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Ecological restoration for future sustainability in a changing environment[1]

Young D. CHOI[2], Department of Biological Sciences, Purdue University Calumet, 2200 169th St., Hammond, Indiana 46323-2094, USA, e-mail: ydchoi@calumet.purdue.edu
Vicky M. TEMPERTON, Julich Research Centre – Phytosphere Institute, ICG-3, Germany.
Edith B. ALLEN, University of California – Riverside, 900 University Avenue, Riverside, California 92521, USA.
Albert P. GROOTJANS, University of Groningen, P.O. Box 72, 9700 AB Groningen, The Netherlands.
Melinda HALASSY, Institute of Ecology and Botany – Hungarian Academy of Science, H-2163 Vácrátót, Alkotmány u. 2-4, Hungary.
Richard J. HOBBS, Murdoch University, 90 South Street, Murdoch, W. Australia 6150, Australia.
M. Anne NAETH, University of Alberta, 114 St.– 89th Avenue, Edmonton, Alberta T6G 2E1, Canada.
Katalin TOROK, Institute of Ecology and Botany – Hungarian Academy of Science, H-2163 Vácrátót, Alkotmány u. 2-4, Hungary.

Abstract: Since its emergence in the past decades, restoration ecology has demonstrated an astounding growth as a new discipline of applied science. At the same time, this young discipline has been criticized for its retrospective goals largely based on the past, its fragmented approach, and its idealistic goals, which do not relate to the real world context. Restoration with past-focused, idealistic, and/or *ad hoc* goals may not work in the future because an ecosystem that is restored for the past environment is not likely to be sustainable in the changing environment of the future, simple recomposition of isolated and fragmented naturalistic patches is not likely to restore ecosystem functions, and unrealistic goals and work plans are not likely to gain public support. We advocate directing the principles and practice of ecological restoration to the future. Future-aimed restoration should acknowledge the changing and unpredictable environment of the future, assume the dynamic nature of ecological communities with multiple trajectories, and connect landscape elements for improving ecosystem functions and structures. In this paper, we discuss the predictability of restoration trajectories under changing environmental conditions, the application of ecological theories to restoration practice, the importance of interdisciplinary approaches and human interventions in ecosystem recovery, and the social context of ecological restoration.
Keywords: ecology, environment, future, restoration, sustainability.

Résumé : Depuis son émergence dans les dernières décennies, la restauration écologique a démontré une croissance phénoménale en tant que nouvelle discipline scientifique appliquée. En même temps, cette jeune discipline a été critiquée pour ses objectifs rétrospectifs, son approche fragmentaire et ses idéaux qui ne sont pas toujours réalistes. Il est fort possible qu'une restauration orientée vers le passé, avec des objectifs idéalistes et/ou *ad hoc* ne sera pas fonctionnelle dans l'avenir. En effet, un écosystème restauré en fonction d'un environnement passé ne sera peut être pas viable dans un futur en changement, la réhabilitation vers un aspect naturel de parcelles isolées et fragmentées ne restaurera probablement pas les fonctions de l'écosystème et des objectifs et plans de travail irréalistes ont peu de chance d'obtenir la faveur du public. Nous recommandons d'orienter les principes et la pratique de la restauration écologique vers le futur. Cette restauration tournée vers l'avenir devrait prendre en compte que les environnements futurs seront changeants et imprévisibles, considérer la nature dynamique des communautés écologiques ayant des trajectoires multiples et assurer la connectivité des éléments du paysage pour améliorer les fonctions et structures des écosystèmes. Dans cet article, nous discutons de la prévisibilité des trajectoires de restauration dans des conditions environnementales changeantes, de l'application des théories écologiques à la pratique de la restauration, de l'importance de l'approche multidisciplinaire et des interventions humaines pour la réhabilitation des écosystèmes et finalement, du contexte social de la restauration écologique.
Mots-clés : écologie, environnement, futur, restauration, viabilité.

Nomenclature: Throughout, latin binomials are those used by the original authors.

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[2]Author for correspondence.

Introduction

Since its emergence over recent decades, the discipline of restoration ecology has demonstrated an astounding growth. It has been regarded as a proactive tool for conservation and management of biological resources, a "testing laboratory" of ecological theories (Bradshaw, 1983; 1987; 2002; Jordan, Peters & Allen, 1987; Hobbs & Norton, 1996; Choi, 2004; Temperton *et al*., 2004; Harris *et al*., 2006), and even a hope for the future (Dobson, Bradshaw & Baker, 1997). As expected, this young discipline (Palmer, Ambrose & Poff, 1997) is experiencing "growing pains", with strong criticisms. Major points of criticism, mainly from outside the restoration professional community, include subjectivity in determining restoration goals, inapplicability of a static approach to dynamic ecosystems, and impracticality of a retrospective approach because of irreplaceable losses and/or irreversible changes (Davis, 2000; Davis & Slobodkin, 2004). A constructive examination of, and response to these criticisms is necessary if the discipline of restoration ecology is to mature fully.

At least 3 major concerns, although they are not universal, are often raised within the field of ecological restoration.

First, its contemporary paradigm is largely retrospective and thus focused on the past. Contemporary restoration practices often aim to rebuild ecosystems or habitats that once existed in a past environment. Historical information is undoubtedly a valid resource for guiding future restoration. However, the environmental conditions of the future will very likely be different from the past. Therefore, a restored ecosystem rigidly aimed at historical fidelity may not be sustainable in the future (Pavlik, 1996; Choi, 2004; Eagan & Howell, 2005; Harris *et al*., 2006; Choi, 2007).

