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Environmental conservation and restoration ecology: two facets of the same problem

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Restoration ecology has often been regarded as a subordinate component of conservation biology and yet the two disciplines differ from each other. Conservation aims at staving off extinction, i.e. preserving ecological structures and services which still exist, however endangered they may be. On the other hand, the principal objective of restoration is re-building ecological structures and services that have been destroyed. The most distinct focus of conservation is on population response to exploitation, whereas restoration is principally concerned with over-exploited sites and landscapes in which communities/ecosystems are to be re-built. Conservation aims at preserving as many species as possible; on the other hand, the biodiversity approach in restoration may be addressed on three levels viz. 1) initial species diversity, 2) post-restoration increase of diversity via spontaneous species immigration, and 3) age-state diversity of developing plant cover.

The conceptual framework in conservation biology differs from that in restoration ecology. The two basic paradigms used in conservation biology are 1) small-population paradigm and 2) declining-population paradigm, and one of its useful concepts is population viability assessment (PVA). The two principal paradigms used in restoration ecology are 1) nature-in-balance paradigm and 2) nature-in-flux paradigm. Interfaces between conservation and restoration may be recognized when e.g., recovery strategies for threatened species include habitat/ecosystem restoration, or when population processes in non-threatened species are studied to verify their usefulness as restoration material.

Integration of species and ecosystem approaches is already recognizable in ecology. It is to be hoped that in future conservation and restoration become integrated components of ecosystem management, but for the time being they remain two different facets of the same problem which is the negative human impact upon environment.

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There was never a more important time for ecologists to speak out about applied, socially relevant environmental issues (O'Neill and Attiwill 1997, Lubchenco 1998). However, the use and interpretation of terminology and concepts has been arbitrary and inconsistent, and discussion of these matters occasionally denegerates into arguments over semantics (Goldstein 1998).

For any area of study, clarity of terminology is essential to the establishment of a rigorous framework in which both theoretical and empirical work can be placed (Gaston 1994). This need is very clear when conservation biology and restoration ecology are being discussed, because there seems to be some confusion. In particular, restoration ecology is often considered to be a subordinate part of conser-

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vation biology, and its specific features are not always recognized in spite of the considerable "maturation" the discipline has undergone in recent years.

Conservation and restoration are not alternatives, and neither are they synonyms. I propose, thus, to define the main focus of either respective field, and also to recognize some interface areas. This essay provides insights from terrestrial plant ecology.

Focus points and conceptual issues

The difference between conservation biology and restoration ecology may be recognized when the following questions are asked: 1) the when-questions: when is conservation called for? Which situations require restoration? 2) the what-questions: what should be conserved? What should be restored?

Some time ago, Michael Soulé rightly proclaimed conservation biology as a "crisis-oriented" discipline. Considered along the same line of thinking, restoration ecology is an "after damage" discipline. The obvious difference between decline and destruction is further stressed by some key-words applicable to either of the two fields. Conservation key-words such as decline, structure reinforcing, process improvement refer to a "five to midnight"-situation, whereas restoration key-words: destruction, structure rebuilding, process initiation indicate a "five (fifteen?) past midnight"-situation.

Conservation has started and still mainly is focusing on species i.e. populations, usually ones that are rare or whose abundance is rapidly declining esp. where man has been the cause. Hence the emphasis on patchiness of the population rather than that of the environment, which means patch theory rather than patch dynamics (Wiens 1997). On the other hand, restoration started with and still is focusing on community/ecosystem: new populations are created mostly as parts of new communities. Also, the first step towards sustainability of restored area is the restoration of ecosystem function (Bradshaw 1996).

Another aspect which enables us to recognize the difference in central focus between conservation biology and restoration ecology is exploitation. Conservation biology is obviously concerned with population responses to exploitation; sometimes, this central focus may be related even to a sub-population level, e.g., to one gender only (Caro 1999); for instance, we all are familiar with the dramatic impact of heavy ivory poaching on male African elephant *Loxondata africana*. On the other hand, restoration ecology is principally concerned with over-exploited sites and landscapes in which structure and function of community and ecosystem has been destroyed.

