz 2013). Yet, the importance of freshwater inflow for the normal functioning of coastal floodplain forests is often overlooked in water allocation planning (Gillson 2011, Wetz and Yoskowitz 2013). Also, the water quality along rivers feeding coastal wetlands may be degraded by development activities, which can alter salinity, nutrient, and sediment input (Montagna et al. 2013). Regular freshwater inundation of floodplains may be necessary to reduce salinity or other factors but may not be possible if the water delivery system is broken.

Freshwater extraction is a particular culprit in the salinization of coastal water systems because this practice alters the mix of fresh and saltwater (Naiman et al. 1995) and also reduces nutrient and sediment loading (Montagna et al. 2013). When fresh groundwater is extracted from aquifers adjoining the coast, marine saltwater may move into freshwater aquifers (Walther et al. 2013). Freshwater extraction can also increase the salinization of wetlands in inland arid regions (Jolly et al. 2008).

Not surprisingly, arid river vegetation is particularly sensitive to precipitation change, because precipitation affects freshwater availability (Carpenter et al. 1992). Some estuaries already have increased frequency and/or intensity of droughts, floods, hurricanes, and very hot temperatures. Climate models predict that these conditions will spread to other world estuaries (Wetz and Yoskowitz 2013). More worrying is that according to IPCC (2014) models, there is a very high confidence level of submergence of many coastal ecosystems, which will further compound problems created by the human-induced drivers of landuse, development, and pollution (Wong et al. 2014). Rapid human population growth on the coasts has undermined natural ecosystems because of freshwater diversion, sea-level rise, and coastal subsidence. As coasts become inundated, species of freshwater tidal forests could migrate inland, but this may not be realized because many coastal zones are highly urbanized (Craft 2012). Also, floodplain forest species have a

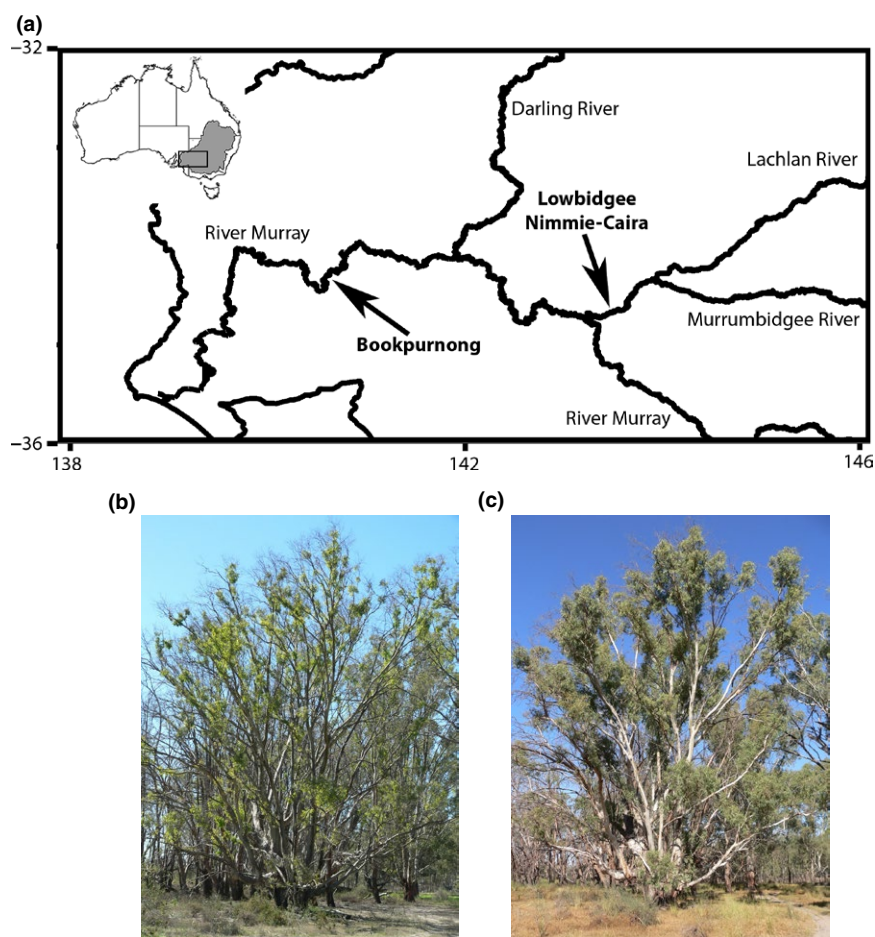


Fig. 5. (a) Map of the River Murray and Darling River in South Australia. (b) Red gum trees before and after hydrologic remediation in 2005 vs. 2008 before and after hydrologic remediation, Bookpurnong, River Murray Floodplain. Photos by Melissa White of the Department of Environment, Water and Natural Resources, Australia.

limited ability to disperse against the direction of water flow (Middleton 2000).