Second, the discipline of restoration ecology has largely progressed on an *ad hoc*, site- and situation-specific basis (Hobbs & Norton, 1996). Allen and Hoekstra (1992) lamented that ecological restoration has often been "a sort of gardening with wild species", at least in the earlier times. "Ecological restoration" is based on the principle that the restored site should be self-sustaining, often with no or very little further augmentation of energy or materials by humans (Jackson *et al.*, 1995; Ehrenfeld & Toth, 1997). A fragmented restoration approach (*e.g*., "gardening" for recomposition of past flora solely based on botanical interests) is not sufficient to meet this principle (Hobbs & Norton, 1996; Choi, 2007). Numerous restoration attempts have been made at landscape or ecosystem levels, but economical, social, and political constraints often limit the success of our restoration efforts and produce fragmented results. Therefore, a multidisciplinary approach is essential for successful restoration (Majer & Recher, 1994; Hobbs & Norton, 1996; Hobbs, 2004; Halle, 2007).

Third, idealistic restoration goals are often not possible under prevailing economical, social, and political circumstances. Michener (1997) noted that ecological restoration until recently has been viewed as more of an "art" than a "science", often relying upon intuition rather than a documented knowledge base. For this reason, restoration professionals have often not been fully prepared to present a unified package of goals, feasible work plans, and societal benefits of ecological restoration to the public (Hobbs & Norton, 1996; Higgs, 1997; Hobbs, 2004; Throop, 2004). Idealistic restoration may lead to a public perception that ecological restoration is "an expensive self-indulgence for the upper classes" that does not acknowledge the real-world context (Kirby, 1994). Large-scale restorations with landscape or ecosystem approaches, in particular, can often be "idealistic", inflaming public sentiments and resistance if they negatively interfere with economic, social, and political interests. A "realistic" restoration goal has to be ecologically sound, economically feasible, and socially acceptable (Choi, 2004; 2007; Halle, 2007; Hobbs, 2007). The need to provide "realistic" and thus achievable goals in restoration has been clearly advocated and is being increasingly recognized by restoration professionals (Hobbs & Norton, 1996; Michener, 1997; White & Walker, 1997; Hobbs, 2004; 2007).

In this paper, we advocate the direction (or redirection) of contemporary principles and practices of ecological restoration toward the future. Future-aimed restoration should acknowledge the changing and unpredictable environment of the future, assume the dynamic nature of ecological communities with multiple goals and trajectories, connect landscape elements for reinstating both ecosystem structures and functions, and seek public support for setting realistic restoration goals and scopes (Hobbs & Norton, 1996; Ehrenfeld & Toth, 1997; Michener, 1997; Palmer, Ambrose & Poff, 1997; White & Walker, 1997; Choi, 2004; Harris *et al*., 2006; Choi, 2007; Hobbs, 2007). This review paper is an attempt to synthesize a conceptual basis for future-aimed restoration within practical limitations. To this end, we discuss the following themes: (1) predictability of restoration trajectories under changing environmental conditions, (2) application of succession theories to restoration trajectories, (3) a special need for interdisciplinary approaches, (4) human interventions for ecosystem recovery, and (5) ecological restoration in social context.

Predictability of restoration trajectories under changing environmental conditions

The Earth's atmospheric temperature has increased by 0.74 °C in the last 100 y and is projected to increase by 1.1 to 6.4 °C during the 21st century (IPCC, 2007). This rising atmospheric temperature may cause weather patterns to become less predictable, with an increased frequency of extreme meteorological events. Such changes in weather patterns will likely affect restoration outcomes and make it difficult to set goals (Harris *et al*., 2006).

For example, intensified precipitation in the 1990s as a part of local climate change is having strong effects on vegetation composition in the coastal dunes of the Netherlands (Verbeek *et al.*, 2003), thereby frustrating efforts to restore fen vegetation in dune slacks (called swales in North America). Physical removal of sods was successful in both 1954 and 1986 within the context of restoration project. The target pioneer vegetation, with many orchids and rare sedges, reappeared after a few years, and after 40 y some orchids were still present in the vegetation (Bekker *et al.*, 1999). However, recent restoration attempts using the same method in 1990 and 1995, the years with heavy rainfall, led to outcomes different from those observed in 1954 and 1986. Most of the target species did not appear or disappeared after a short appearance, and the vegetation became an assemblage of species that are adapted to more wet and eutrophic conditions (Figure 1). Vegetation in some wet plots did not change at all during the 12 y of observation (Grootjans *et al*., 2002a,b). A similar case was reported by Van Duren *et al.* (1998) after a failed restoration of fen meadow in a polder area in the Netherlands. Thus, there is need for a new flexibility, given that the restoration technique that worked well in the past (sod cutting) may no longer be suitable under the changing climate.

The Earth's soil and water are continually enriched by atmospheric depositions of nitrogen (Galloway *et al*., 2004) and other nutrients. In the Netherlands, atmospheric deposition of nitrogen has risen significantly from around 10 kg·ha^{-1}·y^{-1} in 1930 to 35 kg·ha^{-1}·y^{-1} in the late 1990s (Stuyfzand, 1993; Ten Harkel & Van der Meulen, 1996; Van Wijnen, 1999). Such increased deposition has likely triggered grass and shrub encroachment in large parts of