Throughout the world in conservation biology biodiversity is considered to be "as many species as possible should be preserved". The biodiversity approach in restoration ecology is more structured; this issue will be addressed in a further part of this paper.

The differences in the principal target between conservation biology and restoration ecology influenced the development and use of a conceptual framework in either discipline. The two basic paradigms used in conservation biology are 1) small-population paradigm and 2) declining-population paradigm. The first paradigm refers to hazards that are particularly grave to small populations, especially stochasticity, both genetic and environmental. The second paradigm deals mostly with decreasing population size and its persistence (see e.g., Holsinger 1995, Holsinger and Vitt 1997). The aim of long-term conservation should accordingly be increasing the size of endangered populations and reversal of the deterministic persistence threats (Caughley 1994).

The focus on population in conservation biology has resulted in the development of several useful concepts. One of them is population viability analysis (PVA) to estimate risk of extinction of a population. From this estimation, the size of a minimum viable population (MVP) may be derived (see e.g., Menges 1991). However, the PVAs have limitations because they refer to single species, do not include considerations on all possible risk sources, and above all because they prognose future developments solely on the basis of current conditions. They may also be unsuited to endangered species management (Ralls and Taylor 1997).

As far as restoration ecology is concerned, the main paradigms used are 1) stability of ecosystem/community (nature-in-balance paradigm) or 2) pattern of ecosystem changes influenced by its past (nature-in-flux paradigm). The balance-of-nature paradigm focuses on the theory of climatic succession (Clements 1916, Egler 1954, Drury and Nisbet 1973, Connell and Slayter 1977). In this theory, temporal and spatial heterogeneity and also site history were relegated to less important, even inconsequential considerations of community change. This was a deterministic equilibrium theory: some species were "better" than others; competition was thus considered the main force structuring the community.

The modern paradigm on nature in flux is based on ecological heterogeneity in space and time. Nature does not tend toward balance but is in a continual state of change (Pickett et al. 1992, 1997). This is a non-equilibrium theory focusing on patchiness, and also on contingency (= importance of history); it points out as well the importance of stochastic events (Pickett et al. 1994, Parker and Pickett 1997). As a result, alternative pathways to alternative end points exist within a single system (Gould 1986, 1989). All species are interesting and unique on various niche axes. Also, there is a growing interest in plant interactions other than competition; in particular, positive interactions are increasingly studied (e.g., Cody 1993, Callaway and Walker 1997, Urbanska 1997b).

I have argued elsewhere that it would be most helpful to distinguish between succession as pattern (pathway) and succession as process or mechanism. The broader model of vegetation dynamics proposed by Pickett may apply better to restoration of very degraded, highly stressed sites than the deterministic model of succession, because it does not need to be directional, stage-wise, or terminate in a repeatable state (Urbanska 1997a). It should be kept in mind, however, that the arrival and use of the new paradigm does not imply an absolute replacement of equilibrium by non-equilibrium, but accepts equilibrial and non-equilibrial phenomena as scale-dependent. For instance, a number of patches of any community type within a landscape may be in dynamic equilibrium with other patch types, although the patches themselves are maintained by a non-equilibrium process (Fiedler et al. 1997).

Conservation biology research

In many terrestrial ecosystems, plant populations which formerly constituted well-developed, interconnected systems, have become small and isolated because of the landscape fragmentation. This reduction in population size accompanied by impaired or non-existent gene flow represents a serious danger of extinction, at least on a local scale (Saunders et al. 1991). Small population size often brings about a risk of genetic drift and the loss of genetic variation, partly via inbreeding depression (Lande and Schemske 1985, Schemske and Lande 1985, Charlesworth and Charlesworth 1987). In a longer run, the population may become extinct because it cannot withstand environmental and demographic stochasticity.

