

Conservation utility of botanic garden living collections: Setting a strategy and appropriate methodology

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ABSTRACT

In the realities of the modern world, when the natural habitat is rapidly disappearing and the number of imperiled plants is constantly growing, ex situ conservation is gaining importance. To meet this challenge, botanic gardens need to revise both their strategic goals and their methodologies to achieve the new goals. This paper proposes a strategy for the management of threatened plants in living collections, which includes setting regional conservation priorities for the species, creation of genetically representative collections for the high priority species, and usage of these collections in in situ actions. In this strategy, the value of existing and future species living collections for conservation is determined by the species' conservation status and how well the accessions represent their natural genetic variation.

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1. Potential role of botanic gardens in conservation

A traditional botanic garden (including an arboretum) is 'a place with an orderly, documented, labeled, collection of living plants, that is open to the general public, with collections used principally for research and education' (Watson et al., 1993). With time, this initial scope has been broadened and started to include conservation issues, such as preservation of threatened plant species (Simmons et al., 1976; Raven, 1981; Heywood, 1989; Glowka et al., 1994; Wyse Jackson, 1997), although investment in creating and maintaining ex situ collections of wild species has been neglected by many countries apart from some material of crop wild relatives in crop genebanks (Heywood, 2009). While for wild plants, ex situ conservation has long been regarded as subsidiary to in situ conservation, in the agricultural sector ex situ (seed banks or on farm) was the primary conservation strategy and in situ was not recognized formally until 1996 (FAO, 1996). Not surprisingly, most of the ex situ protocols about sampling, gene bank standards and storage techniques were prepared by the agricultural sector and then adapted by the wild plant sector, notably by botanic gardens.

Nowadays, the Convention on Biological Diversity (CBD), although not referring to botanic gardens explicitly, recognizes the

value of ex situ conservation, undertaken "preferably in the country of origin" and as a support to the "recovery and rehabilitation of threatened species and for their reintroduction into their natural habitats" (<https://www.cbd.int/convention/text/default.shtml>). The IUCN Species Survival Commission policy on ex situ conservation recognized the primary goal of ex situ activities as "to help support the conservation of a threatened taxon, its genetic diversity, and its habitat" (IUCN, 2002), and later stated that "for a growing number of taxa ex situ management may play a critical role in preventing extinction as habitats continue to decline or alter and become increasingly unsuitable" (IUCN/SSC, 2014). The Global Strategy for Plant Conservation highlighted that role by setting a goal of a minimum of 75% of threatened plant species being preserved within ex situ collections, with at least 20% available for recovery and restoration (Wyse Jackson and Kennedy, 2009).

Nowadays, as was noted by Cavender et al. (2015), conservation of threatened plant species is explicitly included in the mission statements of many major botanic gardens yet few maintain ex situ collections with significant in situ conservation value. There are several reasons for the poor conservation utility of ex situ collections as discussed below.

2. Botanic gardens' limitations

Botanic gardens have numerous constraints that limit their role in conservation (reviewed in Heywood, 1999; Aplin, 2008). A need

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for regeneration of planted material leads to genetic erosion (Schoen and Brown, 2001) and divergence of ex situ collection from the wild source population over time (Ensslin et al., 2011; Rucinska and Puchalski, 2011; Lauterbach et al., 2012). Due to space limitations, and because collecting and regenerating seed samples are costly, botanic garden living collections cannot accommodate sufficiently large population samples. Usually the population samples in seed banks are small and often from fewer than 50 individuals in the wild. The small sample sizes of ex situ collections and a need for regeneration inevitably lead to genetic drift and increase in the level of inbreeding in regenerated collections (Schoen and Brown, 2001). The latter can result in fitness decline due to inbreeding depression (Havens et al., 2004; Vitt and Havens, 2004). Especially vulnerable to drift are plants with short generation times such as annuals, biennials, or short-lived monocarpic perennials, compared to long-lived perennials. Maintaining viable ex situ collection for even a small fraction of the overall genetic diversity demands significant resources, in terms of land size and budget.

