Restoration as experiment

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Abstract: Large-scale ecological restoration generally employs un-replicated trial and error to re-create habitats destroyed or degraded by human activity. Trial and error follows a management plan that employs the "best available practice" for each habitat type; adaptive management reflecting experience subsequently corrects errors. The process is slow because each restoration is often well-advanced before adjustments are attempted. Rare are simultaneous replicated trials during the initial restoration or corrective process. "Systemic experimental restoration" would design replicated planting or management contrasts at the outset of large-scale public and commercial restorations. Alternative treatments create mosaics of different manifestations of a community within a mosaic of habitat types. Replicated contrasts within habitats allow the inference of cause and effect of success and failure on scales of communities, landscapes and ecosystems. For the long-term development of restoration ecology as a science, semi-natural communities of known contrasting histories will be important to encourage managers or investigators to create additional projects beyond explicit management or scientific objectives of an initial management plan. By consciously, integrating heterogeneity throughout restoration management plans, systemic mosaics of treatments within and between habitats, would facilitate multiple habitat responses to unpredictable changes in climate and land use.
Key Words: alternative states; ecological processes; mosaics within mosaics; systemic experimental restoration; trial and error restoration; typology.

Resumen: A menudo se intenta recrear hábitats degradados o destruidos por actividades antropogénicas mediante restauración por ensayo y error que no incluye réplicas. Los proyectos por ensayo y error son comunes y están basados en planes que incluyen "la mejor técnica disponible" para cada tipo hábitat. El manejo adaptativo incluye usar la experiencia y corregir errores durante el proceso. Este proceso es lento debido a que cada proyecto de restauración se encuentra ya muy avanzado antes de que se intenten realizar ajustes. En estos casos, rara vez se hacen pruebas que incluyan réplicas al inicio o durante el proceso de rectificación. Aquí se propone "la restauración experimental sistémica", que buscará diseñar desde el inicio plantaciones o áreas de manejo en proyectos públicos o comerciales que incluyan réplicas con tratamientos contrastantes. El uso de tratamientos alternativos permitirá la creación de parches que serán expresiones diferentes de una comunidad dentro de un mosaico de hábitats. Hacer réplicas de tratamientos contrastantes permitirá hacer inferencias acerca de las causas y los efectos del éxito o fracaso de un procedimiento a la escala de comunidades, paisajes y ecosistemas. El desarrollo de la ecología de la restauración como ciencia a largo plazo se verá beneficiado con la creación de comunidades semi-naturales que tendrán historias contrastantes, e invitará a los administradores y científicos a crear proyectos valiosos, más allá de las metas explícitas de manejo o aquellas científicas incluidas en un plan inicial. Si intencionalmente se integra heterogeneidad en el ambiente mediante planes de manejo, mosaicos de tratamientos sistémicos dentro y entre hábitats, facilitará que estos tengan respuestas múltiples a cambios impredecibles del clima y del uso de suelo.
Palabras clave: estados alternativos, mosaicos dentro de los hábitats, procesos ecológicos, restauración experimental sistémica, restauración por ensayo y error, tipología.

Restoration ecology will be a critical science of the future. Restoration aims to accelerate the transition of abandoned agricultural land back to grassland, wetland or forest, recover soil and water retention, increase carbon fixation, help recovery of biodiversity, and allow movement of species trapped in fragments due to accelerating changes in land use and climate (Chazdon, 2003; Damschen and Brudvig, 2012; see Feeley and Rehm, 2012). Bradshaw (1987) claimed the practice of ecological restoration to be the acid test of our ecological understanding. Many have embraced

his challenge, but face practical problems of small scale of experiments and no budgetary support for long-time monitoring (Michener, 1997). Assisted regeneration that creates heterogeneity and connectivity between habitat remnants is more likely to restore diversity and ecosystem services in a timely fashion than slow natural succession, plantation monocultures or trial and error restoration.

