We dedicate this book to all those whose life work is to protect and restore diversity.

Part V from this book

Restoring Diversity

Strategies for Reintroduction of Endangered Plants

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The twenty-five Participating Institutions of the Center for Plant Conservation constitute some of the best expertise in the biology of rare plant species to be found anywhere. Their staffs, who manage the National Collection of

PART V

Guidelines for Developing a Rare Plant Reintroduction Plan

The reintroduction of any species is inherently complex. For endangered species, the complexity is exacerbated by a shortage of sound policies, effective models, and strong scientific underpinnings.

As is clear from several chapters in this book, the science of reintroduction is in its infancy; so too is the development of a policy framework to guide the use of reintroduction as a conservation tool. This policy vacuum is most evident with regard to the relationship between introducing new populations and conserving existing ones—a relationship that remains poorly articulated. For example, the policy of the International Union for Conservation of Nature and Natural Resources (IUCN 1987) describes translocations as powerful tools that can materially advance the diversity and viability of populations and habitat but notes that "like other powerful tools they have the potential to cause enormous damage if misused." (p. 1). The potential damage referred to—and much of the overall concern about reintroduction—is that its use will in some way displace the imperative to conserve existing populations and communities. The challenge, therefore, is to unlock the creative potential of reintroduction while guarding against its possible misuse. As the chapter on compensatory mitigation illustrates clearly, the ecological "meaning" of reintroduction depends largely on whether the natural populations still exist, or if they are somehow lost in the course of a project.

This chapter is intended to assist biologists and managers considering the use of reintroduction as a conservation tool. We have prepared these guidelines in the belief that well-planned and well-executed reintroduction can contribute materially to the goals of biodiversity conservation. While no single book can answer all relevant questions, the chapters in *Restoring Diversity* provide an excellent basis for advancing the understanding of the issues.

Re-establishment of populations is too variable to be reduced to simple formulations; there is no cookbook of reintroduction recipes. We have instead developed a series of questions that practitioners of reintroduction are likely to encounter. To form the framework for a reintroduction plan, the organizers of any well-planned project should be able, at a minimum, to provide coherent and well-researched answers to the following questions:

Is reintroduction appropriate?

1. What guidance can be found in existing policies on rare species reintroduction?
2. What criteria can be used to determine whether a species should be reintroduced?

3. Is reintroduction occurring in a mitigation context involving the loss or alteration of a natural population or community?
4. What legal or regulatory considerations are connected with the reintroduction?

How will reintroduction be conducted?

5. What are the defined goals of this reintroduction, and how will the project be monitored and evaluated?
6. Has available ecological knowledge of the species and its community been reviewed? What additional knowledge is needed to conduct the project well?
7. Who owns the land where the reintroduction is to occur, and how will the land be managed in the long term?
8. Where should the reintroduction occur?
9. What is the genetic composition of the material to be reintroduced?
10. How will the founding population be structured to favor demographic persistence and stability?
11. Are essential ecological processes intact at the site? If not, how will they be established?

These questions and the discussions that follow can help the restorationist cope with the likely event that the operative environments more closely resemble the imperfect world of compromise than they do the ideal. Our intention is, if not to provide definitive answers, at least to provoke good questions complementing other policy discussions (Falk and Olwell 1992; BGCI 1994). The guidelines are intended to be a template for further critical thinking about reintroduction. We hope that these ideas will serve as a foundation for scientists, agencies, non-government organizations, and others to develop specific policies and handbooks relevant to their own work.

We have discussed, in length, the semantics associated with reintroduction, introduction, and augmentation in the Introduction to this book. Restorationists should consider these concepts when setting goals or selecting the site or the source material. Guidelines 5 and 8 discuss the distinction between reintroduction, introduction, and augmentation; therefore, we have not redefined them here.

The biophysical aspects of a rare plant species—that is, the ecological community, the ecological processes, and the environmental context into which the species will be placed—make up the core operational details of reintroduction planning. Each species is unique in its taxonomy, history, ecology, and biogeography; each reintroduction thus presents novel challenges. Several guidelines stress the importance of matching the ecological and physical characteristics (process and structure) of the rare plant in its native habitat with those in the reintroduction site. Some of these elements (such as site selection and selection of source material) are limited to certain phases of a reintroduction; others (such as genetics and ecological processes) are general aspects to consider during several phases of a project.

Is Reintroduction Appropriate?

1. What guidance can be found in existing policies on rare species reintroduction?

In an effort to assess the state of existing rare plant reintroduction policy, the editors and contributors surveyed a wide range of agencies, organizations, and corporations. We reviewed dozens of documents from international conservation organizations, U.S. federal agencies, state agencies, national conservation organizations, private corporations, native plant societies, and professional organizations. These policies, along with the chapters in this book and other published literature, served as the primary materials for development of the guidelines (Table 1).

Many policies were in draft form, reflecting the evolving state of the field. Some policies are broad formulations, while others (such as Gordon 1994) apply only to a single preserve system. United States federal agencies focused primarily on the legal aspects of reintroduction and the ways in which such activity relates to the Endangered Species Act.

In addition to commentary on various specific topics treated in the following sections, several "take-home" messages emerge in existing policies about reintroduction:

- *It is far better, where appropriate, to conserve existing populations and communities than to attempt the difficult and imperfect task of creating new ones.*
- *Reintroductions are fraught with uncertainty and difficulties and should be viewed as experiments.* As such, it is unwise to rely on "successful" outcomes, given the risks of failure are significant (as is often the case in compensatory mitigation).
- *Determining the outcome of reintroduction takes time.* It certainly takes years, and probably takes decades, depending on species and community characteristics. For instance, Birkenshaw (1991) describes a detailed four- to five-year process for initial preparation, outplanting, and preliminary monitoring alone. As Sutter (Chapter Ten) points out, this means that for all practical purposes, monitoring should continue for the foreseeable future in most reintroductions.
- *Learning opportunities exist throughout the reintroduction process.* To reintroduce confidently, we need extensive and detailed knowledge about the species, its community, and the larger ecosystem. For most rare species this knowledge base is minimal and unevenly distributed among species or communities. Most projects will thus have to proceed on the basis of incomplete knowledge and preferably incorporate learning into the project design.
- *Documentation of outcomes of every reintroduction effort is extremely important.* Many journals accept data from reintroduction projects in progress; practitioners and scientists alike should publish preliminary results or progress reports, including negative outcomes (it is every bit as important to learn which techniques failed as it is to learn which ones worked.) If a project is well-conceived and executed, any outcome will yield useful ecological information.
- *Planning and long-term commitment are of utmost importance to the success of a reintroduction project.* Nearly all policy discussions agree that reintroduction is best when it is part of a comprehensive conservation and recovery strategy for the species and its community. If such a plan is developed, then reintroduction can be better incorporated into the larger objectives.

2. What criteria can be used to determine whether a species should be reintroduced?

Practically speaking, reintroductions are nearly always experiments. Accordingly, before beginning a reintroduction, organizations considering such projects should examine critically the reasons for conducting them. Reintroduction may not be the most effective or successful means to

TABLE 1. Policies and guidelines reviewed.

Agency/ organization	Type of organization	Form of document	Date of document
American Society of Plant Taxonomists	Professional organization	Resolution	1989
Botanic Gardens Conservation International	International conservation organization	Draft handbook	1994
Botanic Gardens Conservation International	International conservation organization	1988 workshop report	1990
Center for Plant Conservation	National conservation organization	Journal article (Falk and Olwell 1992)	1992
Florida Nature Conservancy	State conservation organization	Journal article (Gordon 1994)	1994
Illinois Endangered Species Protection Board	State government agency	Policy	1992
IUCN (International Union for Conservation of Nature and Natural Resources)	International conservation organization	Draft guidelines	1987 1992
National Park Service	Federal government agency	Policy	1988
Nature Conservancy Council (U.K.)	National conservation organization	Guidelines (Birkenshaw 1991)	1991
New England Wild Flower Society	Regional conservation organization	Policy	1992
Native Plant Society of Oregon	State native plant society	Policy	1992
The Nature Conservancy	International conservation organization	Draft policy	1992
U.S. Army Corps of Engineers	Federal government agency	Policy guidance letter	1991
U.S. Bureau of Land Management	Federal government agency	Policy	1992
USDA Forest Service	Federal government agency	Policy	n.d.
U.S. Fish and Wildlife Service	Federal government agency	Draft policy	1992
Waste Management	Private industry	Guidelines	1992
Wisconsin Department of Natural Resources	State government agency	Draft policy	1991

advance the conservation of an endangered species. Careful thought should be given to the reintroduction's potential effects on the future of the species and its community (Reinartz 1995). The expense and effort of reintroducing rare plants and establishing new populations should be undertaken for specific, defensible reasons, and not simply for opportunistic reasons, such as the availability of plant material.