Alterations to the water delivery system change the vegetation structure and function of estuaries (Wetz and Yoskowitz 2013), but these changes also hamper opportunities for regeneration. As an example, *Sabal palmetto* regeneration has been limited in coastal Florida as tidal inundation increases, even though adult individuals persist in relict stands until they are replaced by salt marsh (Williams et al. 1999). Regeneration is a key process in the maintenance of forests (Middleton 1999a), so that salinization-limiting forest regeneration is a critical emerging problem for wetland managers.

River regulation alters freshwater status in forested wetlands by changing freshwater input from groundwater discharge, surface flooding from the river channel, and moisture availability pattern on the floodplain (Berens et al. 2013). Weirs are a particular problem because these may hold water levels constant in the channel, causing inappropriate levels of seasonal flooding on floodplains, e.g., along the Murray River (Fig. 1A; Walker 2006). In these same floodplains, salinity has moved

upward through the soil to the rooting zone and soil surface via groundwater discharge (Overton et al. 2006). While trees die in situations where the groundwater is below the rooting zone, mortality may also occur if the groundwater is too salty for species (Cunningham et al. 2011). Along the Murray River, *Eucalyptus camaldulensis* trees have died in floodplain forests because of groundwater salinization related to drought and river regulation (Jolly 1996, Slavich et al. 1999, George et al. 2005, Cunningham et al. 2009, Souter et al. 2014). Further away from the channel in floodplain grasslands (*Pseudoraphis spinescens*), *E. camaldulensis* trees have invaded the grassland because of related changes in hydrology (Barmah-Millewa Forest; Bren 1992).

Geographical distribution shifts in a warming world

Whether or not floodplains have been altered, biotic changes will follow temperature rise as predicted by the IPCC (2014). Suitable climates for southern tree species are likely to emerge northward of their

current range in sync with global temperature rise (Prasad et al. 2007, Iverson et al. 2008, Fisichelli et al. 2014). Exotic species with appropriate life history requirements may invade floodplain forests, facilitated by flow regime changes (Jolly 1996, Bunn and Arthington 2002). Climate, invasive species, diseases, and insect pests will all interact to change native species composition in the future (Fisichelli et al. 2014).

Forested Wetland Health Assessment

An assessment of vegetation stress or shift in ecosystem function can signal the need to hydrologically remediate floodplain forests during climate-induced drought or salinization. Vegetation stress is a primary indicator of unacceptable levels of wetland alteration, so that the early detection of stress might be used to signal hydrologic remediation. Recent studies give cause for hope as floodplain trees can have tremendous increases in crown health and/or production after short-term freshwater remediation (2 vs. 4 months; Souter et al. 2012 and Middleton et al. 2015b, respectively). Such studies indicate that future climate change problems may be remediated, reducing tree stress. Moreover, the development of health assessment to signal remediation could be important for reviving forested wetlands suffering from drought or salinization.

The Living Murray Initiative in southeastern Australia serves as a good example of tree assessment used to chart progress toward restoring healthy forests (Souter et al. 2010; Fig. 5a). As part of vegetation health assessments, reference conditions can set standards for water quality and delivery (Baron et al. 2002). Another monitoring approach could be to assess functional attributes following disturbance related to nutrient spiraling length, and rates of vegetation recovery, production, and respiration (Boulton 1999).

The regeneration capability of dominant trees is a simple indicator of forested wetland health (Middleton 1999a). Dominant species that cannot regenerate after disturbance exist in a relict state (Williams et al. 1999), so that ecosystems with such species will not be resilient (i.e., ability to maintain structure and function; Rapport et al. 1998). Regeneration capacity varies across species ranges depending on the fit of the species to the local environment, so that a geographical perspective is important in climate change analysis (Middleton 2009a). Across species distributional ranges, environmental requirements for seed germination and seedling/sapling recruitment are narrow relative to those of adults, so that climate change is likely to affect younger cohorts differentially (Jump et al. 2009). Therefore, early life history requirements of floodplain species may ultimately determine their long-term ability for migration, particularly near the edges of their ranges (Middleton and McKee 2004, Middleton 2009b, Middleton and McKee 2011).