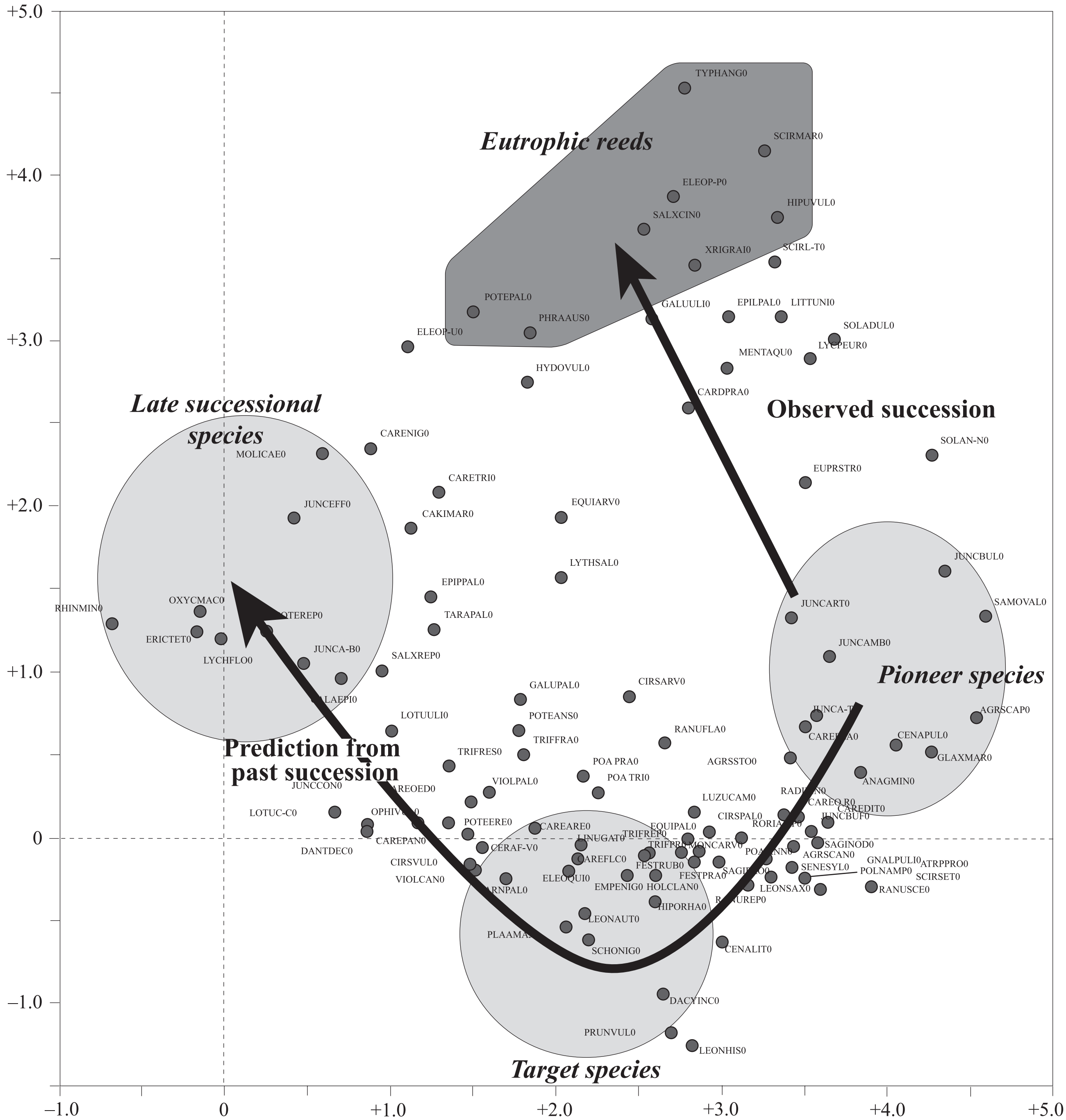

FIGURE 1. DCA of 30 permanent plots. Predicted and observed paths of succession after restoration effort in coastal dune slack wetlands in the Netherlands during the1980s and 1990s. The target community was modeled after an historical community that occurred in the 1920s and in 1956; however, the restoration trajectory did not arrive at the target community, likely as a result of intensified precipitation.

the dry dune areas (Kooijman & Besse, 2002). Simulating natural wind and water processes that may shape new dunes and interdunal wetlands is a new challenge for restoring the natural dune landscapes of the Netherlands (Grootjans *et al.*, 2002a). Nitrogen enrichment is also considered to be a major obstacle in restoration of coastal sage scrub (CSS) vegetation in southern California, USA (Minnich & Dezzani, 1998; Allen *et al*., 2000; Fenn *et al*., 2003). CSS is a semi-deciduous shrubland dominated by ≈1-m-tall shrubs, such as California sagebrush (*Artemisia californica*), California buckwheat (*Eriogonum fasciculatum*), and brittlebush (*Encelia farinosa*), with a diverse understory of native forbs, mainly annuals. Anthropogenic nitrogen deposition, primarily from automobile emissions, is among the major reasons for the relatively recent type conversion of CSS during the past 50 y in areas of southern California. Up to 30 kg $N \cdot ha^{-1} \cdot y^{-1}$ have been measured in western Riverside County (Bytnerowicz, Miller & Olszyk, 1987), where the most rapid losses are occurring (Minnich & Dezzani, 1998). Observations along an urban-to-rural nitrogen deposition gradient showed a loss of diversity of native forb species from 67 to 16 $species \cdot ha^{-1}$ (Figure 2). The loss of CSS, coupled with urban development, has prompted extensive efforts to restore this vegetation type (Allen *et al*., 2000; Cione, Padgett & Allen, 2002; Allen *et al.*, 2005). However, restoration outcomes have been mixed because invasive species are forming a new stable state dominated by exotic grasses. In the midwestern USA, Wilcox, Chun, and Choi (2005) also reported a rapid accumulation of nitrogen in soils of developing black oak (*Quercus velutina*) savanna in sand dunes of Lake Michigan. These rapid accumulations are likely due to atmospheric deposition, and such nitrogen enrichment could add uncertainty to the future trajectory for vegetation development of black oak savanna.