The need to maintain and re-create landscape heterogeneity and connectivity in human-dominated landscapes is evident with the reality of rapid and accelerating climate change, often accompanied by unpredictable changes in interactions among species (Vitousek *et al.*, 1997; Corlett, 2011; Blois *et al.*, 2013; Diffenbaugh and Field, 2013). Ecological restoration will be critical in species-rich tropical landscapes, so degraded by logging, forest fragmentation, and agriculture that only a small fraction of flora and fauna remain as native sources of pollen, seeds and animate colonists (Martínez-Garza and Howe, 2003; Chazdon, 2008). If a substantial proportion of biodiversity is to be saved in landscapes profoundly altered by human activities, ecological restoration will have a central role in assisting ecological succession for the remainder of this century.

Definitions and practices of restoration ecology vary. A broad definition of ecological restoration is "assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER, 2004). In practice, in the United States for example, restoration ecology occurs at many levels, ranging from re-introduction of wolves to the Rocky Mountains to re-creation of a tallgrass prairie ecosystem in Illinois (Betz *et al.*, 1997; Ripple and Beschta, 2004). Ecological restoration of an ecosystem establishes mosaics of communities in which target habitats are created that are thought to be most suited to differences in physical parameters (Jackson and Hobbs, 2009). Within a framework of habitat mosaics, it is fair to ask how best to maintain ecological heterogeneity, connectivity, genetic diversity, biodiversity, and biotic interactions.

Here, we argue that large public or commercial ecological restorations should be established as mosaics of replicated treatments within mosaics of habitats, a design that we term "systemic experimental restoration". Mosaics of replicated treatments within habitats allow populations to serve as sources of colonization for each other in highly altered landscapes (*i.e.* metapopulations, Vandeermer and Carvajal, 2001). This differs from the norm, which is standard creation of plant assemblages in large areas according to a management plan that follows the "best available practice" and the schedule for implementing it for a habitat type. A goal of trial and error restoration is to create communities comparable to one or more reference communities in landscapes with similar physical features and species pools. Alternative manifestations of a target community –for instance, different species compositions of tallgrass prairies– are not consciously created in development of management plans.

They are central to creating mosaics of treatments within mosaics of habitats. We use examples from our personal experience with experimental restoration of two species-rich ecosystems, tallgrass prairie, and tropical forests to illustrate these points.

Form and consequences of species-abundance distributions

In all but the most complete samples of the most complete communities (a patch itself is a sample of a wider community composed of local populations), species-abundance distributions are skewed in lognormal or even more attenuated distributions (Preston, 1948; Hubbell, 2001). Each manifestation of a natural or restored habitat patch has an abundance of a few species, while most are infrequent or rare. At the landscape scale, the inherent rarity of species in a habitat patch increases as the biological diversity increases, as is evident from the long tails of species-abundance distributions representing a few common species, with many infrequent or rare (Maina and Howe, 2000). The more skewed the species abundance distributions, the more likely rare and infrequent species will vary from patch to patch due to random loss of small populations or dispersal limitation. For communities as rich as tropical forest, knowledge about propagation and ideal individual size for planting are available for tree species of commercial interest (see Evans and Turnbull, 2004) or for those few species tested in restoration trials (Román-Dañobeytia *et al.*, 2012; Alvarez-Aquino and Williams-Linera, 2012; Encino-Ruiz *et al.*, 2013; Martínez-Garza *et al.*, 2013b). Rare species will be more difficult to introduce because of lack of propagation knowledge by seed or cuttings (see Castellanos-Castro and Bonfil, 2013). Extending the mosaic logic, we advocate replicated mosaics within mosaics of habitats in large-scale ecological restorations using as many species as possible, including also rare species. If patches of variable composition are established, different processes of dispersal and demographic change will take place and resilience may be reached.

A single "best available practice" for each set of physical parameters imposes stereotypes on how each community will be created, that leaves many –in highly-diverse communities most– species vulnerable to local extinction. Replicated alternative communities within a habitat type will lose rare species in skewed species-abundance distributions, but may best provide assemblages of viable and mobile populations that serve as pollen and seed sources for each other (see Menz *et al.*, 2011; McConkey *et al.*, 2012). If each habitat type within a landscape of habitats is created as an experimental mosaic, ecological restoration, the practice, becomes *systemic* experimental restoration.