Another negative consequence of cultivating plants ex situ is a potential risk of adaptation to the ex situ environment with a loss of adaptations to the original natural environment (Havens et al., 2004; Ensslin et al., 2011, 2015). Very few studies have so far addressed trait change and adaptation as a result of ex situ cultivation, and for this reason a study of Ensslin et al. (2011) requires a close attention. Ensslin et al. (2011) compared, using a common garden experiment, ex situ populations from 12 botanic gardens with five natural populations of the perennial herb *Cynoglossum officinale*. Garden populations exhibited strikingly lower seed dormancy than natural populations, and garden plants had larger inflorescences but less flowering stems than wild plants. These changes are consistent with domestication syndrome which typically includes loss of seed dormancy and the production of larger inflorescences (Zohary et al., 2012). Botanic gardens tend to act in the same way as the early farmers did, imposing unconscious selection: they usually plant out only the early germinants and collect seeds mainly from tall plants with a long main inflorescence. The resulting trait changes can be maladaptive in nature (e.g. too early germination).

Physical proximity of plants leads to a high risk of infestation by pathogens and, if they have different origin, may result in spontaneous hybridization. A risk of hybridization seriously limits utility of botanic garden ex situ collections for conservation purposes because the hybrids (non necessarily inter-specific but also between subspecies and ecotypes) may lack genetic integrity and harbor maladaptive gene combinations (Maunder et al., 2004b). To prevent these risks, sampled individuals must be maintained separately or through controlled breeding and pedigree design, which is problematic because of gardens' space limitations and high cost of maintenance.

All the aforementioned limitations of botanic gardens, however, are not the major obstacle to making living collections useful for conservation. A more important one is a lack of coherent conservation strategy with clear and agreed upon guidelines on collecting, maintenance and utilization of wild species germplasm that would replace the 'serendipitous collectionism' still commonly found in botanic gardens (Heywood, 1992). A historic tradition of regarding living collections in the same way as stamp collections, when one individual per species is considered appropriate (Cavender et al., 2015) is no longer acceptable. In a new concept of ex situ conservation, the current focus of the living collections on maintaining limited number of species representatives must be changed to one that maintains species' genetic diversity and has an explicit orientation towards recovery and reintroduction in situ.

3. A strategy of threatened plant management in living collections

Recently, Cibrian-Jaramillo et al. (2013) proposed an approach for management and use of botanic garden living collections that is aligned with in situ conservation goals. In this approach a particular species living collection is assigned a conservation value based on species risk assessment, the genetic representation of the collection in the context of the total species genetic variation, and the operational cost of maintaining a collection, with this information shared via online databases (e.g. BGCI's PlantSearch database). An approach like this one can ease coordination among botanical garden collections, as well as between collections and interested organizations pursuing in situ conservation.

Following an idea of Cibrian-Jaramillo et al. (2013) to create a unified management strategy of botanical gardens' collections, I propose a version of a strategy of management of threatened plants in living collections. This strategy includes setting regional conservation priorities (in association with the conservation agencies at a national/regional level) for the species to be conserved, creation of genetically representative collections for the high priority species and usage of these collections in in situ actions. The value of the existing and future species living collections for conservation will be a function of the species conservation status and how well the collection represents its natural genetic variation. The strategy includes the following components:

1) Regional focus

Modern plant conservation is utilizing a widely accepted operational approach called systematic conservation planning, a procedure that includes a set of steps from planning to conservation actions (Margules and Pressey, 2000; Knight et al., 2006; Sarkar and Illoldi-Rangel, 2010). The first step in this procedure is delimitation of the planning area within a spatial framework of units based on ecological or political criteria (Pressey and Bottrill, 2009; Sarkar and Illoldi-Rangel, 2010). As the distribution of species and communities rarely coincides with administrative units, delimitation using biologically defined units (i.e. ecoregions, Omernik, 1987, 1995; Omernik and Bailey, 1997) is preferred. The process of ecological land classification, which fuses the ecological concept of ecosystems with the geographic concept of regions dates back to Crowley (1967) and is one of the most important concepts in understanding and managing landscape and biodiversity. Defined on varying spatial scales from geomorphology, climate and vegetation types, ecoregions provide a consistent spatial framework for biodiversity conservation planning and management at the national and sub-national level (Cleland et al., 1997; Bryce et al., 1999; Groves et al., 2000, 2002; Bottrill et al., 2012). Conservation assessments within the framework of ecoregion units are favored by major international conservation organizations and many governmental agencies (e.g. Mittermeier et al., 1998; Ricketts, 1999; Groves et al., 2000). The Convention on Biological Diversity (<http://www.cbd.int/sp/targets/rationale/target-11/>) and the Global and European Strategies for Plant Conservation (<http://www.plants2020.net/implementing-the-gspc-targets/>) specifically direct their targets towards the effective conservation of ecoregions.

Botanic gardens have their own suite of particular environmental (first of all climatic) conditions. Formally, every botanic garden can be assigned to a particular ecoregion, i.e. a regional conservation unit. If conservation planning and implementation has a regional base, creation of botanic garden living collections must also have a regional basis, and be an integral part of the latter (Fig. 1). This will allow botanic gardens to focus predominantly on the local (i.e. having natural populations in the region) species, and

by virtue of this, to use more efficiently their limited land and financial resources (Fig. 2).

2) Prioritization

Many gardens grow threatened plants in their collections, but there are many more threatened species than the countries' conservation infrastructure including botanic gardens is able to include in conservation or recovery programmes (Maunder et al., 2004a; Havens et al., 2006). Due to the limited resources many gardens can invest in the conservation of only a few or even just one target species (Cavender et al., 2015). On the other hand, the currently existing collections can be optimized by gardens' focus on the most threatened, endemic and adapted to the local climate species. The basic resource detailing the global conservation status of plants and serving a reference for many conservation decisions is the International Union for the Conservation of Nature (IUCN) Red List. This uses a set of detailed and generally accepted criteria to evaluate the extinction risk of species and to which category they should be assigned. It is also often used for determining threat status at a national level (although many countries also employ their own criteria for setting conservation priorities for species). Ecoregion inventory of threatened species will result in a list of locally occurring species with each species assigned one of three IUCN categories (CR, EN and VU). However, prioritization of species based on the IUCN categorization alone can be misleading and is not recommended by IUCN itself, as species having the same IUCN category can dramatically differ in many important attributes. For example, one of two endangered species can be a regional endemic and the second one to have its major distribution outside the region. Or one can represent a monotypic genus and the second one a genus with several hundred closely resembling each other species. Thus, regional conservation planning must have a more nuanced procedure for ranking species by their conservation priority. A way to do this is to use, instead of a single qualitative variable (IUCN category), a composite variable combining several quantitative estimates of species rarity and vulnerability.

Selection of threatened species to be maintained in a garden can utilize the unified regional conservation planning species scoring such as one based on Freitag and Van Jaarsveld (1997) and presented in Table 1. Freitag with colleagues proposed to complement quantitative analogs of the IUCN categories (RV) by three other estimates: of relative endemicity (RE), regional occupancy (RO) and taxonomic distinctiveness (RTD) (Freitag and Van Jaarsveld, 1997). These estimates can be calculated in different ways depending on spatial scale and quality of the species occurrence data using either special units (e.g. map grid cells), area occupied or number of populations. Below are the examples to their calculation.

Relative endemicity score (RE) is a proportion of the species' total distribution range or "extent of occurrence" falling within the ecoregion. Using distribution area in km², RE will be:

$$RE = \frac{\text{Ecoregion area covered (km}^2\text{)}}{\text{Total area covered (km}^2\text{)}} \times 100$$

According to this equation, species that are increasingly restricted to the ecoregion receive increasingly larger RE scores. Species with RE scores of one have distributions entirely restricted to the ecoregion (true regional endemicity).