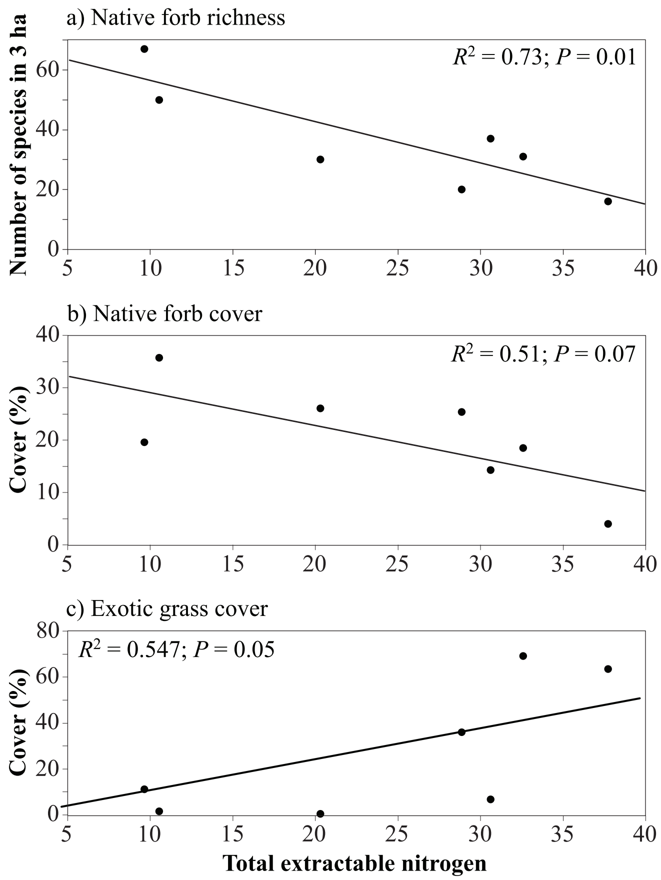

FIGURE 2. Richness of native forbs in 3-ha plots (a), percent cover of native forbs (b), and percent cover of exotic grass (c) along an anthropogenic nitrogen deposition gradient in coastal sage scrub vegetation, western Riverside County, California, in April 2003. Extractable soil N is nitrate-N plus ammonium-N. Native forbs, both annual and perennial, are the main contributors to diversity of this vegetation type.

Nitrogen enrichment of soil, along with other factors such as climate change, is often linked to invasions of exotic species (Hobbs & Mooney, 2005). Nitrogen enrichment may promote species invasions to the detriment of native species. In southern California, cover of exotic grasses increased in nitrogen-enriched soil (mostly red brome [*Bromus rubens*] and wild oats [*Avena barbata*]) (Figure 2). Eutrophication has also been cited as a probable cause for expansions of exotic species in grasslands (Wilson & Tilman, 1995), wetlands (Galatowitsch, Anderson & Asher, 1999; Choi & Bury, 2003), and woodlands in the Great Lakes states of the USA (Wilcox, Chun & Choi, 2005). The combination of high soil nitrogen, high exotic grass production, and invasive species presents a major challenge to restoration efforts in the CSS (Allen *et al*., 2000). Such invasive exotic species often form a stable state (Hobbs & Norton, 1996; Temperton & Hobbs, 2004) of monospecific stands of exotic species and inhibit (Connell & Slatyer, 1977) or divert the desired restoration trajectory.

Levin (1989) characterized modern ecology as "uncertainty and variability". These characteristics will likely be intensified in our future environment, making the need for a shift in restoration approach from "historic" to "futuristic" greater than ever (Choi, 2004). Earlier definitions of ecological restoration (*e.g.*, Jordan, Peters & Allen, 1987; National Research Council, 1992; Society for Ecological Restoration International as cited by Aronson *et al*., 1993; Jackson, Lopoukhine & Hillyard, 1995) clearly set the goals of restoring the natural, historic, or prehistoric ecological communities and ecosystems that preceded disturbance by human activities. Cairns (2002) noted that ecosystem restoration is possible when climate conditions suit the species that once inhabited the area. However, the trends of global climate change are unlikely to reverse in the foreseeable future. Atmospheric deposition of nitrogen will likely increase along with the rising demand for fossil fuels and nitrogen fertilizers (Galloway *et al*., 2004). Current rates of invasions by exotic species are faster than ever (Lonsdale, 1999; Mack, 2005; Moony, 2005). While the past is undoubtedly a valuable guide for projecting restoration outcome (Higgs, 2003; Eagan & Howell, 2005), it should not be a straightjacket. Harris *et al*. (2006) warned that "valuing the past when the past is not an accurate indicator for the future may fulfill a nostalgic need but may ultimately be counterproductive in achieving realistic and lasting restoration outcomes." With this perspective, the Society for Ecological Restoration International (SERI, 2004) issued a new definition of ecological restoration: "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed." This definition has major implications for human interventions, suggesting that the recovery of ecosystem functions and processes, rather than reassemblage of past floras or faunas, should be the goal.