Current "best available practice" may include the establishment of only one combination of a limited number of commonly used plant species. For example, in the tropics,

few pioneer trees are usually established at restoration sites (reviewed in Lamb *et al*., 2005). Changes in global climate will likely impose extreme variations in temperature and precipitation regimes (Wright, 2010). If the few species used in a restoration site happen to be negatively affected by climate change, risk of failure is bound to increase.

On the other hand, a larger number of species including higher intraspecific genetic variation in different combinations of species in systemic experimental restoration has a higher probability of buffering climate changes, depending on the response of different species to extreme climate conditions. For example, in the Tampa Bay estuary in Florida, restoration efforts involve the establishment of habitat mosaics including typical species of salt barren, salt marsh or mangrove habitats; this is seen as an option to secure habitat resiliency if sea levels rise due to climate change (Sherwood and Greening, 2014). A restoration effort including many species of different habitats allows for an ample response of this habitat mosaic to different climate changes resulting from rising sea level and severe weather along the coast.

Species loss and exchange

Restored communities are generally less biologically diverse than native communities. For instance, in central North America natural prairie wetlands of ~2 ha have a mean plant richness of 46 species, while restored prairie wetlands of similar size have, on average, 27 species (Galatowitsch and van der Valk, 1996). Containing only a subset of the landscape species pool, restorations begin as depauperate shadows of native communities and subsequently lose species because of limited area, priority effects of aggressive species and other biotic interactions (*e.g.* tallgrass prairie plantings, Howe *et al*., 2006). For small areas, local impoverishment within restored habitat patches may be unavoidable. Larger restorations offer opportunities to establish and maintain heterogeneity and biodiversity within, as well as between, habitat types.

The need is apparent for much larger experimental restorations than are commonly attempted (Cornu and Sadro, 2002; Wagner *et al*., 2008; Rodrigues *et al*., 2011). A challenge is to maximize persistence of species capable of living in or moving through highly altered landscapes, recognizing that some rare and infrequent species of species-abundance distributions will succumb to diminished demographic potential or reduced genetic variation (*e.g.* Montalvo *et al*., 1997; Kramer *et al*., 2008). Where hydrology and available land permit, we advocate pervasive integration of experimental methods of restoration ecology, the science, with ecological restoration, the management practice. Some alternative communities (*e.g.* grass-dominated or dicot-dominated prairie, Figure 1) may quickly converge in species composition and species-abundance distributions. For most, we predict, systemic experimental restoration will create viable diversity within, as well as between habitat types. In this way, systemic restoration will help maintain landscape heteroge-
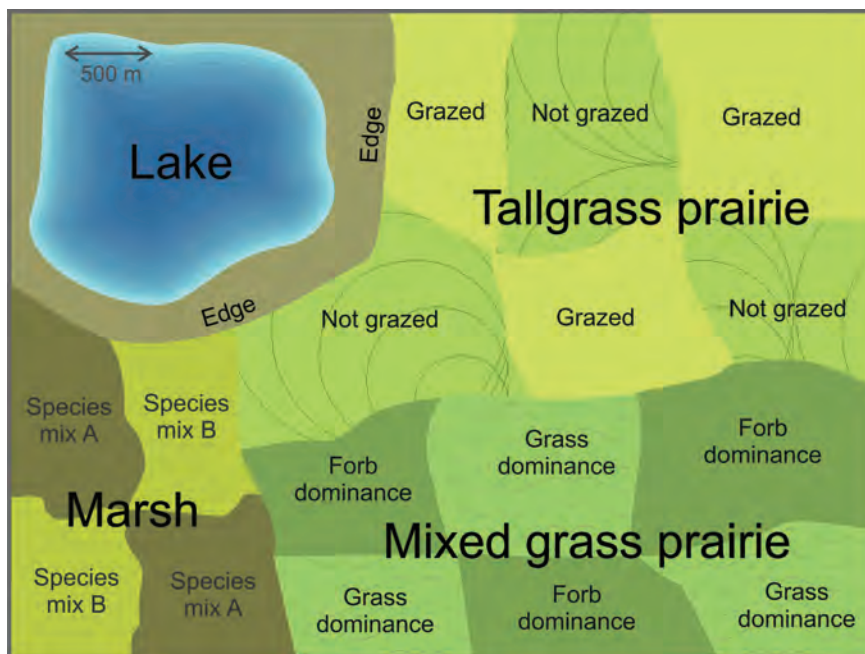