Regional occupancy score (RO) is a more refined estimate of regional extent of occurrence. Higher scores get regionally less common species (those with smaller areas of occupancy and smaller number of populations). For example:

$$RO = \frac{1}{\sqrt{\text{No. of populations within ecoregion}}}$$

Relative taxonomic distinctiveness score (RTD) estimates the taxonomic distinctiveness of a species. Higher scores get taxonomically more distinct taxa as contributing proportionately more to regional biodiversity (Vane-Wright et al., 1991, 1994). An equation that can be applied to all hierarchical classifications was proposed by Freitag and Van Jaarsveld (1997):

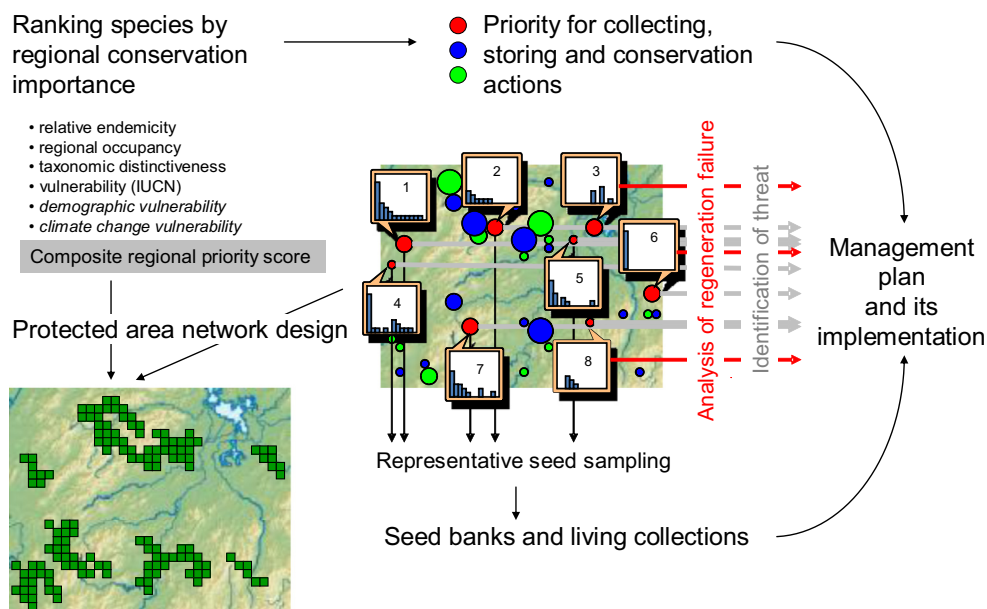


Fig. 1. A scheme of regional conservation planning. Each colored circle denotes a population of one of three species with the circle size and color corresponding to a population size and species identity, respectively. All populations of one species (in red) are provided with size class distributions. In size class distribution histograms the x and y axes are size classes and plant density per unit area, respectively. The populations 3, 6 and 8 have easily identifiable regeneration problems.