While floodplain species might establish toward the geographical poles of their current range, in many cases, floodplain species are not readily dispersed upstream (Middleton 2000). Species have limitations for establishment; certain boreal species already may be experiencing regeneration failure at the southern edge of their distributions (Fisichelli et al. 2013). In addition to these constraints, species will face other limitations to their range migration, e.g., species living in environments at the edge of their physiological tolerance may not compete successfully with other species (Parmesan et al. 2005).

Tree structure and regeneration assessment

Vegetation health assessment can detect critical levels of stress and may be useful for planning restorative actions. Unfortunately, while the detection of pre-threshold conditions would be useful in alerting managers of the need to deliver freshwater to forested wetlands, early tree stress indicators have been difficult to identify (Eamus et al. 2006). A few successful field techniques have emerged, particularly from Australia. To determine tree stress, tree characteristics can be assessed including crown extent and foliage density, epicormic growth, branch tip growth, reproductive activity, leaf die-off, mistletoe infestation, bark condition (Souter et al. 2010), and percent live basal area (Cunningham et al. 2007). Field surveys should record tree status along with dbh and dominance class because trees of various size classes may respond differently to water stress (Souter et al. 2010). Changes in crown health can be assessed with measurements of leaf litter production (Middleton et al. 2015b). In riverine forests of eastern Colorado, a decrease in crown volume preceded the death of adult *Populus* spp. after groundwater decline (Scott et al. 1999).

Visual assessments of tree health are an integral component of many ground surveys in determining tree stress, but these have some level of subjectivity. Blind methods of comparing assessments of crews have reduced biases related to personal differences in the scoring of tree health (Souter et al. 2010). Reference photographs of tree crown conditions before and after freshwater remediation can also improve the accuracy of the visual assessment of tree health (Fig. 5A, B; Solberg and Strand 1999).

A lack of regeneration of dominant species particularly after natural disturbances should send a powerful message to managers that target species have entered a relict state (Middleton 1999a, Williams et al. 1999). Thus, one simple way to address the status of vegetation health is to compare the regeneration potential of various life history stages by observing the relative contribution of adults, seeds, seedlings, and vegetative propagules to the re-vegetation process (Middleton 2009c). Similarly, the regeneration capacity of invertebrates, protists, and algae in dry temporary wetlands

are useful indicators of ecosystem health (Skinner et al. 2001, Norris et al. 2007).

Tree production, photosynthesis, and growth assessment

Measures of ecosystem function before and after remediation provide useful information on the effects of such strategies (Eamus et al. 2006). A sharp increase in tree litter or seed production was detected using collection devices after freshwater remediation along the Murray and Mississippi Rivers (litter vs. seed captured: Middleton et al. 2015a vs. Jensen et al. 2008, respectively). As a prethreshold detector of high levels of tree stress in arid forests (*Populus* spp.), the detection of a sudden decline in annual branch growth signaled tree health problems (Scott et al. 1999). Branch growth increment was a more sensitive detector of stream flow change than tree ring increments for *P. angustifolia*, *P. balsamifera*, and *P. deltoides* (Willms et al. 1998). Similarly, after hydrologic remediation in *Taxodium distichum* forests, tree trunk growth was less responsive than litter production (Middleton et al. 2015b).

Remote assessment of forest health

Managers often maintain huge and inaccessible conservation areas, so that ground detection of vegetation stress can be difficult. In such situations, remote sensing could allow managers to preemptively detect vegetation stress. Along the River Murray, tree health has been assessed by combining Landsat images with groundtruthing assessments of live basal area, plant area index, and crown vigor (Cunningham et al. 2009). Based on the remote sensing assessments, only 30% of the upstream *E. camuldulensis* stands were in good condition during the Millennium drought, with condition declining further downstream in areas with more river regulation (Cunningham et al. 2009). QuickBird images have been used to map riparian savanna health based on spectral reflectance patterns from the Alligator River, Kakadu National Park, northern Australia. These images can be used to detect specific structural characteristics related to tree stress including the aerial patterns of canopy gap, foliage clumping, tree crown, and stand density (Johansen and Phinn 2006).