Figure 1. Hypothetical restoration of a large public area in an agricultural landscape with a systemic experimental design. Three major habitats determined by physical conditions of drainage and soil characteristics (marsh, tallgrass prairie, mixed grass prairie) are of sufficient area to accommodate replicated management regimes, as shown. Of the total area of 35 km², ~82% is planted in one of three replicated designs, virtually all of the area not in a 4.5 km² lake or lake-edge habitat. Marinés de la Peña-Domene created the figure.

neity, will provide pollen and seed sources to neighboring restoration treatments, and will provide a mixed of species as sources of colonists that respond to future changes in land use and climate. Methodical protocols of public and private management plans inevitably creates accidental variation. Systemic experimental restoration would consciously establish diverse treatment mosaics within and between habitat types, thereby building variation into community assembly at the outset.

Are small restorations pointless? The answer is a qualified "no." Small restorations offer refuges and connectivity for subsets of mobile animals and plants, if not for the entire flora and fauna, by increasing the heterogeneity and quality of the matrix between large remnants (Vandermeer and Carvajal, 2001). In the tropics, small habitat patches maintain landscape heterogeneity by hosting somewhat different plant and animal assemblages in different islands, by creating temporary refuges or feeding sites for mobile animals, and by facilitating pollen flow and immigration of animals and seeds that are capable of crossing alien agricultural habitats (Turner and Corlett, 1996; Arroyo-Rodríguez *et al*., 2009). Small restorations broaden effective population sizes of species capable of exchanging pollen or immigrants over inhospitable matrices between remnants (Benayas *et al*., 2008; Kramer *et al*., 2008). Moreover, tropical cropland and pasture are often settled and abandoned over cycles of a few years to a few decades. Small remnants or forest patches created to increase landscape heterogeneity becomes nuclei from which succession spreads when land is abandoned (Holl *et al*., 2011). Alternative planting regimes will doubtless favor more heterogeneity than any one-method-fits-all approach could, in small restorations as in large ones. For mobile species, replicated variations of stepping-stone or corridor habitats creates mosaics of habitats within agricultural landscapes, and hedge against slow succession when land is abandoned.

Limitations of trial and error

A need for a new epistemology is apparent from experience. The challenge that restoration ecology could provide "acid tests" of ecological theory has been frequently met in small-scale academic studies, but seldom in large government or commercial restorations (see Jordan *et al*., 1987). One challenge is quality of inference. Definitive tests do not occur in unreplicated restorations because comparison of habitats created in different places at different times is associational; strong inference requires experimental control of variables (Platt, 1964). Also, objectives for restoration are usually conditioned as much by history, law, aesthetics, and cost, as by scientific hypotheses (Gobster *et al*., 2007; Shackelford *et al*., 2013). Unless created with an *a priori* design for analysis, much of the ability to determine reasons why developing communities take the trajectories they do is lost.

A second and particularly important challenge is that the field itself is dominated by typological thinking. Policymakers may assume that restoration already rests upon sound ecological principles that lead to well-defined ecological results. Even though gigantic advances in restoration ecology have taken place in isolated instances (Shackelford *et al*., 2013), the state of ecology as a science and the dramatic and rapid changes in land use and climate that alter the conditions under which flora and fauna interact obligate us to be cautious (Table 1 of Young *et al*., 2005; Blois *et al*., 2013). Using ecological theory to guide restoration is not equivalent to use of Newtonian mechanics to build an engine. Ecological communities are by comparison unbounded and characterized by infinite variation in composition and structure.