Application of succession theories to restoration trajectories

Ecological theories have provided conceptual frameworks for projecting future restoration outcomes (Bradshaw, 1983; Dobson, Bradshaw & Baker, 1997; Naeem & Li, 1997; Palmer, Ambrose & Poff, 1997; Parker, 1997; Wali, 1999; Naeem, 2002; Walker & del Moral, 2003; Choi, 2004; Temperton & Hobbs, 2004; Falk, Palmer & Zedler, 2006; Naeem, 2006; Walker, Walker & Hobbs; 2007). In particular, successional models have been applied to restoration trajectories (Walker, Walker & Hobbs, 2007). For example, MacMahon (1987) suggested that Clements' (1916) 6 steps of plant community development (from nudation to climax) were applicable to the restoration of major terrestrial biomes. However, this model was found inadequate for restoration trajectories (Cairns & Heckman, 1996; Hobbs & Norton, 1996; Palmer, Ambrose & Poff, 1997; Parker, 1997; Wali, 1999; Choi, 2004) because of its deterministic nature. Ecological succession is not typically deterministic; it is stochastic, at best only generally directional and often reticulate, regressive, or cyclic (Fekete, 1992; Walker & del Moral, 2003). Therefore, ecosystem processes do not necessarily undergo an ordered development toward a single end point but often undergo rapid transitions between different metastable states toward multiple end points (Hobbs & Norton, 1996; Suding, Gross & Houseman, 2003; Suding & Gross, 2006), which are often not predictable (Temperton & Hobbs, 2004).

In North America, Zedler (1996) and Zedler and Callaway (1999) did not find any clear changes toward a determined target community in wetland restoration sites in southern California. Simenstad and Thom (1996) reported that the target community of a restored estuary in the Pacific Northwest was not attainable, and Shear, Lent, and Fraver (1996) warned that a restored bottomland forest in Kentucky would not sustain its species composition in the long term. Wilcox *et al*. (2005) reported that the development of black oak (*Quercus velutina*) savanna in disturbed sand dunes of Lake Michigan did not necessarily move along a single path. In Europe, Bartha (2002) and Halassy, Torok, and Marko (2005) found reticular and multiple paths of vegetation development after removal of alien black locust (*Robinia pseudoacacia*) in their grassland restoration

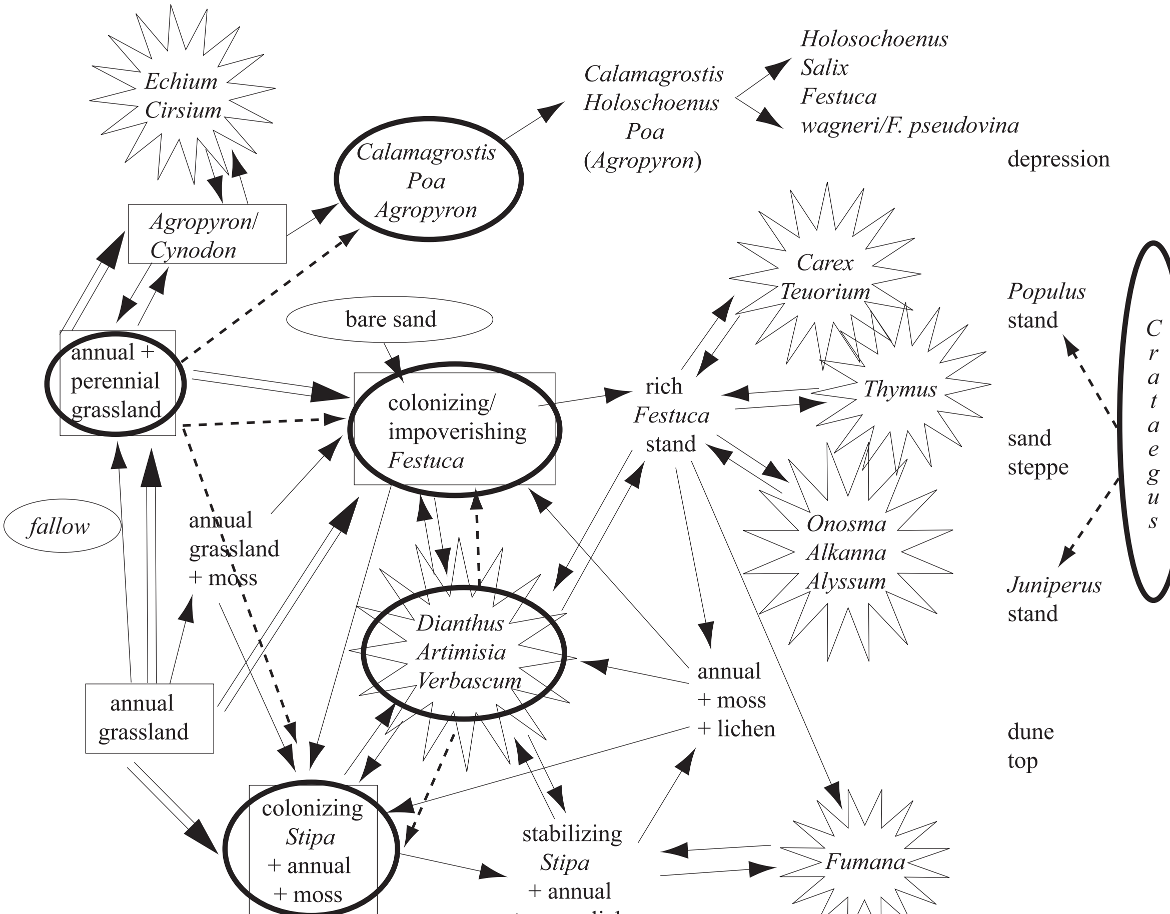

FIGURE 3. Multi-path developments of plant communities in unmown (bold ellipses and dotted arrows) and mown plots (squares and double-lined arrows) in a Hungarian grassland restoration site. From left to right the vegetation is closing, and from top to bottom it is getting more arid. Simple arrows indicate the original pathways of the conceptual frame. Species in stars are occasional accumulations of forb species on the original scheme. Modified after Bartha (2002).

study (Figure 3). Unpredictable courses of community succession in a changing environment add uncertainty to restoration trajectories.