By what criteria, then, can populations and species be selected as promising candidates for reintroduction? The following characteristics may render a species or population a good candidate:

- A species or population is extinct (or nearly so) in the wild. This depends on whether appropriate genetic material is available and whether threats can be managed.

- It has unnaturally few, small, or severely declining populations. Many new tools are emerging that can improve the traditional classification schemes used to identify the most endangered species. Among the most promising are those that use population viability analysis (PVA) to make quantitative, probabilistic predictions about the likelihood of a species becoming extinct (Mace and Lande 1991). While these methods are not without theoretical and pragmatic difficulties (Taylor 1995), they represent a potentially more powerful way to identify species that may be deserving candidates for reintroduction provided that other conditions listed below can be met.

- It has poor protection of existing natural populations.

- It shows evidence of problems with dispersal and/or fragmented habitat. Reintroduction may be a valuable conservation tool for overcoming the inability of some rare plants to disperse effectively to appropriate habitat, especially in fragmented natural habitats.

- It is anticipated to be affected adversely by climate change. Rapid climate change may place new demands and constraints on conservation of rare plant species (Kutner and Morse, Chapter Two; Morse, Chapter One); reintroduction may be part of the solution to these conservation challenges.

- It has available high-quality source material. This material should be genetically diverse, disease free, and of an appropriate provenance.

- It can be successfully propagated and established in experimental trials.
- Its reintroduction is supported by a recovery team. The team agrees that reintroduction will contribute positively to the conservation of the species.

Conversely, certain characteristics may render a species inappropriate for reintroduction:

- Reintroduction or establishment of new populations will undermine the imperative to protect existing sites.
- Feasibility of growing and establishing new populations has not been demonstrated, if the project involves loss of a natural site.
- High-quality appropriate source material is not available.
- Existing threats to natural (or other reintroduced) populations have not been controlled.

3. Is reintroduction occurring in a mitigation context involving the loss or alteration of a natural population or community?

Reintroduction of threatened species is most controversial when practiced in a context of compensatory mitigation. Mitigation directly challenges the relationship of restoration to conservation, in that it requires us to judge the value of existing nature against an artificial substitute. Recognizing that every mitigation case is different, we discuss some of the issues that should be examined.

A broad consensus exists among conservation biologists and planners that it is better to protect existing native populations and communities than to create new ones. Most policies that address compensatory mitigation emphasize the importance of protecting existing diversity. The highest possible priority must be given to avoiding or minimizing impacts to natural populations, especially where rare species or communities are concerned.

However, it is manifestly impossible to fulfill the mandate to "always protect existing sites from development" (Birkenshaw 1991, p. 4). If this were the case, mitigation would not have to occur at all. Compensatory mitigation represents a strategic gamble that the net goals of biological conservation will be furthered if resources of land development and commodity extraction can be diverted to protect some species and some habitats. In addition, legal protection for rare plants often does not apply on privately held lands (Bean, Chapter Sixteen; Klatt and Niemann, Chapter Fifteen). While many private and corporate landowners voluntarily attempt to avoid damaging rare plants, they are often under no requirement to do so except in the case of wetlands and certain government-permitted activities. In such circumstances there may be no legal way to prevent a development-related translocation.

These realities suggest difficult questions: What, if any, are the characteristics of a good mitigation? Under what circumstances should species or communities be off-limits to any form of tradeoff? Are there circumstances in which mitigation-related reintroduction involving the destruction of a naturally occurring population advances the cause of conservation? Planners should consider the following:

SPECIES RARITY AND VULNERABILITY

Somewhere along the continuum of increasing abundance, an implicit judgment is made that a species is not of conservation concern, and that not every population of a species needs to be protected. For very rare organisms, by contrast, every individual may warrant protection. Somewhere between these two extremes lie the many species for which the fate of an individual population has an uncertain relationship to the future of the species. It is this middle zone of species for which mitigation policy is most important.

Any mitigation policy needs to state clearly that certain species and populations are categorically off limits to destruction. In particular, this applies to extremely rare species—those with very few populations, a small number of individuals, or an extremely restricted geographic range. Unfortunately, terms such as *few*, *small*, and *restricted* lack dimension and can thus be interpreted in several ways. For example, minimum viable population (MVP) standards could conceivably be used to ascertain the sustainable size of a population. But MVP analyses result in probabilistic statements about extinction or persistence, not absolute values. Similarly, rarity is a multidimensional quantitative attribute, not a simple categorical state (Fiedler and Ahouse 1992). The uncomfortable fact is that the threshold for tolerance of a possible destruction event is often difficult to define in intermediate cases of species viability.

There is no standard of rarity, numerical or otherwise, that can be applied across taxonomic and ecological lines. The conservative approach is thus to set limits high: only populations of abundant or stable species should be subjected to mitigation tradeoff. This places the burden of proof squarely on the mitigation proponent, where arguably it should be. The biological rationale for every case must be worked out individually, but mitigation should proceed only when it can be demonstrated with acceptable certainty that there will be no irreparable harm to the species as a whole.

Community or Habitat Uniqueness

Unique habitats are as important to save as are populations of rare species. Certain communities represent an irreplaceable combination of ecological history and function. Many also harbor populations of rare or habitat-restricted species. Mitigation tradeoffs of rare community types should be avoided altogether, if for no other reason than to prevent more species from becoming endangered (See Gann and Gerson, Chapter Seventeen; Zedler, Chapter Fourteen).

Uncertainty and the Distribution Risk

The natural processes of colonization and establishment are often very low-probability affairs. While some aspects can be made more predictable in a deliberate outplanting, a great deal of uncertainty surrounds any newly established population. Reviews of existing literature (Hall 1987; Fiedler 1991) indicate that failures—low germination and establishment rates, losses due to droughts or floods, massive herbivory events, and other obstacles to successful colonization—are more common than success (Howald, Chapter Thirteen; Case Studies). These difficulties may be only the *visible* evidence of failure. Less obvious problems may lurk in reduced gene pools; absent pollinators, dispersal agents, or mycorrhizae; and compromised functional parity with undisturbed natural systems (Zedler, Chapter Fourteen).

The threshold of acceptable certainty must be set substantially higher any time a natural population is proposed for destruction. Where reintroduction is practiced "proactively" (*sensu* New England Wild Flower Society 1992) as a conservation measure to heal past harms, this uncertainty may be tolerated because existing populations are not being placed at additional risk. When the equation involves the destruction of natural populations, however, the balance potentially shifts to the negative. Mitigation often involves trading off existing, naturally occurring habitat for created systems of unknown ecological value and an uncertain future.

One of the central problems with mitigation is the unequal distribution of risk in various parts of the process. For example, when a population is to be destroyed by construction activity and replaced by a newly established population elsewhere, the destruction is certain and immediate; it *will* happen. The replacement, however, faces an uncertain future; its prognosis fifty or even five years in the future cannot be predicted. The brunt of uncertainty, therefore, falls primarily on the replacement population. This asymmetry of risk constitutes a major problem for many proposed mitigation projects.

The condition of the reintroduction site poses an additional difficulty. If, as is often the case, the outplanting site is itself in a degraded or altered condition, then the prospects for successful establishment are reduced further. Altered or degraded sites will rarely offer suitable conditions for trading against any naturally occurring population.