A third challenge is that ecosystems are dynamic on many time scales. Basic changes in composition and function occur on scales of decades to millennia, and now include the birth of novel ecosystems (see Hobbs *et al*., 2013). A risk is that large publically-funded restorations amount to expensive, sometimes irreversible one-treatment manipulations that rest on weak conceptual foundations, that target geologically ephemeral communities, and that do not pinpoint causes of success or failure.

Limitations of "best available practice" based on changing views of a habitat might be illustrated with a major prairie restoration in northern Illinois. Beginning in 1975 and continuing every few years since, areas of ~4-34 ha each of tallgrass prairie vegetation have been planted at the Fermi National Accelerator (Betz *et al*., 1997). Managers recognized monopolizing potential of dominant grasses in early plantings by later increasing representation of early and mid-successional species, changing burn time, and changing burn frequency. This adaptive management used trial and error to encourage diversity within plantings by limiting dominance of large C_4 grasses. Chronosequences established at different times consciously, as well as coincidently, created differences in species composition between plantings.

Fermi chronosequences yielded important insights. Plantings quickly improved soil texture (Jastrow, 1996) and allowed estimates of the years to decades required to restore carbon and nitrogen stocks in soil and plant tissues (Lane and BassiriRad, 2002; Matamala *et al*., 2008). The effort also provided clear evidence that chronosequences failed to maintain biodiversity when dominant species (*e.g.* big bluestem, *Andropogon gerardii*) crowded out others (Sluis, 2002; also see Emery and Gross, 2006). Moreover, chronosequences demonstrated that increasing dominance of C_4 grasses resulted in a decrease of patchiness in availability of soil resources in comparison with much more varied native prairie remnants, further favoring the dominance of only one species (Lane and BassiriRad, 2005). Another discovery using Fermi, soils from a restoration 28 years

earlier, in combination with those from other sites, showed that mycorrhizal associations of at least one dominant C_4 grass (*A. gerardii*) adapt to local soils (Johnson *et al*., 2010). Contributions of Fermi efforts are indisputable; native vegetation was established where agriculture had profoundly altered the landscape, and decades of discovery yielded fundamental insights into prairie recovery. Could replicated alternatives within chronosequences to have yielded more?

In hindsight, chronosequences without replication expose lost-opportunity costs. By comparison, well-replicated Park Grass experiments *definitively* show that fertilization increases productivity in English meadows at the expense of diversity because aggressive species monopolize resources and exclude competitors (Crawley *et al*., 2005). Additional studies that do not make explicit use of planting designs also add insights. For instance, Kunin (1998) rejects great importance of "mass effects" of invasive species from edges on community composition, in this case allowing the lack of an effect to be assessed under quite different management regimes. At Fermi, absence of simultaneous planned replication prevents a definitive determination of how a variety of factors influence tallgrass development, or how to do it better.

One might argue that prairie restoration practices of 1975 are now irrelevant. That exposes an important point. If a particular "best available practice" for restoration of tallgrass prairie in 1975 is now obsolete, "best available practice" in the early 21st Century may well be obsolete in 40, or 400 years. The question is not whether valuable things have been learned but whether much more could have been learned from planned contrasts within chronosequences.

A more effective hedge against failure in a largely unpredictable world would be to establish plausibly realistic alternative manifestations of communities (*e.g.* dicot- *vs* grass-dominated prairie, Figure 1). The goal would add intentional variation to accidental variation, establishing diversity of assemblages within units of a habitat, as well as between habitats (for measures, see Magurran and McGill, 2011). Plausibly realistic variation within and between restorations may be the best way to hedge bets against invasive species, land-use change, loss of pollinators or dispersal agents, and climate change over for many, albeit not all, species in many human-dominated landscapes in the near and distant future.