In spite of such unpredictable processes as discussed above, predictive models are undoubtedly an essential tool for restoration planning (Palmer, Ambrose & Poff, 1997; Walker & del Moral, 2003; Walker, Walker & Hobbs, 2007). Ecological restoration researchers face a paradoxical challenge of developing predictive models from unpredictable nature (Parker, 1997; White & Walker, 1997; Klotzli & Grootjans, 2001; Choi, 2004), and restoration ecologists and practitioners in particular need those models and tools to test in real restoration projects. Acknowledging such unpredictability, a realistic restoration for the future needs to assume multiple alternative goals with multiple trajectories (Hobbs & Norton, 1996; Palmer, Ambrose & Poff, 1997; White & Walker, 1997; Suding, Gross & Houseman, 2003; Choi, 2004; Suding & Gross, 2006; Choi, 2007).

The individualistic concept of succession (Gleason, 1926) does not necessarily assume the deterministic end of succession; therefore, it may provide flexibility for setting multiple trajectories, particularly in conjunction with the facilitation, inhibition, and tolerance models of Connell and Slatyer (1977). Individualistic assembly of vegetation during "human-aided succession" and evidence for facilitation, inhibition, and tolerance pathways in post-mining sites have been documented in the literature (Choi & Wali, 1995; Cairns & Heckman, 1996; Choi & Pavlovic, 1998; Wali, 1999).

Recently, a new concept of holism, based on assembly rules (Diamond, 1975), has been introduced to ecological restoration (Temperton *et al*., 2004; Jentsch, 2007; Nuttle, 2007). Fattorini and Halle (2004) applied a "dynamic environmental filter model" (DEFM) for predicting spatial and temporal changes in ecosystem regeneration and invasion of species. According to the DEFM, every new species entering a system must fit through both an abiotic and a biotic filter, and disturbance can change the mesh of this 2-step filter. Nuttle (2007) found the DEFM useful for evaluating the current status of degraded ecosystems compared to non-degraded ones. However, the outcome of species assembly may not be clear (Hobbs & Norton, 1996; Wali, 2007). No "rule" of assembly has been identified yet, and thus the assembly rules at this time may serve as a hypothetical "guideline" rather than a "rule" for setting restoration trajectories (Temperton & Hobbs, 2004).

A special need for uniting community, ecosystem, and restoration ecology

Our earlier examples of eutrophication of habitats in Europe and California indicate very clearly that a target community in the future, after restoration action, may not be identical or even similar to past community or ecosystem assemblages (Jansen *et al*., 2004; Choi, 2007). Harris *et al.* (2006) stressed the importance of reinstating ecosystem functions as well as structure within newly restored assemblages under a changing climate. We endorse this argument; a future-aimed restoration should focus as much on reinstating certain ecosystem functions (Naeem, 2006) as on reinstating certain key species that are linked to specific functions. Luckily perhaps, restoring certain key ecosystem processes and functions is usually easier to achieve than restoring specific species to a site (see review by Lockwood & Pimm, 1999).

Ecosystem processes and functions (such as water and nutrient cycling) are affected by abiotic and biotic factors (Ehrenfeld & Toth, 1997; Naeem & Li, 1997; Naeem, 2002; 2006). In past decades, most restoration projects aimed to reinstate abiotic functions (*i.e.*, raise water levels in the case of degraded, drained peatland or maintain low-intensity grazing/mowing management in the case of European species-rich grasslands; Grootjans *et al*., 2002b) and hoped for a spontaneous return of displaced species. With the "if you build it they will come" mind-set, many restoration projects re-established specific abiotic disturbance regimes or settings, but the desired species did not necessarily return, and undesired non-native species often quickly became dominant (Levine *et al*., 2006). Restoration researchers have only relatively recently begun to take the issue of biotic limitations (such as need for pollinators, symbiotic interactions of plants with mycorrhizae, dispersal limitations, and microsite limitations) more seriously (Palenzuela *et al*., 2002; Gillespie & Allen, 2006; Harris & van Diggelen, 2006).

Recent cross-fertilization between community and ecosystem ecology has led to more focus on the link between the abiotic and the biotic functions. Biodiversity–ecosystem functioning experiments (BD-EF) are one such attempt at linking and understanding the effects of biodiversity on ecosystem properties and functions (Schulze & Mooney, 1993; Huston, 1997; Tilman, 1999; Hooper *et al*., 2005), as opposed to the more traditional ecological investigations on the effects of the environment (or abiotic factors) on species distributions. In the BD-EF experiments, where effects of a diversity gradient of species on primary productivity, flows of energy, and cycling of matter in a system were investigated, the 2 often disparate fields were effectively unified (Naeem, 2006). Many BD-EF experiments, especially in grasslands, have shown a positive relationship between species and functional group richness and ecosystem functions (Tilman, Wedin & Knops, 1997; Hector *et al*., 1999; Hooper *et al.*, 2005; Roscher *et al*., 2004; 2005; Spehn *et al*., 2005). A major debate in current BD-EF research relates to the exact nature and extent of the specific link between species or trait diversity and ecosystem processes.