Because mitigation efforts are so uncertain, they should be viewed as a last recourse in dealing with development impacts. Draft U.S. Fish and Wildlife Service policy states that propagation and reintroduction should supplement, not replace, conservation of existing populations (McDonald, Chapter Four). Some corporate policies also recognize avoidance or minimization of impacts to naturally occurring, sensitive populations as preferable (Klatt and Niemann, Chapter Fifteen). Only when impacts to rare species are genuinely unavoidable, after a good-faith effort, should compensatory mitigation be considered as an acceptable alternative.

The Mitigation Time Scale

Transplants take a long time to become part of a functioning ecological community, if they ever do (Pavlik, Chapter Six; Zedler, Chapter Fourteen). There is little research on establishment times for new populations under natural circumstances, let alone artificial outplantings. Whatever insight exists comes largely from the literature on postdisturbance recovery and succession, which suggests that community-level relationships can take decades to equilibrate.

As with the allocation of risk, the relative time scales of destruction and replacement are asymmetrical. Once a project begins, destruction of an existing population or habitat is more or less instantaneous. The "creation" of a new population or habitat, by contrast, is a matter of many years or decades. In combination with the high degree of uncertainty, the long time frame can make the promises of mitigation-related tradeoffs difficult to evaluate (Berg, Chapter Twelve; Zedler, Chapter Fourteen).

Mitigating Impacts Across Biological Levels

One commonly used compensatory mitigation technique involves salvage or rescue of individual rare plants that are about to be destroyed. In some cases entire populations consisting of hundreds of individuals are dug up and relocated. Under the best of circumstances, plants are taken in blocks of soil, in the hope of exporting site-level symbionts to the new location (Johnson, Case Study Six). Most of the time, however, what is removed from the site consists primarily of individual plants to be transplanted elsewhere.

As a form of mitigating impact, this practice obscures the different levels of biological organization affected in both sites. A natural, complex ecological community is lost or destroyed, involving many species and their interactions with each other and with the abiotic environment. What are "saved" are a few individuals representing some fraction of a single population of a single species, with no supporting context. Even in the case of very rare species, this is not an acceptable exchange; if the species is of conservation concern, then so should be the habitat in which it exists.

ELIMINATING CAUSES OF DECLINE OR THREATS

A replacement population can be established only if the original causes of decline have been eliminated. These threats can include invasion by exotic weeds or feral herbivores, disease, suppressed or altered fire regimes, flood suppression, elimination of native pollinators or dispersers, or more pervasive effects such as weather or climate changes (Ledig, Chapter Eleven; Case Studies 2 and 6, this volume). If factors that led to the species' decline remain present, then there may be little reason for confidence in a replacement site. (See Pavlik, Chapter Six; and Pavlik, Nickrent, and Howald 1993) As with reintroduction into physically altered or degraded sites, a mitigation-related outplanting is unlikely to succeed in the long term if the threatening processes have not been eliminated.

4. What legal or regulatory considerations are connected with the reintroduction?

Legal protection for plants is far more limited than legal protection for animals. The most important legislation dealing with the reintroduction of rare plants is the U.S. Endangered Species Act of 1973. The act provides protection for federally listed plants on federal lands and in situations where federal funds, permits, or other actions are involved. The act does not protect endangered plants on private lands.

Reintroduced populations of federally listed plants on federal lands are automatically protected under the act (U.S. Fish and Wildlife Service 1988), and reintroduced populations are protected exactly as the other populations of the listed species, unless the reintroduced populations are listed under the act as experimental. Experimental populations are designated as either *essential* or *nonessential*. An essential population is protected as a threatened species, and a nonessential population is treated as a proposed species under the act. However, the experimental population designation has yet to be used for plants. If a federal agency is worried about reintroducing a listed species onto its lands because the reintroduction would limit their management actions, then an experimental population designation may be useful. In most cases the federal agency considers all sites, puts the reintroduced population in an area with less conflict, and avoids the use of experimental population designation. To date, no experimental population designation has been used for a plant reintroduction.

If the reintroduction involves federal agencies, then a Section 7 consultation with FWS may be necessary. Permits may be obtained from the U.S. Fish and Wildlife Service (FWS) to collect propagules from lands under federal jurisdiction or to reintroduce a federally listed species on federal lands.

As McDonald (Chapter Four) indicates, draft policy guidance for the U.S. Fish and Wildlife Service states that "propagation programs will not be employed in lieu of habitat conservation (USFWS 1992, p. 1)." Protection of the species and its existing habitat is the foremost objective of a recovery program, with reintroduction being a tool to assist in the recovery of the species.

The lack of federal protection for plants on private lands simplifies the reintroduction process and may increase the likelihood of finding a private property owner who would allow a reintroduction on their property. In the case of *Amsonia kearneyana*, an endangered plant in southern Arizona, it was the goodwill of the owners of a canyon just east of its only known canyon locality who volunteered their property as the site for the reintroduction (Reichenbacher 1990).

State laws regarding endangered plants differ, and not all U.S. states have such legislation. Individual state laws should be checked to see which plants are covered, what activities are allowed, and how the permit process works. (For information on states with rare plant laws or contacts at federal or state agencies see the 1995 *Plant Conservation Directory* [Center for Plant Conservation 1995].)

How Will the Reintroduction Be Conducted?

5. What are the defined goals of this reintroduction, and how will the project be monitored and evaluated?

Once it is determined that reintroduction can help conserve a species or community, the planners must determine what the objectives are and how outcomes will be evaluated. As Pavlik (Chapter Six) notes, however, there is little consensus on standards of success. Moreover, reintroduction projects are so diverse that a single evaluation standard cannot be offered here. Consequently, the evaluation of each project will be based on some combination of standard measures and ad hoc criteria. Further project activities can then be subject to adaptive management if the outcome does not meet those criteria.

Definitions of Success

Pavlik (Chapter Six) defines success at the population level as "meeting taxon-specific objectives that fulfill the goals of abundance, extent, resilience, and persistence." Pavlik is careful to distinguish between this definition of project success and biological success, which "only includes the performance of individuals, populations, and metapopulations of a targeted taxon."

Monitoring

Monitoring is essential for evaluating success in a reintroduction project. Sutter (Chapter Ten) observes that "Monitoring is the foundation of success . . . not a luxury." Sutter sets out four criteria for a reintroduction monitoring program: (1) monitoring data must have a known and acceptable level of precision; (2) data-collection techniques must be repeatable; (3) collection of data must be done over a long enough period of time to capture important natural processes and responses to management; and (4) the monitoring design must be efficient.

In addition, monitoring objectives must be specific and quantifiable and must define the framework for specific tasks. To evaluate outcomes, Sutter (Chapter Ten) suggests four elements of a reintroduced population that need to be monitored: (1) plants reintroduced to the site, (2) recruitment of new individuals, (3) condition and functioning of the community and ecosystem ,and (4) genetic variability of the population of reintroduced plants.

If we begin to think of all reintroductions as experiments, a vital step in any project will be to use the information from monitoring in managing the species or community. Such feedback is crucial because reintroduction should be an iterative process (Pavlik, Chapter Six). Agency plans must be flexible enough so that the original design can be modified to include information gleaned from the monitoring process (a process called adaptive management). However, those conducting reintroductions should not be too quick to change a monitoring or evaluation scheme simply because a project doesn't appear to be progressing as planned. The failure of a new population to establish provides important information about the biology of threatened and endangered plants and about the frequency of successful establishment in reintroduction programs (Pavlik 1994).

6. Has available ecological knowledge of the species and its community been reviewed? What additional knowledge is needed to conduct the project well?

Although most guidelines and policy formulations state that reintroduction should be based on a sound understanding of species and community ecology, there remains a general shortage of reliable information about many rare species. In such cases, should a project proceed or be delayed until an adequate (however defined) knowledge base exists? Moreover, it is commonly recommended that each reintroduction be treated as an experiment, in terms both of acknowledging uncertain outcomes and gleaning opportunities for learning. But designing an experiment to generate knowledge and designing an implementation project to maximize chances of short-term success may require different approaches. Can reintroduction be designed simultaneously as potential successes and as experiments?