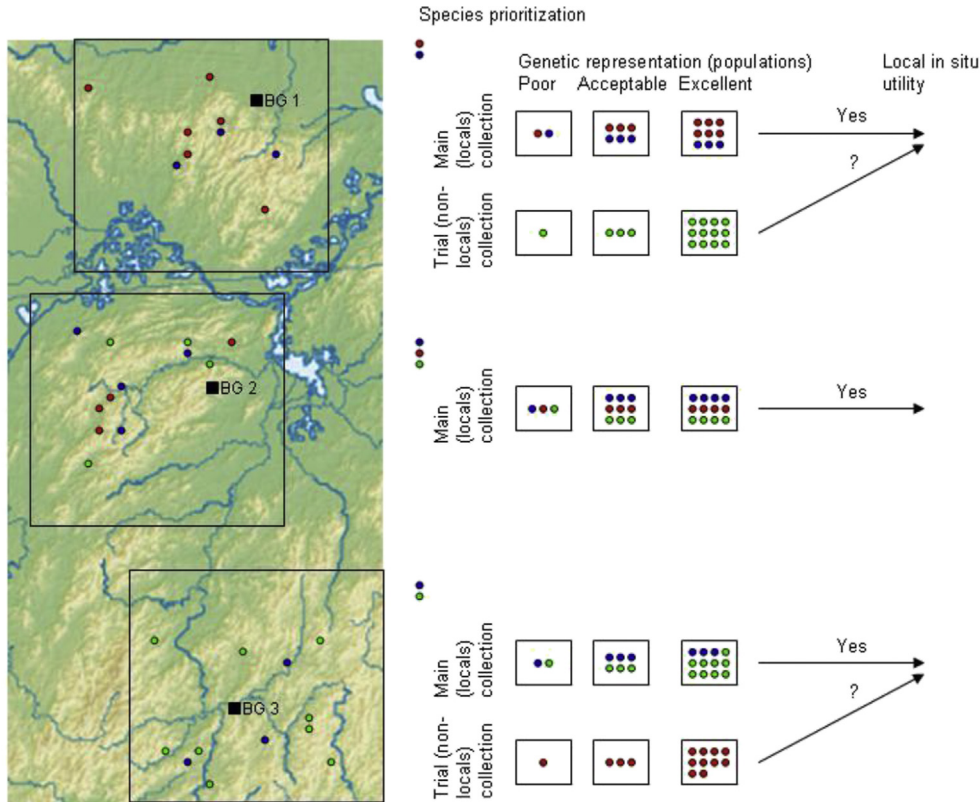


Fig. 2. Living collections in three botanic gardens (BG1–3). Colored circles denote populations of three different species in three ecoregions denoted by rectangles. Species in each ecoregion are prioritized based on a set of criteria. For the species having the highest regional priority, representative collections are created. Only collections representing all known in the region populations (excellent representation) or at least three populations (acceptable representation) can be used for in situ actions. In addition to the main collections of regional locals, trial collections of non-local threatened species can be used for regional in situ actions based on plants' performance and SDM predictions.

$$RTD = \frac{1}{\sqrt{(f \times g \times s)}}$$

where f is the number of families in the order to which the taxon belongs, g is the number of genera in the family and s is the number of species in the genus to which a particular species belongs.

The above three estimates complement the quantitative analog of the IUCN categories, *Relative vulnerability score (RV)*. The latter weightings correspond to the Red Data Book categories, i.e. “critically endangered”, “endangered”, “vulnerable” and “near threatened”, being 1.0, 0.75, 0.5 and 0.25, respectively. Species not recorded in the Red Data Book are given a score of zero.

The values for each of these four estimates range between zero and one and are similarly distributed (Freitag and Van Jaarsveld, 1997). For this reason, and for simplicity sake, Freitag and Van Jaarsveld (1997) proposed to give the four scores equal weights, to sum and average to obtain a total composite score for each regionally occurring species, referred to as regional priority score (RPS), I propose to complement the above four estimates in ranking threatened species by conservation value, by two novel ones as described below.

The IUCN Red List categories and criteria were designed to provide an explicit, objective framework for the classification of the broadest range of species according to their extinction risk. The IUCN classification uses population decline, area of species occurrence/occupancy or the number of mature individuals in populations assuming that the populations exhibit a normal demographic structure when all the life cycle stages are present. This, however, in many cases is a wrong assumption. Populations of many threatened

species lack juveniles or seedlings, or produce no seeds. Application of the IUCN rules about decrease in area of occurrence/occupancy or population size will have no meaning if mature individuals comprising a population do not reproduce or the produced seeds do not germinate. Regeneration status of the representing species populations must be known because the species future can be insured only upon successful regeneration in its populations. An example of such evaluation is an assessment of regeneration status of 68 most ecologically and economically important Bolivian forest tree species (Mostacedo and Fredericksen, 1999). In this study each species was assigned one of four categories of regeneration status and provided with identified mechanisms of poor regeneration. The latter information turned out to be invaluable for conservation planning involving these species.

Even a snap-shot of the population demographic structure (Fig. 1) can reveal regeneration problems. As a result of demographic survey, the populations can be classified into two categories: those with regeneration naturally occurring (even if limited) and those having regeneration problems (i.e. with some life cycle stages missing) (Fig. 1).