Two main questions for the relevance of BD-EF experiments have emerged. The first question is whether positive effects of diversity on ecosystem processes are driven mainly by dominant species within a community (called sampling effect) or by more efficient resource use within an ecosystem when many species differing in their traits are present (called the complementarity effect) (Loreau & Hector, 2001). Recent meta-analyses suggest that both sampling and complementarity effects play a role in producing positive diversity effects on ecosystem processes (Hooper *et al.*, 2005; Spehn *et al*., 2005; Balvanera *et al*., 2006). The second question is whether effects found in BD-EF experiments, where abiotic conditions are kept as stable as possible while manipulating biotic components such as species richness, are relevant in more natural ecosystems exposed to far more complex interplay of abiotic and biotic factors (see Kahmen, Perner & Buchmann, 2005).

Plant communities in BD-EF experiments are *per se* young and probably far from maturation, such that interactions between the biotic and abiotic components of a more mature ecosystem may show very different key drivers of the system (Grace *et al*., 2007). Autogenic positive effects of biodiversity on ecosystem processes may only occur in early stages of ecosystems, or they may be compounded by allogenic factors (*e.g*., environmental conditions) in more natural settings. Restored ecosystems almost always are also young ecosystems. Here is where the 2 fields (BD-EF and ecological restoration) seem set to gain from each other.

Ecological restoration has a lot in common with BD-EF experiments, including the manipulation of abiotic conditions, the artificial assemblage of biotic species, and monitoring of ecological processes at community and ecosystem levels. More importantly, a number of BD-EF experiments have found that positive complementary effects increase over time among the species-rich assemblages (Pacala & Tilman, 1994; Lambers *et al*., 2004). In addition, some ecologists predict that positive interactions should increase as environmental conditions become more severe across an environmental gradient (Callaway, 1997). This could and should have implications for restoration. Certain restoration contexts, especially on degraded sites with extreme environmental conditions, such as post-mining sites, could prove to be the kinds of habitats where positive interactions may play a special role in establishing functioning ecosystems, at least at initial stages of restoration, and where there may be the potential for increasing facilitative effects within certain assemblages over time. Equally potentially useful, but not yet adequately tested, is one of the main insights of BD-EF experiments, that only a small number of plant species, given a certain environmental framework, can (re)establish productivity, nutrient and water cycling, decomposition, and humus forming processes in an ecosystem.

Given the increasing lack of predictability of restoration trajectories in a changing environment, it will be critical to advance our knowledge of dominant species effects *versus* complementarity effects in resource use within diverse species assemblages on ecosystem processes and functions. Application of the knowledge from BD-EF to restoration projects has the potential to enhance restoration success considerably. In addition, restoration projects, given collaboration between scientists and restoration professionals for adequate planning, may offer a panoply of opportunities for cross-fertilization. Ecological restoration sites and projects could and should provide a testing ground for evaluating effects of differently diverse species assemblages on ecosystem functions (as found in BD-EF experiments) in a real-world context.

Human interventions for ecosystem recovery

Human interventions for ecosystem recovery are central to our contemporary restoration practice (Cairns & Heckman, 1996; SERI, 2004). Human interventions may include, but need not be limited to, removal of invasive exotic species, amendment of site conditions, and provision of propagules or safe sites in the form of nurse plants (Temperton & Zirr, 2004). For example, Halassy, Torok, and Marko (2005) found that mowing prevented encroachment of woody shrubs (*e.g*., *Crataegus monogyna*) and guided the vegetation trajectory to the target community of *Festuca vaginata* and *Stipa borysthenica* in a Hungarian grassland restoration. In North America, Choi and Pavlovic (1994; 1998) found that herbicide treatment was more effective than burning or sod removal for controlling exotic invasive species and restoring native plants in Lake Michigan sand dunes. Such human interventions aim to allow progressive dynamic changes in the restored community over time in the future. Particularly, the initial site amendment and species introduction (Fukami *et al*., 2005) may determine the restoration trajectory, the time frame for trajectory development, and sustainability of the ecosystem after recovery (Cairns & Heckman, 1996; Naeth, 2000).

In Canada, Reid and Naeth (2005a,b), Gardner *et al*. (2003), and Graham and Naeth (2004) found that applications of urban wastes (*e.g*., biosolids and compost) could accelerate soil development and enhance bioremediation of hydrocarbons and metals in some post-industrial sites. Soil amendments needed to be accompanied by provision of propagules for initiating restoration processes. Mackenzie and Naeth (2006) found litter and humus layers of local forests had great potential for soil restoration because the litter and humus contained many propagules of native species that could increase revegetation success. Schaefer, Naeth, and Chanasyk (1999) constructed vegetation islands, by amending soils with compost and transplanting native species from adjacent undisturbed sites, in a disturbed boreal forest site in Elk Island National Park. The plants in the transplanted vegetation islands survived and continued to establish on this terraced slope. Plant species diversity increased with low mortality, often less than 10%. Over 75 species were identified in the area of egress around the islands. Similar results were found in Aspen Parkland and Jasper National Park (Naeth, Westhaver & Wilkinson, 2002; Naeth & Wilkinson, 2005a,b). The construction of vegetation islands on amended soil seemed to be a successful model for linking abiotic and biotic restoration in several drastically disturbed post-industrial lands and for providing an initial "kick-off" to the restoration trajectory (Naeth, 2000).