PolSAR data have been used to detect salt marsh stress, but this technique also might have application in forested wetlands (Ramsey et al. 2014). PolSAR detects pigments levels, which are related to the photosynthetic capacity of plants, and thus analysis of these images can detect vegetation stress (Ramsey et al. 2014). Another advantage of these remote sensing techniques is that plant stress may not be readily detectable by direct human observation. Early physiological responses to stress are more easily detectable in leaf optical properties, such as

detection of carotene and chlorophyll pigment changes (Ramsey and Rangoonwala 2005). While the detection of stand condition by either field or remote sensing techniques is critical to managers for determining vegetation health, these vegetation techniques can also give good proximal estimates of faunal impacts (Mac Nally et al. 2014).

Water availability, modeling, and assessment

Human demands for freshwater underlie many of the current challenges of forested wetland management, so that socioeconomic models of water stress can be made by observing the balance between water availability and the rate of water withdrawal. Over-extraction and conflict can make freshwater less accessible to humans, so that poor human health and suffering ensues (Pfister et al. 2009). Despite population increase, there has been an encouraging trend in water usage in the United States (13% less water withdrawal in 2010 vs. 2005) because of improvements in water use efficiency in almost all categories (e.g., public supply, irrigation, industrial) so that water withdrawal in 2010 was at its lowest level since before 1970 (Maupin et al. 2014). In Australia, total water consumption dropped by 28% during the Millennium drought, with an 11% drop in domestic consumption (Australian Bureau of Statistics, 2012). However, water use has rebounded after the drought, largely driven by increased agricultural consumption, while domestic water use has remained about the same as at the end of the drought (Australian Bureau of Statistics, 2014). With predicted droughts in the future, water availability is expected to decrease and knowledge of potential effects of water scarcity can be gained through modeling (Naiman et al. 1995). Such efforts will give insight into necessary levels of freshwater availability for humans and natural ecosystems.

Achieving Riverine Ecosystem Integrity

The ecosystem integrity of floodplain forests is driven by dynamic drivers such as flood pulsing, which shape ecosystem structure and function (Junk et al. 1989), yet flood pulsing is disrupted by water control structures (Middleton 1999a). In coastal and arid settings, freshwater pulses from rain or rivers are even more critical to flush salt from the soil to maintain freshwater habitats (Jolly et al. 2008). The challenge for forested floodplain managers is to maintain ecosystem integrity with flood pulsed conditions (Middleton 1999a), and also to have enough freshwater for potable water supply, irrigation, and power generation (Tockner and Stanford 2002). In addition to describing the importance of ecosystem integrity, this section will describe three case studies' management responses, which may come to characterize responses during climate change including hydrologic remediation along the

Murray River and Mississippi River. Also, we will discuss the challenges ahead for Cambodia using the Tonle Sap Great Lake as an illustration.

Overall, the pragmatic conservation management with respect to climate change is: "How little water is enough?" and unfortunately knowledge gaps make this question difficult to answer. Wetland ecosystem integrity relies on more than just water volume, but also water quality, seasonality and flow variability. Climate change will complicate attempts by managers to remediate disturbance responses of wetlands along hydrologically altered rivers (Palmer et al. 2008), but a number of techniques may be useful to help maintain freshwater floodplain forests. Downstream of dams, water release techniques designed to simulate unregulated conditions might be employed to ameliorate ecosystem damage (Baron et al. 2002). Water control structures in rivers might be removed, and this removal process has occurred along portions of the Kissimmee River in south Florida (Baron et al. 2002). Water might be re-diverted from another river to supply freshwater to a floodplain cut off from its original source of water (Middleton et al. 2015a). Levees and embankments might be removed or moved back from the channel. For example, forest regeneration commenced after embankments were removed along the Regelsbrunner Au, River Danube, Austria (Hughes and Rood 2003). Dried floodplains along incised river channels are a special problem for restoration because of the difficulties of rewetting the floodplains to recapture the correct moisture setting for regeneration (Middleton 1999a, Dixon et al. 2015).

Management to support freshwater flow to coastal wetlands is critical in modulating physicochemical conditions and the necessary flow depends on tides, wind, topography, ocean currents, and latitude. Seasonality of this flow is also important; peak flows occur in the spring in northern areas with snowmelt, but have high flow variability in monsoonal environments (e.g., eastern Australia; Gillson 2011). Water flow itself is important to remove toxic anaerobic products from the soil, increase oxygenation, and promote vegetation health (Kozlowski 2002). Flushing time of a coastal waterbody determines its ability to deal with stresses and can be accelerated by waterway re-engineering (Wolanski et al. 2004). While freshwater inflow is critical to the function of coastal floodplain forests, this factor is often overlooked, so that estuaries are often on the losing side of the battle between humans and nature for freshwater (Gillson 2011, Wetz and Yoskowitz 2013).