Demographic vulnerability score (DV) is the proportion of the populations of the second type to the total number of populations within the ecoregion:

$$DV = \frac{\text{No. of inviable populatins}}{\text{Total no. of populations}}$$

According to this equation, species with regeneration problems receive increasingly larger scores.

Table 1

Components of a composite regional priority score used for ranking threatened species by their conservation priority.

Component	Calculation	Definition	Relevance	Source
Relative endemicity score (RE)	$RE = \frac{\text{Ecoregion area covered (km}^2\text{)}}{\text{Total area covered (km}^2\text{)}} \times 100$	A proportion of the species' total distribution range or "extent of occurrence" falling within the ecoregion	Species that are increasingly restricted to the ecoregion receive increasingly larger scores. Species with a score of unity have distributions entirely restricted to the ecoregion (true regional endemicity)	Freitag and Van Jaarsveld 1997
Regional occupancy score (RO)	$RO = \frac{1}{\sqrt{\text{No. of populations within ecoregion}}}$	A more refined estimate of regional extent of occurrence	Higher scores get regionally less common species (those with smaller areas of occupancy and smaller number of populations)	Freitag and Van Jaarsveld 1997
Relative taxonomic distinctiveness score (RTD)	$RTD = \frac{1}{\sqrt{(f \times g \times s)}}$ where <i>f</i> is the number of families in the order, <i>g</i> is the number of genera in the family and <i>s</i> is the number of species in the genus to which a particular species belongs	An estimate of the taxonomic distinctiveness of a species	Higher scores get taxonomically more distinct taxa as contributing proportionately more to regional biodiversity	Freitag and Van Jaarsveld 1997
Relative vulnerability score (RV)	These scores are based on Red Data Book categories, i.e. "critically endangered", "endangered", "vulnerable" and "near threatened" having weightings of 1.0, 0.75, 0.5 and 0.25, respectively	A quantitative analog of the IUCN categories	Higher scores get more endangered species.	Freitag and Van Jaarsveld 1997
Demographic vulnerability score (DV)	$DV = \frac{\text{No. of inviable populations}}{\text{Total no. of populations}}$	A proportion of the populations having regeneration problems such as lack of seedlings, young plants or reproducing adults	Species with regeneration problems receive increasingly larger scores	This paper
Climate change vulnerability score (CV)	$CV = \frac{\text{Predicted range by 2080 (km}^2\text{)}}{\text{Predicted range under current climate (km}^2\text{)}}$	A proportion of the currently suitable for the species habitat within the ecoregion that will remain suitable despite climate change	Species with increasingly higher proportion of their range that will have remained within the ecoregion receive increasingly larger scores	This paper
Composite regional priority score (RPS)	$RPS = \frac{RE + RO + RV + RTD + DV + CV}{6}$			Freitag and Van Jaarsveld 1997

Another overlooked aspect of species endangerment is vulnerability to climate change. Multiple lines of evidence suggest widespread impacts of global warming on species and ecosystem processes, threatening the continued persistence of many plant populations (Thuiller et al., 2005; Grimm et al., 2013; Staudinger et al., 2013). Because of the rapidly changing temperature and precipitation patterns, species that are locally adapted to the current climatic conditions will either adapt to novel conditions, shift ranges to track the changing climate or go locally extinct.

Species climatic tolerances to a large extent determine the range of environments which a species can occupy. The latter range called environmental niche space can be evaluated through species distribution modeling (SDM) involving the use of spatially explicit environmental and species occurrence data through geographic information systems (GIS) to predict current or future areas suitable for species to grow (known as "suitable areas") (Guisan and Thuiller, 2005; Franklin, 2009; Guisan et al., 2013). A species distributional response to climate change can be determined by comparing the present and future suitable area size, and calculating these areas overlap. Range overlap is a useful measure for identification of the areas where the habitat remains suitable over time, and its estimation can be routinely conducted with Maxent (Phillips et al., 2006; Phillips and Dudik, 2008) for the current and anticipated by 2080 climate using the 19 climate data variables available from Worldclim (Hijmans et al., 2005) (<http://www.worldclim.org/>) with a resolution of 1 km².