The very brief outline given above points out the importance of demography, reproductive biology, and population genetics for conservation of rare and endangered plants. It is therefore not suprising that these subjects are increasingly taken up in recent studies. For instance, the relative impacts of extrinsic and intrinsic factors on the demography were clearly demonstrated by Pavlik (1995) in the founding population of the annual endemic *Amsinckia grandiflora* in California. The influence of genetics and reproductive biology on the viability of small populations has also been well documented in the perennial *Gentiana pneumonanthe*, presently rare in the Netherlands (see e.g., Raijmann et al. 1994, Oostermeijer et al. 1994). However, some authors (e.g., Holsinger and Gottlieb 1991) rightly argue that a large part of the available information on genetic population structure of rare plants is based on data from electrophoretic surveys of soluble enzymes. These surveys are not expected to reflect the pattern of genetic diversity at loci determining ecologically relevant characters. Many rare species are likely to maintain considerable amounts of ecologically significant genetic variation such as e.g., reproductive traits and/or genes involved in defence against herbivores.

Conservation biology studies on endangered or rare plant populations should, thus, pay much attention to reproductive behaviour. In this context, I propose now to have a closer look at button wrinkleworth *Rutidosis leptorrhynchoides* (F. Muell.), an endangered perennial composite from Australia. This non-clonal, preferentially outcrossing species with a distincly disjunct distribution is currently listed as endangered for the whole Australia (Gullan et al. 1990). It is missing in the biological reserve system in the Victoria state (Scarlett and Parsons 1990), and rather poorly conserved in other areas (Briggs and Leigh 1990). Practically the whole distribution area of *R. leptorrhynchoides* is now intensively influenced by farming or urban development. Recruitment in the populations of this inter-tussock species of grassy *Eucalyptus* woodlands and grasslands is apparently limited by seed production (Morgan 1995 a, b); gap availability is important, too, and appropriate measures for population management are recommended (Morgan 1997). A presumed individual life-span of *R. leptorrhynchoides* is 10–15 yr (Scarlett and Parson 1990, Bartley unpubl.). Data on molecular genetics of the species are not available to date.

The very recent, brilliant study of Morgan (1999) deals with effects of population size and density on seed production and germinability in *R. leptorrhynchoides*. The eight populations studied by the author were assigned to two categories: 1) small sparse populations including < 30 reproducing individuals, and 2) large dense populations consisting of 500 to over 5000 flowering plants. Maternal features of plants i.e., total number of stems and percentage of stems flowering did not differ generally among sites, and the differences between populations related to maternal plant size were marginal at $p = 0.052$. Seed set proved to be significantly associated with population size and density because small populations produced only about half as many seeds per head as did the large dense populations. On the other hand, between-population differences in seed germinability were not significant.

The results of Morgan (1999) suggest that in many years small populations of *R. leptorrhynchoides* face a handicap due to low seed production. The conclusions of the author are as follows: 1) the problem for short-term conservation of the small populations of *R. leptorrhynchoides* appears to be primarily demographic: habitat preservation is important but it may be not enough to conserve the populations if seed set and the subsequent recruitment rates are not sufficient to replace senescing adults 2) since population density seems to be an important determinant of reproductive success, it may be a more relevant criterion of population size than the total number of individuals.

The second conclusion of Morgan (1999) is truly intriguing because densely clustered individuals in outcrossing populations are likely to be at least partly full- or half-sibs, so that the genetic structure of population may change in time towards a more limited number of alleles. However, a positive influence of local density of conspecifics on reproductive success was reported by few other authors (e.g., Roll et al. 1997). It would certainly be worthwhile to address this issue in further studies including other endangered plant species. A particular attention should be paid to conspecific nurse effects at the establishment phase (Urbanska 1997b, Wied and Galen 1998, Weltzin and McPherson 1999).

Restoration ecology research

Degraded sites persist as scars on landscapes. Restoration aims to return these sites to self-sustaining state via initiation of ecosystem processes; the establishment of functional plant communities represents a decisive step towards this goal. The crucial elements in a successful restoration are thus: 1) a proper assessment of damage, 2) selection of plant material best-suited to restoration purposes, and 3) appropriate site manipulations involved in the implementation of restoration schemes (Urbanska 1986, 1990, Chambers et al. 1987, Urbanska and Hasler 1992, Densmore 1994).

Restoration thus includes amelioration of site conditions, re-introduction of plants and optimizing their performance. Site manipulations range from relatively simple interventions as e.g., a single, moderate fertilizer application (Tschurr 1992), to extensive re-contouring or massive amendment of soil texture (Karle and Densmore 1994, Chambers 1997, White and Fuller 1999); microclimatic considerations and understanding of hydrological conditions should not be neglected either. This problem is particularly acute in sites which not only are strongly degraded but also situated in extreme physical environments. For instance, severe life conditions in alpine ecosystems reinforce the effect of disturbance and increase the difficulty of restoration (Urbanska 1995, 1997b, Chambers 1997).