The creation of dams to deepen water in dewatered wetlands may prolong flooding, but does not mimic flood pulsing nor does it necessarily improve vegetation health (Middleton 1999a). Trees in permanent artificial impoundments have lower production levels than elsewhere (Middleton and McKee 2004, 2005). Nonetheless, impoundment has been used successfully to revive *Eucalyptus* trees along the Murray River in Australia (Fig. 5B, C; Holland et al. 2009, Berens et al. 2013).

Reducing salinity in groundwater requires special consideration, especially for the conservation of coastal forests. Groundwater can be freshened by groundwater pumping, reducing discharge or increasing recharge from freshwater sources, increasing aquifer recharge, desalinizing the aquifer, stopping inflow of saline water, and/or assisting influx of freshwater from other aquifers (Werner et al. 2013).

Lower River Murray management response to the Millennium drought (Australia)

The River Murray in southeastern Australia is among the world's most regulated rivers (Fig. 1A). The combined impact of this regulation and the Millennium drought produced considerable stress in its forested floodplain wetlands due to lack of freshwater. Water is extracted primarily for irrigated agriculture and only 29% of the rivers annual flow reaches the sea (CSIRO 2008). However, flow to the sea completely stopped for a prolonged period during the Millennium drought of 2000–2010 (Murray-Darling Basin Authority 2015). The Millennium drought curtailed freshwater flow to the floodplain with two major consequences including widespread vegetation die-back (Souter et al. 2014) and the exposure of wetland acid sulfate soils along the river's lower reaches (Kingsford et al. 2011). Overall, water regulation has caused numerous ecological impacts in this region including tree mortality (Souter et al. 2014) and limited regeneration (George et al. 2005, Jensen et al. 2008).

During the Millennium drought, prolonged lack of overbank flooding combined with naturally highly saline groundwater produced widespread stress and mortality of floodplain trees. With insufficient water to meet even agricultural needs, and without the infrastructure required to flood large areas of floodplain, a variety of management approaches were used to save some of the stressed trees. At the start of the drought in 2000, river level was raised using in-channel weirs to inundate a small portion of the floodplain (Siebenritt et al. 2004). This inundation caused flood-intolerant plants to die-back, while only a few flood-tolerant species germinated in the remediated environments (Siebenritt et al. 2004). Because the inundated floodplains had a depauperate seedbank, flow management only maintained degraded communities without restoring flood-dependent vegetation (Siebenritt et al. 2004). Mid-way through the drought in 2005, unregulated flows to South Australia allowed the river level to again be raised using in-channel weirs (Souter and Walter 2014, Souter et al. 2014). This raising of the water level in the channel inundated some of the floodplain providing benefit to some (Souter et al., in review and unpublished data), but not all, understory floodplain species (Souter and Walter 2014). The rise in water level also recharged local saline groundwater with fresh river water, which resulted in improved *E. ca-*

maldulensis health (Souter et al. 2014). Healthy trees responded more positively to freshwater because healthy trees are three times more likely to respond than stressed trees and thirty times more likely to respond than defoliated trees (Souter et al. 2014). However, 48% of the trees that had lost all of their leaves produced new growth (Souter et al. 2014). The effects of this remediation lasted more than 1 year. Hydrologically remediated trees during the Millennium drought had nine times more seed production than unremediated trees (Jensen et al. 2008). In general, there was little tree regeneration during this drought because of poor moisture conditions and high soil salinity (George et al. 2005).