Basic Knowledge of Rare Species Biology

The field is wide open for research initiatives into rare species biology, especially for work involving the ecology of reintroduction and restoration (Wildt and Seal 1988; Falk and Holsinger 1991; Bowles and Whelan 1994; Schemske et al. 1994). The published literature will rarely be sufficient to answer all relevant questions about the ecology of a rare plant species proposed for reintroduction. Since these ecological relationships are especially germane to the process of reintroduction, it is unlikely that the practitioner will have the desired scientific basis in hand. This leaves reintroduction planners in the position of making more or less educated guesses about the response of the species, and makes the practice of restoration generally one of informed speculation. This predicament is most troubling in circumstances in which "failure" has significant consequences, such as critically threatened species, those for which very limited source material is available, or any situation involving a destructive tradeoff with an existing natural population.

Translocations as Experiments

To some extent, simply documenting and publishing methods and outcomes will improve empirical understanding of reintroduction ecology. But if reintroductions are to serve as more refined investigations, they must conform to the criteria that would make them good experiments. This means including the basic elements of experimental design: controls, replication, a limited number of variables, and tests of statistical significance. These conditions are not automatically satisfied in applied restoration work, where the proximate objective may be to succeed according to the terms of a contract, rather than to learn. There is no standard formula for achieving the correct balance of immediate results and expanding knowledge, although the two should be recognized as complementary in the long run (Zedler, Chapter Fourteen).

Any reintroduction project can contribute to the empirical knowledge base by recording baseline conditions. This includes carefully recording the number and type of individuals outplanted, their genetic diversity (if known), outplanting protocols (spacing, depth), soil treatments, management measures, and a detailed description of the receptor site and locality, preferably in a Geographic Information System (GIS). At the very least, this information should be recorded in the archives of a public or private conservation agency; a better practice is to offer the data for publication. Without such information, the knowledge surrounding reintroduction projects may be as ephemeral as the memories of those who conducted them; with proper documentation, projects can serve as empirical tests to which restoration ecologists can return decades later to interpret long-term outcomes. This orientation to the long term is vital to understanding natural ranges of variation and performing trend analyses for ecological responses such as population size and genetic variation. It is also probably the only means by which long-term empirical studies of time scales in the reintroduction process will be carried out. The key to all this is to capture baseline data early.

Some of the best research opportunities are for study of the ecological processes that reintroduction mimics: founder events, small population dynamics, establishment-phase competition, dispersal and disturbance ecology, and patch dynamics. Studies can be directed at colonization of ephemeral or disturbed habitats and at the effects of succession on population persistence, resilience, and stability over time. Cohort studies of reintroduced populations can provide data on the natural range of variation in survival, mortality, and recruitment. Reintroducing plants along gradients of key habitat parameters (moisture, light, elevation) will allow examination of the influence of these and other measures of habitat specificity.

Designing a study for research purposes may require a different approach than for maximizing short-term "success," at least in some instances. Research studies typically focus on a limited number of variables, and provide a wide enough range of conditions in the chosen variables to permit hypothesis testing. This means that some translocated plants may "fail" by growing, reproducing, or surviving at a lower rate than plants exposed to

other conditions in the test matrix. In other words, an experiment may "succeed" in explaining different outcomes, but "fail" to result in the establishment of a permanently viable population. By contrast, if the primary objective is to establish a viable population, then outplanting may need to be restricted to (or at least centered on) conditions known to offer the best prospects for survival. Hybrid approaches are possible, in which a somewhat larger number of variables are tested across a more limited magnitude of values, without the full range of conditions or controls. Under these circumstances, outplanting may provide some usable scientific information and create a viable population.

While the outcome of any individual reintroduction project is unknown at the beginning, the complex interactions of many factors offer an exciting and important opportunity for learning. As reintroduction progresses from trial-and-error to an adaptive ecological management tool, its design can increasingly accommodate needs for both science and conservation practice (Pavlik 1994). Over time, careful experiments will improve both the base of knowledge and the prospects for success.

7. Who owns the land where the reintroduction is to occur, and how will the land be managed in the long term?

Long-term funding and land management are important elements of any reintroduction plan. Many reintroductions are conducted without adequate planning for land use, management, or financial support. Since a reintroduction may take years or even decades to stabilize, inadequate planning can seriously compromise the long-term prospects for success. A reintroduction project that needs two decades of monitoring may outlast the tenure of most agency personnel.

Landowner Commitment to Permanent Site Protection

For rare plant reintroduction to be of permanent value to conservation, the habitat must be securely protected for the long term. Landowners and land managers of the sites must be brought into the dialogue early in the planning phase. The Knowlton cactus reintroduction project (Cully, *Pediocactus knowltonii* Case Study Two) began with considerable dialogue and commitment between seven federal and state agencies and one conservation organization. It is this continued dialogue and commitment that keep the project going ten years later. There are, unfortunately, many cases in which a rare plant was reintroduced onto a site and eventually forgotten because of personnel or priority changes. In some cases these "forgotten" sites were later used for other purposes—parking lots, building sites or, mowed roadsides—not compatible with the population's survival (Hall 1987). Texas snowbells (McDonald, Case Study Three) illustrates of public and private landowner cooperation in a political climate that would otherwise be unsupportive.

A useful litmus test is to determine if the reintroduction is basic or incidental to the owner's primary interest or the managing agency's mission. Commitment by the landowner or manager to the project in general, and the use of the reintroduction site specifically, needs to be secured prior to initiating the project. Without a long-term commitment to protecting and managing the site, reintroduction projects are exposed to elevated long-term risk. A reintroduction plan should outline the main elements of the long-term program. This plan may take various forms, such as a recovery plan or land use document. Private lands can be secured by an easement or, as in the case of *Amsonia kearneyana*, by voluntary agreement with the landowner (Reichenbacher 1990). Reintroduction plans should cover a period as long as required by the institutional, legal, biological, and monitoring requirements of each species reintroduced; this will rarely be less than five years.

Funding for Long-Term Management

Two reviews of the outcomes of rare plant reintroduction work in California (Hall 1987; Fiedler 1991) found follow-up to be lacking in many projects. The main reason was the shortage of funds allocated to monitoring and ongoing habitat management. To design a legitimate reintroduction project, planners must define the time frame associated with measuring those outcomes and then incorporate these costs associated with the full life of the project. Such planning will make many reintroduction projects more expensive, but two benefits are realized: first, the process provides a more realistic assessment of the real long-term costs. Second, once such costs are made explicit as part of the agency's commitment, it may be more difficult for them to be rescinded than if they were "invisible" in the organization's budget.

Even if costs can be estimated for the entire life of the project, it may be difficult to secure financial commitment for a long-term project. Funds are usually appropriated to federal and state agencies on an annual basis, although programmatic commitments extend into decades. Funding for reintroduction projects undertaken in a mitigation context and paid for by a developer is often subject to even more stringent funding constraints, unless longer management is required by law. Institutional commitment and involvement in the project may improve prospects of funding for the full life of the project. In the extreme cases, conservationists may find it necessary to

refuse to undertake an outplanting project without long-term commitment and funding. Such a position would send a strong message about what is required to reintroduce rare plants successfully.

Bean (Chapter Sixteen) and Klatt and Niemann (Chapter Fifteen) suggest various ways for securing a long-term commitment. These include dedicated trust funds, surety bonds, and other irrevocable financial guarantees to be used for ecosystem management. Many statutory and contractual models exist for such guarantees, which can be adapted from construction and performance bonding or siting hazardous waste facilities. Although financial assurances for endangered species projects are generally not required at present, such guarantees could be included by regulatory agencies as a permit condition. Such up-front financial commitment is preferable if it can be obtained, in part because of the security afforded for the future.

Habitat Management

A reintroduction site must be managed as an ongoing ecological unit long after the initial outplanting. Processes that need to be addressed include controlling exotic plants and animals, restoring disturbance regimes such as fire and floods, and reducing new sources of anthropogenic impact (Huenneke and Thompson 1995).

The reintroduction plan should also consider the bioregional context of the project. Since all reintroduction projects should aim to become part of landscape-scale conservation efforts, knowledge of current and future land use is imperative. By focusing on landscape-level management one can better ensure that the reintroduction is nested in a larger context and not subject to short-sighted decisions and policy changes.