Since biodiversity maintenance represents an essential ecosystem service, restoration ecology address biodiversity on three levels strongly related to temporal aspects:

a) Initial diversity. The plant material used in restoration typically consists of numerous species, representing different growth forms and different seral stages. This is a rather consistent feature of restoration, not related to the site type or geographic area. For instance, Robinson and Handel (1993) planted 17 different trees and shrubs in landfill restoration, whereas two different transplant mixes used by Francis and Morton (1995) in restoration of woodland herb layer included 10 forbs each. The number of species used as transplants in our restoration trials on high-alpine ski runs mostly ranged from 12 to 19; except for trials which specifically involved one particular life form (Urbanska et al. 1987, Hasler 1992), the species represented mixtures of graminoids, legumes and forbs. Of the 41 species used in an extensive restoration of high-altitude road area, 26 were forbs, 10 – graminoids, and 5 – trees and/or shrubs (Lange and Lapp 1999).

b) Post-restoration increase in diversity via spontaneous immigration. The biodiversity issue in restoration is not limited to the initial introduction of various species: equally important are the procedures aiming at biodiversity increase via spontaneous immigration of further species from neighbouring communities. Reliable information on extant natural vegetation in the area in which sites to be restored are located is, therefore, very helpful, and should include both the actual species inventories and also data on dispersal. Assessment of dispersal vectors and possible traveling distances is particularly valuable to planning and implementation of restoration schemes because it may enhance seed input by use of relatively simple methods. In high-alpine areas, seeds are mostly dispersed by wind, thus seed entrapment is needed. The use of biodegradable erosion blankets proved to be an effective strategy in this respect: in some of our trials, at least 13 immigrant species were recorded already one year after restoration (Tschurr 1992). Trees and shrubs planted on a closed landfill by Robinson and Handel (1993) attracted birds which brought seeds of fleshy-fruited species from nearby woodland fringes; in this way, the initially introduced plant community became enriched with 20 new species one year after restoration.

c) Age-state diversity in developing plant cover. Vegetation in a restored site should include not only various species and various life forms, but also various developmental stages i.e. age-state classes. One of the important restoration tactics is therefore enhancement of reproduction by seed in situ. Contrary to seeding, a local use of container-grown transplants enables the risk-exposed stages of germination and establishment to be circumvented; also, grown transplants usually reach their reproductive phase earlier than individuals recruited from seed. High-alpine plants, for instance, do not begin reproduction by seed before they are at least three years old (see e.g., Schütz 1988); on the other hand, clonal transplants produced from adults harvested in the wild may flower within the first year after restoration and sometimes are already flowering even before they are brought up to the restoration site (Urbanska et al. 1987, Hasler 1992). The production of seeds and seed rain alone do not suffice to ensure recruitment; safe sites which promote establishment should therefore be provided in each restoration scheme (Urbanska 1997b). The age-state diversity in restored sites may be assessed by demographic monitoring of whole stands. In such monitoring schemes, species do not need to be determined; instead, a few broad demographic categories, e.g., seedlings, juveniles, reproducing, and non-reproducing adults, may be used (Urbanska 1994, 1995).

Interfaces of conservation and restoration

Although conservation and restoration mostly differ from each other as to their principal focus, there are some situations which represent interfaces between the two disciplines.

The ecosystem has received little explicit recognition as a conservation goal, and the dialogue has been almost exclusively in the context of species conservation (Rogers 1996); in this context, habitat was to be preserved as the home of a given species (= the species "address"). Interest in preserving ecosystem function has developed only recently as a part of shift to ecosystem management. There still is no clear consensus on the subject (Yaffee 1999). In a few recent studies, however, habitat/ecosystem restoration has been given much attention when recovery plans for threatened species were made. Another interface between conservation and restoration may be recognized when single-species-oriented studies are carried out to verify usefulness of particular species as restoration material. In such cases, however, plant species are not threatened although they may be locally rare. To illustrate these interfaces, I propose to consider two examples.