Troubled future for community stereotypes

Ecological restoration as a management practice reconstructs approximations to reference communities or historical images (see for example Beas *et al*., 2013), which may be risky. Reference communities, analog communities because they are contemporary and known, are imperfect reflections of non-analog communities of the past or predictors of non-analog communities of the future (Davis, 1986; Millar and Brubaker, 2006). One step forward has been to propose the use of at least two reference communities (Ruiz-Jaen and Aide, 2005). However, defining reference communities is particularly difficult if almost all of a biome is gone, as is often the case. For instance, < 1% of original North American tallgrass prairie remains east of the Mississippi River (Noss *et al*., 1995). In many other cases, evidence exists that habitats, such as ancient Mesoamerican forests occupied by Mayan civilizations, have been managed for millennia (see Gómez-Pompa and Kaus, 1999; Robinson and McKillop, 2013). Selecting reference communities under varying interactions of physical environment, prehistoric human intervention, climate, and landscape species pools –or that might occur in the future– is not a simple matter, and often may not be realistic.

Typological thinking is a recurring liability in restoration ecology. Untenable in modern ecology, the idea of self-replacing climax communities that reflect fixed end points of community development is alive and well in *policy* objectives governing habitat restoration. Policies establishing reference communities for restoration do not reflect the view that species assemblages exist in alternative natural states (Beisner *et al*., 2003; Cortina, *et al*., 2006; Suding, 2011), that native communities are dynamic on small to continental scales (Davis, 1986; Polley *et al*., 2005; Questad and Foster, 2008), that species-rich assemblages develop by interaction of chance and adjustments of species to physical environmental factors and each other (Comita *et al*., 2010; Blois *et al*., 2013), and that in biological communities some species are common, more are infrequent, and many are rare (Mac Arthur, 1972; Maina and Howe, 2000). Even enlightened approaches that create restorations within an existing *range* of reference communities, perhaps biomass components of forest production or nutrient recycling in wetlands as criteria (*e.g.* Brinson and Rheinhardt, 1996), would likely improve if replicated variation were built into restoration designs.

A common assumption in restoration policy is that managed succession proceeds in a linear fashion, with predictable, incremental assembly of species over time. In reality, a variety of rates and trajectories of succession are possible and often likely (Zedler and Callaway, 1999; Suding, 2011), and in the future will be virtually certain (Millar and Brubaker, 2006). Unexpected events or interactions in ecological communities may create thresholds that shift succession in quite different directions than those envisioned at the start. An instructive example of unintended, in this case beneficial, consequences is re-introduction of wolves (*Canis lupus*) to the Rocky Mountains in North America (Ripple and Beschta, 2004; Wolf *et al*., 2007). Willows (*Salix* spp.) along streams in and near Yellowstone National Park in the absence of wolves are heavily browsed by elk (*Cervus elaphus*). Elk foraging choices reduce food and dam materials for beaver (*Castor canadensis*), which become scarce in the landscape. Without beaver dams, ponds do not form. Reintroduction of wolves that hunt from willow stands increases

risk of predation for elk, which then forage elsewhere. The result is expansion of willow thickets, restoration of beaver food supplies, and establishment of dams that create pond edge and wetland habitats. In this case a trophic cascade alters hydrology, plant habitat, animal habitat, animal behavior, and the nature of montane landscapes. Replicated treatments of the sort we recommend for extensive plantings are infeasible in this example of recovery of a population of a top predator, but the case illustrates an unexpected benefit of restoration of a single top predator. It is not yet clear how common such positive shifts are in community restorations.

By their nature, experiments have unpredictable but often enlightening outcomes. Uncertain consequences involve risks, but they are integral to the experimental process. For instance, planting mixed stands of animal- as contrasted with wind-dispersed trees in tropical landscapes may initially, produce roughly similar cover, carbon sequestration, soil formation, and water retention (Tobón *et al.*, 2011). Later, higher productivity in stands of wind-dispersed species was a surprise (Valencia-Esquivel, 2012). Planted stands of tree species greatly preferred by fruit-eating animals could dramatically alter the input of animal-borne forest seeds (*e.g.* Howe, 1984; Martínez-Garza and Howe, 2003). Alternatively, quite different plantings might converge in species composition if wide-ranging birds and bats bring similar seeds to both kinds of synthetic communities, that is, if composition is limited by establishment priorities or microenvironments rather than dispersal. In one such experiment, the jury on convergence or divergence will be out until trees mature (de la Peña-Domene *et al.*, 2013). Even if planting schemes ultimately converge in this or other experimental restorations, treatments will sustain quite different species compositions and species-abundance distributions for decades.