As the new ranges of many threatened species can no longer be within the ecoregions they currently occupy, prioritization of species for long-term regional conservation planning must account for their anticipated range shifts. There will be little sense in creating a living collection of species in a botanic garden that match the ecoregional climatic conditions if these conditions are expected to become unsuitable for that species.

Climate change vulnerability score (CV) is a proportion of the currently suitable for the species habitat within the ecoregion that will remain suitable despite the climate change:

$$CV = \frac{\text{Predicted range by 2080 (km}^2\text{)}}{\text{Predicted range under current climate (km}^2\text{)}}$$

If increase in the species range is predicted, the estimate will assume a unity value. The proposed estimate has a drawback that can be used only for species that currently occur in the ecoregion, but, if found useful, can be refined to include species currently not present there.

The proposed two estimates, like the other four, range between zero and one. For simplicity, they can be utilized in the same way as was proposed by Freitag and Van Jaarsveld (1997), with the final composite score being

$$RPS = \frac{RE + RO + RV + RTD + DV + CV}{6}$$

Utilization of a unified scoring system like the above one giving the highest priority to the most endangered local endemics with regeneration problems can help to optimize not only living collections, but also seed bank collections and reserve design (although the latter two are beyond the scope of this paper) (Fig. 1).

3) Genetic diversity

Botanic garden collections must better conserve the original (i.e. present in natural populations) genetic diversity. Unfortunately, majority of the botanic garden stocks of threatened species are genetically depauperate relative to the wild founder stocks. For example, *Lysimachia minoricensis*, whilst numerically secure in cultivation, is thought to derive from a single founder, and the

cultivated stocks of *Lotus berthelotii* are self-incompatible and probably represent one clone (Maunder et al., 2001).

Thus, collecting must target as many as possible populations of a species, be done in multiple years, with sampled plants separated by a distance ensuring a high probability of sampling genetically unrelated individuals, and each population in the living collection be represented by multiple accessions and managed separately. In living collections, minimization of genetic threats can be achieved by maintaining large population sizes, providing close-to-natural growing conditions, decreasing number of generations in captivity and periodic immigration from wild populations (Havens et al., 2004, 2006).

In order to become useful for in situ actions, existing collections of threatened species that do not represent properly the species genetic diversity must be enriched by either exchange of accessions among the gardens, or by collecting in wild. This, however, must be done with caution to exclude outbreeding depression if the new germplasm comes from outside the ecoregion.

4) Redundancy

In the case of critically endangered species, more than one living collection should be established and maintained to prevent an accidental loss due to extreme climatic event or disease, or inevitably happening in small collections genetic drift. For example, *Attalea crassispata* is imperiled by habitat reduction and seed consumption in its natural environment in southwest Haiti, where it has less than 30 individuals (Timyan and Reep, 1994). Three large ex situ living collections ensure that this critically endangered species can survive in cultivation (Griffith et al., 2011). Another example is *Brighamia insignis*, functionally extinct in the wild endemic Hawaiian succulent species represented in nature by one remaining extant individual. Fortunately, it is cultivated ex situ in more than 50 botanical collections around the world (Fant et al., 2016).

If done in a range of environmental conditions, duplicating collections through sharing plant material, in addition to increased likelihood of long-term survival can also provide vital information about species climatic tolerance. Botanical gardens have largely unutilized utility for climate change research. Because one of the major goals of botanical gardens traditionally was (and still is) creation of collections of taxonomically and ecologically diverse flora, and because plants in these collections are maintained under as optimal as possible for these species conditions (mulching, weeding, fertilization, pest control), effects of climate on plants in these collections are not confounded with other effects, allowing inferences of origin by climate interactions like in the common garden experiments (Primack and Miller-Rushing, 2009). Observations on key phenological events (leaf bud burst, flowering, fruiting, leaf color changes and leaf senescence), beside mortality and reproduction, across botanical gardens representing different climatic zones, are invaluable for understanding effects of changing climate.