Habitat/ecosystem restoration has been given an important place in recovery plan for Pitcher's thistle *Cirsium pitcheri* (Torr.) T. and G., listed as threatened species in the United States and Canada (Harrison 1988).The non-clonal composite is rather slowly maturing and monocarpic. It is predominantly outcrossing with a dispersal distance of ca 4 m from the mother plant (Keddy and Keddy 1984, Ziemer 1989). Genetic diversity of *C. pitcheri* is generally low (Loveless and Hamrick 1988). The persistence of interacting populations depends on the occurrence of structurally variable, dynamic dune landscapes within the Great Lakes area where population structure of *C. pitcheri* widely ranges under influence of local environmental factors (McEachern 1992).

The exemplary plan of recovery worked out for the species (Pavlovic et al. 1993, McEachern et al. 1995) has been based on metapopulation theory which links landscape processes with population dynamics. Population dynamics were considered on two scale levels viz. the landscape (= metapopulation) and a local habitat (= population). Depending on particular situation, recovery goals included not only habitat identification and protection, but also restoration of natural shoreline dynamics or dune systems, and also visitor use control. On the other hand, work at the population scale included augmentation to increase population size, and also manipulations improving individual reproductive vigour as well protection from grazers. New populations were also created.

The case study of *C. pitcheri* represents a very instructive example of interface between conservation and restoration since it beautifully links population processes with environmental damage. The fine distinction between two land use types viz. the one that threatens population, and the one which destroys habitat and community, clearly demonstrates the borderline at which conservation does not work anymore and restoration is required.

Let's consider now a population study on bulbine lily *Bulbine bulbosa* used in urban restoration in SE Australia (Hitchmouth et al. 1996). The attractive species is widely distributed in grasslands and open woodlands. It produces rather heavy, apparently non-dormant seeds; seedlings are vigorous. Established plants reach reproductive phase after 6–9 months of active growth (Hitchmouth et al. 1989). The effect of gap width and turf type on establishment of bulbine lily in grassy swards was studied experimentally in 1996 by Hitchmouth and co-authors. Pot-grown juveniles were planted in two types of turf 1) a native low-productive *Danthonia setacea* , and 2) tall-growing introduced *Festuca arundinacea*. Prior to the planting, gaps of various width were made; some of them included subterranean root barriers. Site manipulations carried out at various dates included irrigation, weeding, and clipping or cutting followed by removal of the cut biomass. *Bulbine bulbosa* established very well in the *Danthonia* turf, and no gap specifications were important for practical purposes. However, in the highly competitive turf of *Festuca*, successful establishment and growth depended on gaps > 200 mm. According to the authors, these differences may be related to phenological differences resulting in competition for light, and also to competition for water and nutrients. Although the study was of relatively short duration, the Australian authors have been able to provide suggestions helpful for use of *B. bulbosa* as restoration material. A more general suggestion of the authors is that tolerance for shade and relative growth rate should be considered generally when decisions are made about use of indigenous forbs in restoration schemes.

The case study of *B. bulbosa* clearly shows that knowledge of species behaviour is very valuable for restoration purposes. Studies on population or community processes carried out in the context of restoration ecology are not oriented towards conservation of those populations or communities: they should help with choice of the best-suited plants because selection of restoration material which follows a "trial and error" system may come very expensive. They are also very useful in assessment of restoration success.

Concluding remarks

Restoration is still frequently identified with conservation. This should be clarified because conservation is focused on preserving ecological structures and services which still exist, however endangered they may be. On the other hand, restoration is focused on re-creating ecological structures and services that have been destroyed or irreversibly impaired. Not all conservationists are automatically qualified to be restorationists.

Integration of species and ecosystem approaches is already recognizable in ecology (Jones et al. 1993, Jones and Lawton 1995). Habitat loss is dramatically increasing nowadays. It is accordingly to be hoped that conservation of endangered ecosystems which include numerous endangered species will in time become the generally acceptable approach. It will then become possible to include conservation and restoration into ecosystem management as its integrated components. For the time being, however, they remain two different facets of the same problem which is the negative human impact upon environment.

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