Restoration mosaics within mosaics should hedge against climate change. On first principles, variability by design should be more conducive to evolutionary and community adaptation to environmental change than management that struggles to maintain a disappearing *status quo* (see Davis and Shaw, 2001; Ashley *et al.*, 2003; Millar and Brubaker, 2006). In practice, this is the best way to create communities capable of responding to regional or global changes in land use or climate (Sherwood and Greening, 2014). Consciously, including variation within restoration designs should ultimately improve management results and hedge bets on future change for restorations intended to last decades or longer.

Issues of scale

A risk is that restoration efforts without replication amount to one-time anecdotes without clear relevance to other places or times (*e.g.* Caughley, 1994). These are local successes or failures. Embedded small-scale experiments add value by addressing basic processes that can be applied elsewhere

(Giardina *et al.*, 2007). For instance, Zedler (1996) discusses major wetland restorations and mitigations using trial and error approaches. For most, failure is final. Limited nitrogen amendments within a larger wetland restoration in San Diego Bay test means of creating tall cordgrass (*Spartina foliosa*) plots as sites for rail (*Rallus longirostris*) nesting, compared with unfertilized treatments less likely to attract the birds. Embedded experiment makes the bird component a permanent asset, whatever the fate of the birds - or for that matter of San Diego Bay. Embedded experiments are not attempted on scales that apply easily to landscape or ecosystem scales, nor do they create substantial variety of species compositions and abundance distributions that would allow restorations to respond in multiple ways to climate change.

Systemic experimental restoration on a large scale needs forethought. The ~1160 page management plan and associated documents for the Midewin National Tallgrass Prairie in Illinois is a design that schedules conversion of ~6,000 ha of farm and pasture to prairie (http://www.fs.usda.gov/main/midewin/home, accessed 13 July, 2013). The US Forest Service is in charge, with input from other federal, state, and local agencies, and non-governmental organizations. The immense scale of the plan developed between 1996 and 2002 and amended since, means that radical changes would require an enormous administrative effort. Systemic replicated restoration in such large public efforts would best be included early in the process of developing a management plan Nonetheless, the existing Midewin plan could be implemented as systemic contrasts. The public might expect plantings of tall native prairie vegetation, but the Forest Service recognizes that such vanishing birds as the upland sandpiper (*Bartramia longicauda*) require short grasses best provided by grazed pastures of alien forage grasses, while the bobolink (*Dolichonyx oryzivorus*) nests in somewhat taller pasture vegetation. Neither thrives in a sea of two-meter high bluestem (*Andropogon gerardii*). The management plan calls for a mosaic of short, somewhat taller and tall prairie vegetation. In a world without grazing lawns with neglected patches left by native bison and elk, cattle and mowing maintain heterogeneity in grass heights suitable for threatened wildlife. At Midewin, transition to a systemic plan would only require attention to legitimate replication in the existing Prairie Plan (Figure 3 of The Plan) to optimize statistical evaluation.

Challenges of systemic experimental restoration

Synthetic plant communities should be less sacred than relict habitats, but there is little evidence to suggest it. Manipulation does not prevent contrasting fire or grazing regimes in natural areas (*e.g.* Collins and Smith, 2006). Moreover, academic journals and anthologies are replete with manipulative small-scale restoration experiments, what we call "restoration boutiques", including some by the authors

(Howe *et al.*, 2006; Martínez-Garza *et al.*, 2011, 2013a,b) and important attempts to critically examine the contribution of experiments to the practice of restoration (*e.g.* Cabin, 2011). Small experiments allow tests of core hypotheses at limited spatial and temporal scales, as well as additional contributions not planned at the outset (Zorn-Arnold and Howe, 2007). Lacking are systemic replicated contrasts built into most of the areas of large public restorations.