As the drought progressed, water became scarcer and it became impossible to raise the river level to inundate the floodplain. Thus, to provide water to increasingly stressed trees, freshwater was pumped to deflation basins and dry floodplain channels to provide water to trees via groundwater recharge (e.g., Holland et al. 2009, Berens et al. 2013). For example at Bookpurnong (Fig. 1A), artificial flooding reduced root zone salinization, and trees responded through the growth of new crown and epicormic foliage (Berens et al. 2013). Remediation sites were selected for feasibility and ecological importance. Priority was given to sites with: minimal earthworks required to retain water, ease of approval from landholders, accessibility, and proximity to water. Due to the emergency nature of the hydrological remediation, ecological importance was assessed by expert opinion. Sites with higher densities of less severely stressed trees were prioritized for hydrologic remediation over sites with low densities of extremely stressed trees as based on tree condition assessment (Souter et al. 2012). At this time, a series of additional experiments were conducted to determine the feasibility of injecting fresh water into the saline aquifer to improve tree condition. An attempt to inject river water into the aquifer failed due to the hydraulic properties of the floodplain soils, which clogged the injection wells. The injected water did not travel far through the aquifer with only a few *E. camaldulensis* trees adjacent to the wells responding (Berens et al. 2009). Furthermore, this technique would not have allowed the long-term persistence of the trees, because the approach did not address seed germination requirements via surface inundation (Berens et al. 2009).

Pumping river-bank-side saline groundwater into a regional salt interception scheme was successful in injecting freshwater from the river into the river bank. After groundwater freshening, *E. largiflorens* health improved near the river, while trees farther from the river stayed unhealthy (Doody et al. 2009). As with aquifer injection approach, this bank injection method did not provide the surface inundation required for seed germination.

The re-flooding of deflation basins, aquifer injection, and groundwater pumping were costly procedures, which produced only localized effects. However, during situations such as the Millennium drought, these

procedures were the only methods available to provide water to ecologically important floodplain vegetation. This problem is also likely to be the case in a climate-constrained future. One of the main lessons for a future of drought was that remediation is more effective when applied to healthier rather than stressed trees, which means earlier rather than later intervention is usually the most successful.

The provision of water to stressed vegetation during the drought was a useful management intervention in South Australia. When floods returned in summer 2010–2011, the ecological responses of the forests depended on their preflooding conditions. Areas that received environmental flows in 2009–2010 rebounded more successfully in rain-induced flooding in 2010–2011. Frog and tadpole abundance increased in 2010–2011 at the Lowbidgee wetlands (Fig. 5A), while endangered southern bell frogs *Litoria raniformis* were found in the Nimmie-Caira wetlands (Fig. 5A), where they had been absent for 5 years. Lowbidgee wetlands with hydrologic remediation increased in water bird breeding, native fish, tadpoles, and plant productivity, along with decreased amounts of blue green algae (Spencer et al. 2011).

The lack of water during the Millennium drought saw many previously inundated wetlands draw down, unexpectedly exposing sulfidic sediments. The exposure of these soils to oxygen produces either sulfuric acid (McCarthy et al. 2006, Fitzpatrick et al. 2009) or noxious odors (Lamontagne et al. 2004). The problems caused by sulfidic sediments demonstrate that unexpected consequences may result from extreme conditions.

Freshwater flow remediation in the Mississippi River and elsewhere

The estuaries of the lower Mississippi River have been greatly altered via isolation from their original freshwater sources because of the construction of levees and other engineering structures (DuBowoy 2013). For this and other reasons, the estuaries of the Lower Mississippi River Alluvial Valley are losing elevation (Paola et al. 2011, Falcini et al. 2012). When some flood pulsing was restored from the Mississippi River during the flood of Spring 2011, water was discharged onto the Atchafalaya Basin (tributary of the Mississippi River), and up to several centimeters of sediment were deposited in the floodplain forests (Falcini et al. 2012).

To address coastal sinking and hydrological alteration along the lower Mississippi River, seven freshwater and sediment diversions are planned or already operating in Louisiana (DuBowoy 2013). Also, to improve floodplain environments, some levee setbacks have been planned or implemented on the Mississippi River (Opperman et al. 2009). Notches and chevrons may be used to minimize the effects of navigation structures by dividing flow between the channel and the floodplain along the

lower Mississippi River (DuBowoy 2013). Ultimately, hydrologic remediation of these floodplains can improve floodplain forest health. After 4 months of hydrologic remediation, coastal floodplain forest litter production increased sharply for more than 2 years, and may have been related to the reduction in salinity, anoxia, or root toxins in the soil (Middleton et al. 2015b).