Enrichment of soil nitrogen by atmospheric deposition is an alteration of an abiotic filter (Hobbs & Norton, 1996) that may lead to invasion of exotic species. Native plants are often capable of growing in high nitrogen soils (Cione, Padgett & Allen, 2002) but are poor competitors with many exotic grasses (Yoshida & Allen, 2004). Exotic grasses can be removed by herbicide application (Allen, 2005). However, recurrence of such grasses is highly likely in the high-nitrogen soil. A number of treatments to reduce available soil nitrogen in an effort to control exotic species have met with variable success (Zink & Allen, 1998; Cione, Padgett & Allen, 2002). The treatments are typically addition of a carbon source (*e.g*., mulch and sucrose) to increase carbon to nitrogen (C/N) ratio and thus to immobilize soil nitrogen by microbial activity (Zink & Allen, 1998; Torok *et al*., 2000; Eschen, Muller-Sharer & Schaffner, 2006). However, these treatments are usually limited to small areas and will not be a large-scale solution to elevated nitrogen deposition.

The ultimate solution to elevated soil nitrogen would be strengthening air pollution legislation to reduce nitrogen emissions. Legislation for air pollution control is a social and political agenda, beyond the scientific realm of ecological restoration. In western Europe, for example, most of the deposited atmospheric nitrogen originates from ammonia/ammonium deposition, mainly from the agricultural practice of spraying liquid manure on the soil surface (Roelofs *et al.*, 1996; Verhagen & Van Diggelen, 2006). The Netherlands has been successful in reducing such ammonia emissions with legislation that forced farmers to inject manure into the soil instead of spraying it on the surface. This measure has reduced atmospheric nitrogen emission in the Netherlands from 80 kg $N \cdot ha^{-1} \cdot y^{-1}$ to 20 kg $N \cdot ha^{-1} \cdot y^{-1}$ during the last decade (Milieu-en Natuurplandbureau, 2006).

Ecological restoration in social context

Restoration ecology, like engineering, is an applied science with a direct link to human interests (Davis & Slobodkin, 2004; Choi, 2007), and extensive gaps between ecological theory and restoration practice (Falk, 2006) present a challenge. Unlike traditional ecology (question driven, skeptical, patient, and often impractical), restoration ecology is goal driven, committed, practical, and often impatient under financial, social, and political constraints (Walker & del Moral, 2003). The social and political aspects of ecological restoration necessitate support from people. To draw such support, restoration plans and goals need to be economically, ethically, socially, and politically acceptable and capable of being justified in all of these arenas (Hobbs & Norton, 1996; Higgs, 2003; Choi, 2004; Halvorson, 2004; Throop, 2004; Choi, 2007).

Restoration goals are determined by us, not by nature, although we may make significant reference to nature (Choi, 2007). For this reason, the goals tend to be determined by preconceptions or misconceptions that often place more value on certain target species or ecosystems. These may limit or bias the discussion of restoration possibilities, thereby preventing the development of more effective and efficient strategies. Thus, while restoration ecology needs to continue to develop an ecologically sound conceptual basis and improved understanding and techniques, it faces an important challenge in tackling societal expectations and improving societal contributions to increase the likelihood of successful restoration (Hobbs, 2004).

Socio-economic and philosophical aspects need increased attention (Gobster & Hull, 2000; Higgs, 2003). There have been a few integrated attempts to tackle various issues in restoration (particularly, defining the concept of naturalness) by a variety of disciplines, including history, anthropology, and philosophy. However, cross-referencing among these disciplines or within the ecological literature has occurred only minimally, if at all. While this may seem of only academic interest, in fact it can have important societal and environmental ramifications. A mix of scientific uncertainty, value-laden decisions, and unrealistic expectations could lead to costly and demoralizing failures, loss of confidence that restoration can deliver useful outcomes, and a redirection of funds to other initiatives, while leaving important ecosystem degradation untreated.

We see the reconciliation of the potential mismatch between ecological constraints, social expectations, and decisions based on disparate value sets as an important challenge to ecological restoration. What is viewed as restoration by some might be viewed as unnecessary or unwanted meddling with valued ecosystems by others, as in the controversy over restoration efforts in the Chicago area in the 1990s (Siewers, 1998; Gobster & Hull, 2000; Gobster, 2001). Consideration of the sociological elements of restoration is likely to be critical in ensuring community support for restoration projects. Therefore, ecological restoration will increasingly have to draw, not only on classical scientific methods such as standard ecological sampling and analysis ("normal" science), but also on the emerging and still controversial approaches embodied in "post-normal" science (Ziman, 2000; Gauch, 2003). This will involve increased interaction among disciplines and an overt acceptance that it is important to include values as a valid component of the process.

Caveats

The Earth's environment of the future, particularly in the wake of global climate change, will be different from the one in the past. Therefore, restoration that is aimed at replication of past ecosystems will not be possible, or at least not sustainable, in the future environment. We advocate directing (or redirecting) the principles and practice of ecological restoration to the future along the following lines.

Restoration plans should set multiple goals and trajectories acknowledging the dynamic nature of ecological communities in the changing and unpredictable environment of the future.

Historical information is a useful guideline but should not become a "straightjacket" for projecting restoration goals and trajectories to the future.

Restoration goals should focus on rehabilitation of ecological functions for the future environment rather than solely on recomposition of past species assemblages. Connection of landscape elements may improve ecosystem functions and structures.

Succession models are applicable to projecting restoration trajectories. The individualistic model, in conjunction with the facilitation, inhibition, and tolerance paths, in particular provides flexibility for projecting multiple trajectories. Assembly rules, particularly the recently proposed "dynamic environmental filter model", are a new holistic concept that need to be tested for such projections.

Various restoration techniques are available for aiding recovery of ecosystem functions and structures. An interdisciplinary approach (*e.g.*, biodiversity–ecosystem functioning experiments) is necessary for better projection and understanding or restoration outcomes.

Restoration ecology is an applied science that has a direct link to human interests; therefore, restoration efforts need support from people. To draw such support, restoration plans and goals should be economically, ethically, socially, and politically acceptable and well-justified in these terms.

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