Some resistance to systemic experimental restoration is practical. It is easier to create experimental differences in species composition, for instance, than hydrological regimes. However, even if only two options can be managed in stream restoration, for instance, contrasts produce enlightening differences in productivity and species diversity (Mitsch *et al.*, 2012). It has to be acknowledged that in other cases a systemic approach is unlikely to work. For example, in control of invasive salt cedar dispersed along rivers (*Tamarix* spp., *e.g.* Shafroth *et al.*, 2008), alternative treatments in a systemic experimental design would result in better control in some places than others, and no suppression of the invasive in control replicates. Linear dispersal of salt cedar along rivers probably makes methodical adaptive management, such as that reported by Shafroth and colleagues, safer than systemic experimental restoration.

More generally, experiment might be perceived as unsuitable for mitigation purposes because less successful treatments could be seen as failures. For many systems, however, the irony remains that a "best-available practice" from a manager's perspective could be viewed as successful, even if *untried* treatments would have yielded better results over either the short or long term. Multiple manifestations of *processes* as "best available practices" will often be more practical than unmanipulated controls. For instance, tree plantings that favor different processes of seed dispersal might work in tropical landscapes with high biodiversity, where most trees require animals for successful reproduction (*e.g.* Muller-Landau and Hardesty, 2005; Vamosi *et al.*, 2006). Logistics, cost, and public values may limit what is feasible to simple replicated contrasts in habitat creation or subsequent management. Even replicated contrasts of alternatives without controls, would represent a quantum advance over large areas with only one treatment.

A challenge in systemic experimental restoration will be choice of the size and number of replicates. Species-abundance distributions contain some species that are rare but viable under a given set of conditions, and other species that are accidental or otherwise inviable under the same conditions (Maina and Howe, 2000; see Shoo *et al.*, 2005). The challenge is to create replicates large enough to harbor viable populations of those species and functional groups most likely to survive in a human-dominated landscape, while guarding against loss of species that reproduce and sustain themselves at low densities. For instance, in prairie communities with approximately 150 plant species (*e.g.* Figure 1 of Wilsey *et*

al., 2005), many persist in densities of a few to hundreds of individuals per hectare. Alternative planting designs such as those illustrated using most of the likely native species (Figure 1) would build in β diversity, thereby preserving much of the native diversity in herbaceous vegetation. For tropical rainforest in which most tree species occur in densities < 1 ha^{-1} (*e.g.* Pitman *et al.*, 2001), scale of the illustrated replicates would preserve a far smaller proportion of the woody flora even if most species could be planted.

Another challenge is bureaucratic. Governmental and Non Governmental Organizations support to restoration is crucial to development of the field. However, it is often the case that experimentation is explicitly forbidden, perhaps assuming that methods and assumptions underlying restoration practice are beyond question (for instance in Mexico, Comisión Nacional para el Conocimiento y Uso de la Biodiversidad and Fondo Mexicano para la Conservación). *De facto* policy implementation produces *the same result* in the United States. Legal requirements to mitigate habitat loss or to re-create stereotypes of history codify typology in laws and in the regulations that are derived from them. Because sponsoring organizations often do not support long-term monitoring, quantification of success or failure is not straightforward, further limiting effective adaptive management.

Conclusions

Simple contrasts replicated two to several times throughout major public and private restorations would further revolutionize ecological restoration. Widespread trial and error restoration without replication limits biodiversity within habitats, provides only weak inference of reasons for success and failure, and limits hedges against changes in climate and land use. Systemic use of replicated alternative versions of ecological communities effectively makes large-scale public or commercial restorations experiments on an ecosystem scale. These would promote plausible alternative states (*e.g.* dominance relations and species compositions) within, as well as between habitats, admit the possibility of determining the reasons why restorations succeed or fail, create mosaics of communities within and between habitats, and provide fertile ground for follow-up studies. Practical challenges will involve deciding what treatments produce and maintain clear and plausibly realistic alternatives, maintaining contrasts, and engaging researchers and managers who use contrasts for their own purposes as communities evolve. Foresight and simplicity facilitate the first two; contrasting ecological communities of known histories will attract the latter.

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