Other Land-Management Considerations

Populations of some rare species (such as *Betula uber* and *Asclepias meadii* have experienced what appear to be intentional acts of vandalism. Reintroduction in controversial locations (such as range allotments on public lands) may be similarly vulnerable. Public ownership can offer strong protection, although private nature preserves may be superior if available. Potential future landscape configurations should be considered in selecting the best site. For example, will land settlement or land management practices around the site result in a biological island for the rare plant?

Availability of superior sites meeting all criteria (see Guideline 8) will almost always be the limiting factor; the best site may be prohibitively expensive to obtain or administratively or practically unavailable.

8. Where should the reintroduction occur?

Selection of a reintroduction site is a central decision in any project; perhaps no other single aspect influences the eventual outcome as strongly. Site selection involves important long-term considerations of security, management, and monitoring physical/geomorphic, biological, and spatial-temporal considerations. And yet, as Birkenshaw (1991, p. 6) notes, "[E]xcept in the case of re-establishment . . . the selection of a translocation site is, to some extent, a shot in the dark."

Once a decision has been made to proceed with reintroduction, a key step is selecting the receptor site on which the new population will be established. Ideally, it is best to match physical and ecological conditions of the species in its native range with the reintroduction site. In practice, however, outplanting may have to occur in areas with introduced exotics, communities that differ from the species' native habitat, and the unpredictable challenges of climate change.

Criteria for Suitable Translocation Sites

The ideal receptor site is not difficult to define hypothetically: it matches the habitat characteristics of the target species (such as biotic community, ecosystem function, and spatial context), and especially of those native populations closest and most similar to the potential receptor site. In practice, however, these conditions are rarely satisfied, for several reasons. First, for only a very few species can we define optimal (or even typical) habitat. The distribution of many rare species has been fragmented or altered, and existing populations often occur in habitat that is far from ideal. Superficially suitable reintroduction sites may prove to be unsuitable because of a cryptic ecological factor in soil chemistry, microhabitat or microclimate relations, absent pollinators, or mycorrhizae (Allen 1993). Second, even where appropriate conditions can be defined with some accuracy, available receptor sites that match these characteristics often do not exist. Available suitable sites may be too small or may lie on unprotected land, while land under reasonably secure management may not offer the appropriate biotic or abiotic environment. And third, sites that are ecologically suitable and protected may nonetheless fall outside of the species' known range. Or they may fall within its overall range but with no evidence that the species occurred at a particular location.

Site selection thus represents a series of tactical compromises. The process begins by determining areas encompassing tolerable variation in key biotic and abiotic parameters, which define suitable potential habitat (Figure 1). These include commonly used physical site indices (soil, available moisture, temperature regime, topographic position) as well as various

characteristics of the biotic community. This envelope of feasibility may be compared to an envelope of security, areas that meet the administrative and management criteria discussed previously. Among areas meeting these criteria, the restorationist can then select sites of known or suspected historical occurrence within a specified time frame. In this manner, potential receptor sites may be evaluated in terms of ecological, administrative, and historical suitability. Note that Figure 1 is a conceptual set diagram, not a "map" of a physical area; actual sites meeting the criteria may be patchy and dispersed across the landscape.

Three groups of sites emerge as possible reintroduction locations. Preferred sites are those that meet all three criteria. A second tier meets habitat and protection criteria but may fail (or not be demonstrated to meet) the criterion of historical occurrence. Ecologically and historically suitable sites on unprotected land would also fall into this category. Third-tier sites meet only the criterion of feasibility. Beyond this envelope, sites are neither biologically

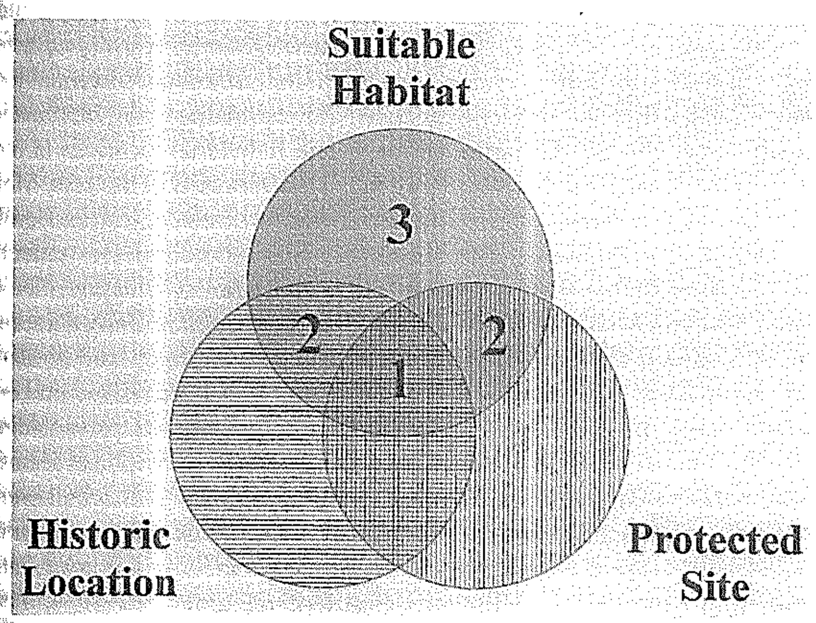

FIGURE 1. Set diagram for evaluating potential reintroduction sites. Sites may be evaluated for the degree to which they meet primary criteria for habitat suitability, protection, and historic locality. Preferred sites (1) meet all three criteria to a high degree; secondary sites (2) meet two criteria, one of which must be ecological suitability. Tertiary sites (3) pass only the test of habitat suitability. All other sites are presumed to be nonviable for both ecological and conservation purposes.

nor administratively viable for the species in question. Any proposed reintroduction site may be classified according to these criteria.

THE HISTORICAL RANGE ISSUE

Existing policies vary in their approach to site selection and historical range. The U.S. Bureau of Land Management (1992, p. 11) permits release outside of historical range "for those threatened and endangered species for which remaining historical habitat has been destroyed or otherwise rendered unsuitable." The Botanical Society for the British Isles (Birkenshaw 1991) categorically restricts reintroductions to sites within 1 kilometer of a documented locality and considers all other outplantings to be introductions. The Illinois Endangered Species Protection Board (1992, p. 1) states unequivocally that outplantings "should not extend the historic ranges of distribution, the range of habitats in which a plant species is known to have occurred, nor exceed the pre-settlement [sic] abundance of a species in a community or in the state." While such policies are a good start, they beg the essential questions of defining spatial and temporal scales at which species distributions are to be determined, and with what fineness of grain individual sites will be defined.

In the approach described previously, more weight may be assigned to feasibility and security considerations than to documentation of historical locations, provided that reintroduction is reasonably occurring within the species' overall range. The overall historical range and individual locations of most rare species remain unknown for more than a few decades in the past. Moreover, the ranges of most species have changed over ecological and evolutionary history; even current distributions may not reflect a species' potential ecological amplitude. Giant Sequoia (*Sequoiadendron giganteum*), for example, is narrowly distributed at present, but the species is planted successfully throughout California's Sierra Nevada and in Europe, New Zealand, and elsewhere in a wide range of habitats. Giant Sequoia's tertiary and quaternary distribution were widespread in North America; its present range probably represents only a portion of its potential range. The species seems to have retained genetic potential for many habitats. Finally, even when overall range is known, individual past localities within that range may be difficult to reconstruct (unless the species has a very strong association with a habitat or community type that is itself limited in distribution).

Even more problematic are species with only one or two extant populations. In such cases, historical range becomes an almost irrelevant notion, and the emphasis must shift to search to sites that are ecologically and administratively feasible (Linder 1995).

In the face of such ambiguity, it may be difficult to apply criteria of historical range and documented localities in real-world practice. As discussed in the Introduction, we recommend instead an approach based on evaluating natural variation in range, distribution, and density of populations over time,

as more realistically reflecting the continually changing "history" of any species.