Often disadvantageous from a conservation biologist's point of view, features of botanic gardens that give an opportunity to plant and monitor both local and non-local species, can be turned into an advantage. If done properly, this can allow reliable inferences about the impacts of climate change on a target species. For example, there are 25 gardens located in Mexico and the southwestern USA of which eight gardens are located within the boundaries of the Mojave, Sonoran, or Chihuahuan Deserts. A reciprocal garden network using these eight desert botanic gardens alone would span a 6 °C mean annual maximum and 10 °C minimum temperature gradient. This network could be used for reciprocal planting of a number of threatened Cactaceae species in a common garden framework, and provide precious information about anticipated responses of the planted species to climate changes (Hultine et al., 2016).

5) Integration with in situ conservation

For every extirpated in wild and critically endangered species maintained in a botanic garden, there must be a program explicitly oriented towards its reintroduction (Fig. 1). This requires a close coordination with conservation agencies. The potential of using garden living collections for reintroduction and even restoration of habitats has been frequently suggested (Pavlik, 1997; Maunder et al., 2001; Hardwick et al., 2011; Cibrian-Jaramillo et al., 2013; Griffith et al., 2015), but the practical implications of this idea are modest and usually are limited to recommendations based on analysis of genetic variation preserved in species' collections (e.g. Da Silva et al., 2012; Yang et al., 2015). Nevertheless, good examples of the use of material from botanic garden living collections in various reintroduction and restoration programs exist. Over 600 rescued individuals from six extirpated wild populations of *Amorpha herbacea* var. *crenulata* maintained at Fairchild Tropical Botanic Garden were used in a translocation program (Wendelberger et al., 2008). Similarly, from a living collection maintained at Fairchild Tropical Botanic Garden, more than 200 seedlings and juvenile plants of *Pseudophoenix sargentii* were produced and used for reintroduction (Fotinos et al., 2015). The Berry Botanic Garden has been directly involved in experimental reintroductions of three endangered taxa. It supplied 1000 seedlings to reintroduce *Stephanomeria malheurensis*, was directly involved in designing and executing a reintroduction of *Lilium occidentale*, and in augmenting with seeds and young plants an existing population of *Arabis koehleri* var. *koehleri* (Guerrant and Raven, 2003). Bok Tower Gardens actively participated in producing outplants for augmentation and experimental introduction of *Ziziphus celata* (Menges et al., 2016). Outplants for translocation of *Dianthus morisianus* were produced at Botanic Gardens of Cagliari University (Fenu et al., 2016). Among the 25 rare or extinct in the wild species introduced at restored wetland sites in Switzerland, outplants for seven species were from populations maintained at Botanical Garden of Bern (Noël et al., 2011).

These examples show that a cooperation between botanic gardens and conservation agencies is possible and that botanic gardens can be an important resource for threatened species management and conservation. Hopefully, with a closer coordination with conservation agencies and participation in development of regional conservation and habitat restoration plans, botanic gardens will become an essential and well integrated part of threatened species conservation.

4. Conclusions and recommendations

The common practice of botanic gardens of growing limited samples of species for public display and research must be revised. A new strategic focus for botanic gardens should be on i) collecting and maintaining species genetic diversity, and ii) better coordination and cooperation with other botanic gardens and in situ conservation practitioners for iii) ultimate utilization of the preserved material in situ. To achieve this, and efficiently use limited resources available for botanic gardens, the following steps are necessary:

- i) planning of the living collections is done as a part of regional biodiversity conservation planning;
- ii) species to be preserved in living collections are chosen using a unified and agreed procedure for ranking species by their conservation priority;
- iii) living collections properly represent the species genetic diversity;

- iv) creation of living collections takes into account such concerns as redundancy and climate change;
- v) each living collection, wherever possible, is an integral part of a programme explicitly oriented towards species conservation, recovery and reintroduction

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