Restoration, Ecosystem

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[illegible] vernmental Panel on [illegible] limited success (Welch and Cooke, 1987) (**Lakes and Rivers**, Volume 2).

Wetland Ecosystems

Climate change threatens to stress wetland species through direct effects of temperature and carbon dioxide (CO_2) concentration and through changes in hydrology due to precipitation, evaporation and sea level changes (Michener *a* ., 1997). Temperature increases are likely to damage wetlands that store carbon in the form of organic matter. Canadian peatlands and the Arctic tundra could release major quantities of carbon to the atmosphere through increased decomposition rates as a result of warming or drying (Intergovernmental Panel on Climate Change, 1996). Experiments with multiple factors and many types of wetlands are needed to make long-term predictions of community change, however. The hydrologic restoration of drained peatlands in advance of climate change would help buffer the globe against these impacts.

Small ponded wetlands and dry-end wetlands will be very sensitive to altered hydrology (Sorenson *a* ., 1998). Because half of North America's waterfowl are reared in prairie potholes, and because duck production is directly correlated with the number of small ponds, there would be measurable impacts to waterfowl following changes in precipitation and evaporation (Sorenson *a* ., 1998). The acceleration of current efforts to restore drained potholes throughout the Midwest could mitigate some of the negative impacts in advance, but biodiversity would still likely decline, as not all species return with renewed ponding (Galatowitsch and van der Valk, 1996).

Many wetlands have lost their natural water regimes, especially flood pulsing and the accompanying mechanical disturbances and nutrient influxes that invigorate floodplain wetlands (Middleton, 1999). In some cases, restoration may be accomplished simply by removing impediments to water flow (Gilbert and Anderson, 1998). Where dams permanently reduce flooding of wetlands, significant improvements can be achieved by periodic water releases that mimic seasonal floods (Vaselaar, 1997; Middleton, 1999).

Hydrological changes may be greatest for coastal wetlands, where sea levels in many areas may rise more than 0.5 m by 2100 (Intergovernmental Panel on Climate Change, 1996). Salt marshes will be lost where sea walls obstruct their migration inland and/or where sedimentation cannot keep pace with inundation. Restoration efforts would help mitigate future losses due to sea level rise. Methods include reestablishing tidal flow through removal of dikes, reintroducing species, and excavating new wetlands (Simenstad and Thom, 1996; Zedler, 1996a; Williams and Watford, 1997). Inundation problems can be combated by encouraging sedimentation and marsh building (Smit *a* ., 1997) or accommodated by 'managed retreat'. In the latter process, levees can be breached to allow salt marsh to reestablish in low lying areas (Packham and Willis, 1997; Gilbert and Anderson, 1998) or broad inland buffers can be set aside to allow salt marsh migration up slopes.

One of the most damaging effects of climate change on coastal wetlands may be from increased storm frequencies and magnitudes. Extreme high tides can carry salts inland on to intolerant vegetation and soils. Salty soils will favor the inland migration of halophytes, which will be necessary to maintain salt marshes where sediment accretion cannot keep up with inundation (Michener *a* ., 1997).

Species will likely migrate into newly inundated areas at different rates and be affected by different substrates and competitors inland, so some species may need to be translocated. The high-intertidal marsh of Southern California, US, is a good example of an assemblage that may need to be translocated in anticipation of sea-level rise. Three high-marsh perennials are already rare and they rarely reproduce from seed. *F a a a* has only one natural occurrence in the US, but in Mexico it occurs in both salt marsh and upland areas. *Sa c a b a* (= *A c b a*) and *M a c a* are more widespread in Southern California but their populations have been greatly reduced by trails, roads, and other disturbances. These species show promise for experimental planting in upland buffers. Plants should be grown from seed to provide genetic diversity, so that selection could operate after tides begin to inundate the populations. The benefits (enhanced genetic diversity) and risks (contaminaflood control), water diversion canals (designed to move water to Southern California), and islands (leveed, because that other forest species of plants and animals will establish a functioning ecosystem (Ashby, 1987). One of the more difficult factors to account for in future restoration of forests will be the tight symbiosis between many plants and mycorrhizal fungi. Most plants depend on mycorrhizae for efficient uptake of water and nutrients, but the abilities of soil organisms to adapt to climate change are unknown. The ability of restoration to promote the rapid reestablishment of forest functioning will depend in large part on our ability to establish healthy populations of soil organisms (Haselwandter, 1997).

Longleaf pine (*P* *a*) forest ecosystems provide a good case history for restoration of forest landscapes. Longleaf pine forests once covered 60% of the Southeastern US coastal plain, stretching from North Carolina to Texas. Longleaf pine forests host a surprising diversity of plant and animal species, including a large number of threatened and endangered species. Critical to the maintenance of these systems are periodic fires carried by understory plants that prevent invasion by deciduous oaks. Today, less than 10% of the original forests remain intact (Johnson and Gjerstad, 1998). This loss of a valuable ecosystem has led to an increasing interest in restoring longleaf pine communities by reintroducing fire and establishing populations of indigenous plants and animals. Some of the largest restoration projects ever undertaken are now in progress. The US Department of Defense military bases alone manage approximately 400 000 ha of longleaf-type vegetation (Johnson and Gjerstad, 1998).

Wiregrass (*A* *a* spp.) forms the dominant groundcover in most natural longleaf communities and provides fuel to carry fires (Means, 1997). Attempts to create a wiregrass understory in longleaf pine restorations provide an informative example of the limitations of restoration efforts. While wiregrass populations are present in many stands undergoing restoration, and plantings can be established with relative ease, these populations have typically failed to spread (Means, 1997; Seamon, 1998). Thus, while the species is present in the community, it fails to expand to serve a necessary function in the ecosystem (e.g., promoting the spread of beneficial fires).

Large-scale restoration of tropical forests has been initiated in both Latin America, Asia, and Australia. In Costa Rica, restoration efforts have focused on facilitating the natural recovery of ecosystems by promoting regeneration of forest trees and by attempting to control disturbance due to fire (Janzen, 1988; Holl, 1998). An alternative approach is to increase the ecological integrity of the large areas of tropical forest that have been converted to timber plantations. These areas may be improved, though not restored to a pristine state, by promoting the use of native species, planting mixtures of species, and altering landscape patterns

6. Carefully monitored, experimental translocations of

how humans can intervene in disturbed communities to

Williams, R J and Watford, F A (1997) Identification of Struc-