Defining potential habitat is itself difficult, because even naturally occurring populations may not reflect habitat and distributional optima. As the landscape becomes increasingly modified by human actions, populations of rare species are increasingly fragmented and pushed into ecological corners that may represent the fringes, rather than the center, of their historical distribution and niche space. Consequently, a search image based on current populations may reflect poorly the optimal conditions for their reintroduction. In southern Arizona, for example, Lemon lily (*Lilium parryi*) occurs in upper-elevation stream systems with moderate stream energy and periodically high flows. California populations are genetically similar but occur mostly in lower-elevation, low-energy cienegas and wetlands—habitat that has been largely destroyed in southern Arizona. A related rare species, *L. occidentale*, occurs in Northern California and southern Oregon on a handful of sites, virtually all of which appear to have been substantially altered by decades of fire suppression (E. Guerrant, personal communication, May 1995). In the midwestern United States, many formerly widespread prairie species (such as the royal catchfly [*Silene regia*]) now persist in only a few marginal sites, which may be wetter or drier (or of different soil composition or community type) than previous mean values.

ECOLOGICAL CRITERIA

At the level of community structure and process, the receptor site should provide resources and opportunities for key life history requirements: dispersal, pollination, germination and establishment, mycorrhizal associations, and other mutualisms. The site should also be similar to the rare plant's native habitat in floristic and faunal composition and structure, successional stage, functional parameters, and disturbance regime (Primack, Chapter Nine). If possible, avoid sites where disruptive exotics (including pathogens) persist, even if other conditions are appropriate. For instance, Rejmanek (1989) has demonstrated that some highly modified communities are no longer invasible by rare plants (Loope and Medeiros 1994; also Pavlik, Chapter Six). Although careful matching of native habitat and receptor site ecology is preferable, in many cases this approach will be difficult to implement (Schemske 1994). Given the paucity of information about many rare species, a directly experimental approach will often be necessary, using outplanted populations on an array of sites to evaluate outcomes over time.

THE SPATIAL CONTEXT

Landscape or spatial context is also important in the selection of the receptor site. White (Chapter Three) describes the importance of selecting sites that contribute to re-establishment of natural patterns of heterogeneity at the landscape level. This may include the mosaic of successional habitats and the disturbance regime that will exist on the site.

The act of establishing a population creates a new source of propagules for the surrounding landscape. Depending on the dispersal ecology of the species, the amount of material translocated, and the characteristics of the surrounding vegetation matrix, each site-level introduction has the potential to influence the vegetation of surrounding areas.

Species characterized by metapopulation ecology require a careful definition of "site." For such species, the functional ecological unit may be a cluster of sites along a riparian corridor or among canopy gaps or edaphic islands. Each individual act of reintroduction should be intended to establish one component of this larger entity, in some cases simply filling in a gap within an existing metapopulation (Primack, Chapter Nine; Bowles and McBride, Case Study Five). Since gene flow is presumably higher among metapopulation sites than among differentiated populations, the genetic makeup of such subpopulations may be of special concern (see Guideline 9).

As with so many other aspects of biology and conservation, islands illustrate questions of spatial context intensively. The definition of a suitable landscape-scale area may be entirely bounded by a single island, even when potential habitat exists nearby. Rates of interisland migration are often unknown, and it may be difficult to ascertain if observed differences among island populations reflect significant ecotypic variation or simply chance events of dispersal, colonization, and survival. The most conservative policy is to restrict normal outplantings to the island that provided the source material (HRPRG 1992).

Whenever possible, reintroduction projects should account both for among-site or contextual factors, and for within-site criteria. Spatial and landscape characteristics of the receptor site should be compared with the species' native habitat, especially populations closest to potential reintroduction sites. Various characteristics of the landscape matrix can be evaluated, including corridors, patch configuration, buffer zones, fragmentation patterns, and watershed position (Naveh 1994).

CLIMATE CHANGE CONSIDERATIONS

Introducing some species outside of their known historical range may be a strategic hedge (or perhaps a response) to potential climate change (Kutner and Morse, Chapter Two). As vegetation zones shift, many rare species are predicted to be excluded from their current range and may face formidable dispersal barriers of both natural and anthropogenic origin. Intentional introduction outside of the envelope of known historical range could be part of the conservation response to the effects of a change in climate (Peters 1985; Peters and Lovejoy 1992).

No species can be introduced, however, outside of its current envelope of ecological feasibility, no matter how urgent the need. Over the next fifty years, changing climates may just begin to affect many plant populations, primarily populations at the ecological margins of species distributions. Uncertainties about the local manifestation of climate change may hinder our ability to predict impacts. Because the actual physical and biological limits for rare plants are often unknown, predicting the movement of their suitable habitat is necessarily a highly speculative venture. For proposed receptor sites close to known localities, it is probably safe to assume some variation in the perimeter of distribution and perhaps even the occurrence of some disjunct or relictual populations over the species' natural history. In the published literature, we find few cases where the movement of a species just beyond its currently documented distribution can be proven to violate the area of previous colonization. Examples where this may be possible would be habitats (such as high alpine areas, riparian zones, or islands) that are by nature persistently patchy and isolated for ecologically significant periods of time. However, such habitats are often also closely defined by gradients in soil, temperature, or precipitation that correspond to the envelope of potential habitat.

Following the approach recommended throughout this book, we define the reasonable test discriminating a outplanting from a biologically new event to be whether the act of reintroduction exceeds the natural range of likely dispersal events over a specified period of time. Crossing a near-absolute dispersal barrier (2,000 miles of ocean, for instance, for most terrestrial species) would in nearly all cases constitute an introduction, while crossing a low range of hills with continuity in vegetation and climate might not. Because of the uncertainty of both future climate and species niche breadth, perhaps the best way to hedge bets on survival is to plan for buffering, resilience, and migration of communities over space and time, including many experimental outplantings that will offer a cushion against attrition.

9. What is the genetic composition of the material to be reintroduced?

Genetic composition and genetic processes at different phases of population growth influence short- and long-term population viability (Kress et al. 1994; Godt et al. 1995). The most important times to consider genetic aspects, however, are when (1) selecting the site, (2) developing and outplanting source material for initial planting and for supplementing the reintroduced population (such as replanting to replace initial mortality), (3) ensuring reproductive and dispersal adequacy of the new population, and (4) designing a monitoring program (Guerrant, Chapter Eight). For excellent overviews, see Fenster and Dudash (1994) and Schemske et al. (1994).

SITE SELECTION

When considering candidate sites, match known or inferred genetic elements with those in nearby healthy populations (Fiedler and Laven, Chapter Seven). Ideally, sites should be avoided if they are surrounded by (that is, within significant gene flow distance of) populations of nonlocal genotypes or races capable of contaminating the reintroduced population. Sites should also be avoided if populations are surrounded by widespread congeners capable of swamping the reintroduced gene pool via interspecific hybridization. Conversely, if the species naturally occurs in a scattered metapopulation structure, matching or creating this structure may be important both for genetic and demographic reasons (Primack, Chapter Nine). Choosing a site that is large enough to accommodate a healthy native population may favor maintenance of genetic diversity by maintaining large effective population sizes.

SOURCE MATERIAL AND DESIGN FOR INITIAL REINTRODUCTION AND SUPPLEMENTAL PLANTINGS

The objective in selecting and developing appropriate germplasm is to establish resilient, self-sustaining populations that retain the genetic resources necessary to undergo adaptive evolutionary change (Guerrant, Chapter Eight). One effective strategy to achieve this objective might be to do whatever possible to maximize initial population growth and to minimize short- and long-term extinction probability. Two aspects are most important: (1) the genetic source of founders (and supplements) and (2) the diversity and number of genetically effective individuals (that is, effective population size). Regarding the source of founders, match germplasm to the reintroduction site by choosing native donor populations that are geographically close and ecologically similar to the reintroduction site. However, the dimensions and characteristics that define "local" are unique for each species—no direct rule applies. "Local" is ultimately determined by the size of the genetic neighborhoods of native populations, the environmental factors that condition selection gradients of genetic change over space, and the historical elements that influenced the evolution of the population's genetic structure (DeMauro 1994). The conservative guide is to choose donors from closest neighboring population(s) if those populations are relatively large, viable, uncontaminated (by nonlocal genotypes of the same species or interspecific hybridization), and healthy. In addition, conditions of the donor site should match the ecological conditions of the reintroduction site. If neither of these conditions applies, choose donors from more distant native populations, following the best ecological knowledge to match donor populations with similar conditions to the reintroduction site and ensuring that collection does not harm donor populations (Lesica and Allendorf 1995). Only exceptional and urgent conditions would justify using germplasm of unknown origin. Even local germplasm grown for several generations in a nursery or botanical garden may be genetically different from native local gene pools (Kitzmiller 1990; Pavlik, Nickrent, and Howald 1993; Lippett et al. 1994).