In another North American floodplain setting, controlled streamflow from dams increased downstream tree regeneration in western *Populus* forests (Shafroth et al. 2002). In general, regeneration on the floodplains of altered rivers is poor (Liu et al. 2005, Rood et al. 2005). To encourage regeneration, dam operators have released water during high spring flow followed by gradual flow decline, and this procedure has created moisture environments conducive to the recruitment of cottonwoods and willows below the Truckee River Dam, Nevada and downstream of St. Mary River Dam and Oldman Dam on Oldman River and South Saskatchewan Rivers, Alberta Canada (Rood et al. 2005). Elsewhere, water has been diverted from the Tarim River in China for development. Subsequently, there has been no channel flow for 30 years and *Populus euphratica* regeneration has been very limited (Liu et al. 2005). Similarly, *Populus deltoides* regeneration has been limited on the developed floodplain of the upper Colorado River (Northcott et al. 2007). Judicious usage of water release during the appropriate season might help improve regeneration opportunities in these regulated river systems. To that end, modeling has been useful to provide sedimentation scenarios and to guide flow manipulations for the restoration of habitat below the Glen Canyon Dam (Rubin et al. 2002).

Tonle Sap Great Lake (Cambodia)

Cambodia's Tonle Sap Great Lake is the largest natural freshwater lake in South-East Asia and is listed as a UNESCO biosphere reserve due to its high biodiversity, but development will complicate conservation efforts if climates change (Yusuf and Francisco 2009). This lake provides important habitat for 17 species of globally threatened and near-threatened vertebrates (Campbell et al. 2006) and its fishery is the foundation of Cambodia's food security (Cooperman et al. 2012). Nearly 6000 km² of the Tonle Sap floodplain is covered by native vegetation (Arias et al. 2013), which is comprised of swamp forests (covering 10% of the floodplain), swamp scrublands (80%), grassland, and cleared farmland (Campbell et al. 2006). While the flood pulse is the underlying driver of vegetation community structure (Arias et al. 2013), the complexity of the vegetation community is a product of the interaction between human use and hydrology (Arias et al. 2013). Local communities alter the floodplain by clearing land for farming, grazing cattle, burning and harvesting firewood and timber (Campbell et al. 2006, Arias et al. 2013).

The combined impact of upstream dam construction and climate change is expected to result in higher water levels during the dry season, permanently inundating and reducing areas of forest (Kummu and Sarkkula 2008); during the wet season, the maximum extent of flooding will be reduced compared to current times (Lauri et al. 2012, Arias et al. 2013). This development activity will likely contribute to decreased native vegetation because these species are limited by flooding duration (Arias et al. 2013). However, climate change impacts increase the uncertainty of predictions related to hydrological changes (Lauri et al. 2012). Because the Tonle Sap is a large, complex, and poorly known ecosystem (Lamberts and Koponen 2008), the wider ecological impact of these potential changes to the floodplain vegetation is unknown, but likely to be significant because the vast majority of the Tonle Sap's primary productivity is produced locally (Lamberts and Koponen 2008). Any lack of adaptive capacity of developing countries to address ecosystem shifts due to climate change may negatively impact these natural environments.

As one of the world's least developed countries, Cambodia suffers from a lack of adaptive capacity and as a result is highly vulnerable to climate change (Yusuf and Francisco 2009). This lack of capacity also limits the ability to collect the information needed to understand and manage ecosystem changes (Junk et al. 2006).

Conclusions

As human demand for freshwater increases, an equitable approach needs to be developed to share freshwater for human needs and forest management. Such a strategy is necessary to maintain the integrity and function of natural ecosystems or we may be faced with system collapse. These perspectives are especially important in view of major droughts predicted, which will be induced by climate change in North America (Cook et al. 2015), southeastern Australia (van Dijk et al. 2013), and elsewhere (IPCC 2014). Freshwater resources are being rapidly depleted from both surface and groundwater sources (Cook et al. 2015, Richey et al. 2015). An enormous challenge lies ahead for land managers who desperately need management solutions in the context of locally specific water availability information, long-term field studies, and predictive modeling tools to inform their contingency planning of management and restoration projects. Hydrologic remediation is an important emerging technology for this future. Studies along the River Murray, Australia show that earlier intervention is most effective in aiding the survival of freshwater forest trees in severe drought. The ability to hydrologically remediate the effects of climate change in the developed world is supported by decades of intensive research and the ability to rapidly mobilize human resources and infrastructure during times of stress. The knowledge and

resources needed to react to new future climates are largely lacking in the developing world, which may have severe consequences for both people and flooded forests. Future conservation and management efforts depend on identifying successful strategies to maintain biodiversity and function in freshwater floodplain forests.

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