Although few valid generalities exist regarding the optimal number of donor populations to use, the conservative guide is to use one if the population is healthy as described previously. Conditions other than these might favor using a mix of several native populations as donors. (See Guerrant, Chapter Eight, for discussion.) The guiding factor is to establish a population with appropriate genetic diversity to provide raw material for adaptation to the site.

In collecting propagules from natural or cultivated populations, the goal is to maintain high effective population sizes throughout the propagation process. This is achieved by maximizing the number of distinct founding genotypes (by collecting propagules systematically throughout the donor population), maintaining equal numbers of propagules from each founder through nursery stages to outplanting, and encouraging rapid early population growth (Lippett et al. 1994). Experience is insufficient to prescribe in general how large the founding population should be. Within obvious practical considerations, the default rule is "bigger is better" (Guerrant, Chapter Eight; Center for Plant Conservation 1991).

GENETIC CONSIDERATIONS FOR REPRODUCTION AND DISPERSAL

The general guideline for safeguarding genetic diversity during this phase of the reintroduction is to mimic the natural life-history characteristics of the rare species, including its pattern of dispersal (Primack, Chapter Nine). This minimizing inbreeding (except for inbreeders) and favoring natural dispersal patterns (numbers of propagules, dispersal distance, vectors). By so doing, genetic diversity is most likely to be maintained, and natural genetic structure will evolve in the new population. There are two ways to minimize inbreeding: (1) plant diverse genotypes scattered systematically over the planting site (that is, don't plant in groups of clones or close relatives such as selfed individuals, full sibs, or inbred propagules), and (2) plant with high stocking density to promote abundant cross-fertilization.

MONITORING GENETIC VARIATION

Genetic monitoring is a research field in itself and not straightforward in design, standards, or interpretation. The conceptual standard for genetic monitoring is that genetic diversity be adequate to maintain population viability (demographic stability and growth) and sustainability (long-term adaptability and resilience to change). In practice, determining how much genetic diversity is enough to meet these goals is nearly impossible, even for species that are well understood genetically. Further, teasing apart genetic diversity from other factors that affect demographic stability and population viability is, with present knowledge and techniques, extremely difficult. The best practical guideline to ensure genetic integrity during the monitoring period is to begin with a good baseline genetic profile of the material to be introduced to the site and then over time to monitor levels and trends in overall genetic diversity. Other proxy data (such as demographic attributes and life-history parameters of population growth and viability) can be used to help interpret genetic status. If results from monitoring these traits indicate a population decline or a significant drop in viability, and genetic diversity similarly has dropped precipitously, then genetic factors may be contributing to the decline. Conversely, if overall levels of genetic diversity are maintained or increase gradually, and the population is viable and healthy, it can be assumed cautiously that genetic diversity is adequate. Abrupt changes in allele frequencies (that is, the appearance of unique alleles) may indicate gene contamination or interspecific hybridization and should be followed by careful inspection of neighboring populations.

The choice of genetic traits to monitor is debatable. *Marker traits* (allozymes, DNA), although expensive, are relatively quick to measure and indicate actual levels of genetic diversity. They are, however, almost always ambiguously and indirectly related to genetic loci controlling adaptive traits, which are usually the traits of interest in monitoring population viability. If used only to measure overall genetic diversity, and used in conjunction with other proxies, markers are probably best. *Quantitative traits* (such as plant height, fruit size, or seed weight) are of more direct interest. The role of genetic diversity in adaptive fitness requires either that the reintroduced population be originally designed as a common-garden test (rarely desirable for other reasons) or that common-garden experiments be undertaken periodically on propagules from the reintroduced population. An appropriate sampling design for monitoring genetic diversity should be followed regardless of genetic traits monitored (Center for Plant Conservation 1991).

10. How will the founding population be structured to favor demographic persistence and stability?

An immediate goal for reintroduction is to establish robust, self-sufficient, and reproductively effective populations. Attention must be paid to the early phases of a reintroduction, especially the demographic consequences of the outplanting materials chosen and the dynamics of early establishment and growth (Guerrant, Chapter Eight; Primack, Chapter Nine). Two important demographic goals are to maximize population growth and to avoid local extinction. Both stage-class (seed, seedling, juvenile, and so on) and age-class of founders affect subsequent population growth and extinction probabilities. How they specifically affect these values depends on the life history of the species. On average, simulations and empirical evidence show that populations expierence lower extinctions when mature plants are outplanted, rather than seeds or very small seedlings. Although larger plants appear to decrease extinction probabilities, the greatest gain in avoiding extinction is between seeds or seedlings and small plants. Therefore, when the goal is not mere persistence but rapid population growth, using the largest outplants appears to be best, since population growth rate generally increases continuously as plant size increases (Guerrant, Chapter Eight). These guidelines are based on simulation results and only tentatively offered practically, since there are some potential downsides of outplanting mature plants: selection under garden conditions; the time required to find out if the seed-to-seedling hurdle can be passed at a given site (McDonald, Case Study Three); and the lack of experimental or quantitative analysis of a significant demographic event (seed to seedling). Putting out seeds, mixed with whole plants, may circumvent these difficulties, especially if large numbers of seed are available (Guerrant, Chapter Eight).

There is no simple, standard answer to the question of how many plants are enough to constitute a viable founder population. Unfortunately, the actual critical values—minimum viable population size, founder population size to avoid an early extinction event, or even the age structure of a normal population—are unknown and probably nearly unknowable because of our uncertainty about future environments (see Pavlik, Chapter Six). The practitioner/researcher will have to explore the literature on the target species or congeners and make a series of best guesses based on the size and design of natural populations that appear ecologically comparable (Ruggiero et al. 1994; Schemske et al. 1994). We offer guidance here on some of the key questions that such exploration should attempt to address.

Founder Population Size

Reintroduction practitioners should become familiar with the literature applying analysis of minimum viable populations, population viability and vulnerability, founder events, and demographic stochasticity to problems in conservation biology (Pavlik, Chapter Six; Guerrant, Chapter Eight; Shaffer 1981; Gilpin and Soulé 1986; Soulé 1987; Menges 1991a, 1992). This work confirms both theoretically and empirically that, other things being equal, small populations are at greater risk of local extirpation due to demographic fluctuations than are large populations. However, the actual details about numbers are almost entirely a matter of speculation for most species, especially rare ones. Mathematically, predicted persistence time appears to be a function of the power of founding population size, but it is influenced as well by predicted growth rates and many other factors (Menges 1990). Persistence over time is also a function of the effective population size, with regard to the maintenance of genetic variability and reproductive processes (Guerrant, Chapter Eight; Lande and Barrowclough 1987; Menges 1991b; Ryman and Lairke 1991).

Moreover, the true effects of demographic oscillations are only evident over many generations. Most demographic extinctions have been simulated in ecological models of populations over tens or hundreds of generations down the line. Chance extinction models based on birth-and-death processes (Goodman 1987) permit population size and growth rates to be correlated probabilistically with time (or number of generations) to extinction. Alternatively, the same parameters can be used to estimate the probability of population persistence.

Since minimum viable population values for individual species may be correlated with various life-history attributes, some insight may be gained by comparing the target species to others with similar characteristics (Pavlik, Chapter Six).

Population Growth, Recruitment, and Survivorship

Although rapid population growth is theoretically favored, management techniques to achieve it (such as fertilization or intensive culture) may not promote population sustainability. Most models suggest that population-level persistence can be enhanced by very large population sizes, high survivorship, or high growth rates; persistence probability can be expressed mathematically as a function of these and other factors. Ideally, the values for growth, recruitment, and survivorship will come from studies of natural populations or closely comparable congeners. In the absence of such baseline data in natural populations, the restorationist must estimate projected growth and survivorship and then monitor the outcomes closely.

Such trials may provide the best information over the long term if the project includes a series of successive outplantings over a period of years. In only a few cases will an initial outplanting be successful, in the sense of establishing a viable, self-reproducing population on the first try. The probability of extinction is very high in any given trial, especially those at the beginning of a reintroduction program. As such, the best strategy for achieving a stable population may be to treat the first attempts as ministudies that will provide qualitative information about survivorship, recruitment, growth rates, competitive interactions, pollination success, and other parameters.

If mature plants are placed on the site, every individual should be marked and mapped to allow survivorship, recruitment, and growth to be tracked accurately in subsequent years. If seed are used, a planting record and map should indicate the amount of material released and its location. Over a period of several years, these methods will reveal a site-specific pattern of establishment and growth for the species. Survivorship studies must be continued for a long enough period to include the natural range of variation in weather and ideally some variation in related environmental factors (such as stream flow for riparian endemics). Laboratory-derived seed germination rates can provide a yardstick for evaluating response on the outplanting site, taking into account that germination rates can be an order of magnitude lower in the field due to suboptimal germination conditions, seed predation, and competition.

Size and Stage Structure of the Reintroduced Population

By itself, stage structure does not tell us all we need to know about a population's health. Species and populations may have a characteristic age structure reflecting multiple demographic, reproductive, and life-history factors, although many species are highly variable. For perennial species, the stage and size structure of comparable natural populations should be observed closely in designing the reintroduction program. Should the reintroduction program attempt to replicate this age structure, or should the population be permitted to establish its own demographic equilibrium over a period of years? In evaluating model simulations, Guerrant (Chapter Eight) argues that decisions about inital stage structure should include considerations of survivorship, cost, effort, and other variables beyond "pure" demogragraphy.

The simulations show that stage structure of the founding population can influence long-term extinction risk significantly, although the outcome depends on growth form (long-lived trees versus herbaceous perennials, and so on). The introduction of some plants of larger size classes (even relatively small or juvenile plants but one stage class only) dramatically reduced extinction risk compared to introductions using seeds. These simulation results suggest that using the largest founders practical may theoretically be the best, although using a diversity of stage classes as founders and multiple introductions may be safest practically. To the extent possible, reintroduction would treat these factors experimentally and track success by size class.

The multiyear outplanting approach r ecommended here will introduce a rudimentary degree of age structure diversity to the population. However, the resulting structure may not approximate the age-class distribution found in existing populations. In some cases, it may be possible to introduce literally a multiple age-class population by outplanting a combination of seeds, young rooted cuttings, and mature individuals of various sizes. Such an approach may be limited by the availability of material or time constraints for project completion, but if resources permit it may be worth considering (very little empirical work with this approach has been conducted to date). Guerrant (Chapter Eight) discusses in detail the relative merits of outplanting various ages and types of plant material (see also Primack, Chapter Nine).

11. Are essential ecological processes intact at the site? If not, how will they be established?

Since an ultimate goal of reintroduction is not simply recovery of the rare plant but restoration of the ecological community, attention should be paid over time to ecosystem processes (White, Chapter Three; Sutter, Chapter Ten). In preceding sections, we have touched on several ecological processes important for successful reintroduction: interactions with ecological associates, symbionts, and mutualists such as mycorrhizae and pollinators; flower and fruit production; seed dispersal; gene-flow (within and among populations); disturbance processes (fire, flooding); and restoration of habitat relationships (Johnson, Case Study Six). Management actions for the reintroduced population should be adapted to promote key ecosystem processes such as nutrient cycling, disturbance and hydrologic regimes, watershed protection, wildlife corridors, and so on (Thomas 1994).

Pollination

Beyond the first outplanted generation of seeds or growing plants successful reproduction is vital to long-term success (Bond 1995; Weller 1994; Sipes and Tepedino 1995). Despite the obvious and fundamental importance of successful reproduction to persistence of the population, remarkably few reintroduction projects include any conscious effort to ensure that pollination can occur. For the many species that require animal vectors for pollen, the project will be short lived or will require hand-pollination, as does *Brighamia rockii* on Molokai, Hawaii (U.S. Fish and Wildlife Service 1995). If at all possible, the assistance of experts in pollination ecology for the taxon (or congeners) should be enlisted.

Dispersal

As with pollination, many plants rely on external agents to disperse propagules. In fact, there is evidence that dispersal failure accounts for at least some instances of recent decrease in range or population numbers of rare species (Primack, Chapter Nine; Primack and Miao 1992). The tendency toward ecological niche specificity in rare species amplifies the importance of dispersal. Moreover, many animal dispersal vectors (such as birds, mammals, ants, and bats) also prepare chemically for germination and may actually initiate the germination process.

In a real sense, reintroduction is an act of dispersal, at least for the first generation (Primack, Chapter Nine). Successful dispersal involves not only getting seeds (or other material) to a suitable macrosite and community but also securing a variety of microsite factors: proper planting or sowing depth, litter cover, sun/shade position, soil moisture, and other parameters. Careful attention should be given to microsite characteristics and microclimatic variation, as they may affect the early establishment phase.

Mycorrhizal Associations and Other Soil Microorganisms

The incidence and importance of root mycorrhizal symbiosis and other microorganismal interactions in rare plants are generally unknown. Allen (1993) has reviewed the role of mycorrhizae in ecological restoration efforts and concluded that their importance has probably been underestimated for long-term vigor and growth of established plants (Allen 1991; Weinbaum, Allen, and Allen in press) If reference natural populations or published literature do not provide data for the target species, the restorationist may have to look to congeners for clues. As an empirical alternative, some reintroduction projects bring soil from an existing population to inoculate the reintroduction site. While this practice may be effective, it may also introduce disease organisms or other undesirable elements into a new ecosystem; for this reason, one should be cautious about moving large amounts of soil.

Disturbance

Few questions in ecology are as complex and controversial as the influence of disturbance on the distribution of species and communities. Many of these decisions will be delimited by the site itself, either because of its biological characteristics or land management regime (White, Chapter Three; Fielder and Laven, Chapter Seven; Guideline 8).

A guiding philosophy must be to understand and work with the dynamic nature of natural processes (Primack, Chapter Nine). Reintroduction should accommodate the reality of short- to medium-term (successional) changes, episodic processes (such as disturbance events), and long-term trajectories (climate change) and accept the stochastic nature of these dynamics (Falk 1990). Changes in landscape patterns due to human settlement and management (such as fragmentation) may have important implications for natural processes within the reintroduced population. In some cases, managers may have to intercede (by artificial pollination or dispersal, by clearing, or by prescribed burns) to promote important processes that are blocked due to highly altered landscape conditions.

Disturbance dynamics at a very fine spatial scale affect many of the more intimate aspects of population-level reintroduction. Many species have seeds that germinate only after fire scarification or contact with damp soil. Post-disturbance processes can directly affect germination and growth rates and can alter profoundly the competitive interactions with other species on the site. Reintroduction is not simply a matter of bringing plant material to a site and then walking away; the disturbance regime is an important determinant of the pattern of distribution of species and communities on the landscape and on the long-term viability of the reintroduced population (Pavlovic 1994).

Recruitment and establishment of new individuals are critical measures of success for a reintroduced population (Pavlik, Chapter Six; Primack, Chapter Nine; Sutter, Chapter Ten). Mimicking known or inferred natural processes for the individual species is again the best guide: yearly recruitment is vital for annuals, while recruitment in long-lived perennials is often more sporadic. Since recruitment depends not only on adequate numbers of sound seeds but safe sites for germination and establishment, it is important to ensure that the reintroduction area can sustain disturbances that provide safe sites (Sutter, Chapter Ten).

Natural disturbances, such as fire, floods, windfalls, and insect and disease outbreaks often create gaps or areas of preferred habitat and should be permitted on the reintroduction site. Suppression of these processes, or introduction of artificial disturbance processes, may detrimentally affect creation of safe seed and seedling sites and thus inhibit recruitment and establishment. This illustrates the importance of ongoing habitat management (Guideline 7) even for single-species reintroductions (Gordon, Case Study Four).

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