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REVIEW ARTICLE

Conservation of Central Asian plant biodiversity

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ABSTRACT

The severely threatened Central Asian (CA) flora requires a detailed conservation strategy well adapted to this region. The present review provides a detailed conservation methodology useful for every CA country that embraces *ex situ* and *in situ* approaches as inter-linked components. The proposed strategy includes the following major points: i) conservation planning gives the highest priority to the most endangered local endemics with regeneration problems; ii) these species are the focus in reserve design, monitoring and assessment of efficiency of a reserve network, and in creation of seed banks and living collections; iii) the species recovery plans include an *ex situ* component to maintain the endangered species germplasm, an *in situ* component to maintain natural populations, and a *quasi in situ* component to link them so as to obtain large quantities of planting material to be used in recovery actions, such as reinforcement and translocation; and iv) assessment of success in species recovery programs based on clearly specified objectives and criteria necessarily including evidence of regeneration in natural populations. Successful adoption and implementation of this strategy in CA requires stricter protection of nature reserves and national parks than is currently done and tighter coordination in development and implementation of the conservation plans between scientists from the five CA countries.

Key words: endangered plants, *in situ*, *ex situ*, *quasi in situ*, conservation guidelines, nature reserves, protected areas.

Introduction

Central Asia (CA) is a region with unique and rich flora that includes 9,341 taxa (Khassanov 2015). This flora is severely threatened because natural habitats throughout the region are under heavy and steadily increasing anthropogenic pressure. The latest editions of the Red Data Books of five CA countries list 387, 87, 267, 115, and 313 species of plants in Kazakhstan (Baitulin 2014), Kyrgyzstan (Shukurov 2006), Tajikistan (Rahimi et al. 2017), Turkmenistan (Annabayramov 2011), and Uzbekistan (Khassanov 2019), respectively.

Conservation of the regional biodiversity has been recognized as an important objective

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by the governments of the five CA countries, which are all members of the Convention on Biological Diversity and the UNESCO World Heritage Convention. Therefore the key 2020 Aichi Biodiversity Targets apply to them. The 2011-2020 Strategic Plan for Biodiversity sets as targets that "at least 17% of terrestrial and inland water areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes (target 11); the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained (target 12); and ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15% of degraded ecosystems (target 15)".

Unfortunately, the current state of plant conservation in the five CA countries is incompatible with the above targets and requires a methodological overhaul. Hopefully, the present paper will help in this respect to clarify the needs and objectives. The content of this paper will be relevant to every CA country because they all share not only a recent political legacy, watersheds and mountain ranges, but also the same problems of unsustainable use of natural resources, environmental degradation and loss of biodiversity.

The conservation strategy proposed in this paper is founded on several cornerstones presented in detail elsewhere (Volis 2016b,d, 2017, 2018a). They include an ecoregional approach to conservation planning and implementation; change in the current practice of passive protection to a pro-active one, including wide-scale plant introductions not only within but also outside the known species historical ranges; and establishing a tight link between *ex situ* and *in situ* conservation as integral parts of systematic conservation planning.

Systematic conservation planning, introduced by Margules & Pressey (2000), became a widely accepted operational conservation approach. This approach, in its latest modifications (Pressey & Bottrill 2009; Sarkar & Illoldi-Rangel 2010) necessarily includes delimitation of the planning area, collection of data, setting conservation goals, evaluation of the existing protected area (PA) network, design of expansion, planning and implementation of conservation actions, and long-term maintenance of biodiversity in the network. The present review will follow these steps, utilizing the above cornerstones and making necessary adjustments due to the specific realities of CA, to achieve the goal of efficient conservation of CA's unique flora.

Delineation and mapping of ecoregions

Delimitation of the planning area within a spatial framework of units based on ecological or political criteria is the first step in systematic conservation planning (Pressey & Bottrill 2009; Sarkar & Illoldi-Rangel 2010). Designing networks of conservation areas as well as decision-making and implementation of conservation actions are most efficient when conducted within the context of biologically defined units, as the distribution of species and communities rarely coincides with administrative units (Olson et al. 2001). In CA the mountain ranges, deserts and vegetation formations usually stretch across country borders. Of the 24 global terrestrial ecoregions represented in the CA countries (Olson & Dinerstein 1998), only 14 are confined to a single country (Fig. 1).

Thus, the first step of systematic conservation planning in CA should be ecological land classification, i.e. dividing the CA territory into regional conservation units based on geomorphology, climate and vegetation types. These units (ecoregions) should serve a spatial framework for ecological modeling, biodiversity management, and systematic conservation planning at the regional and national level, and preferably not be based on political criteria (i.e. country borders). The highest level of subdivision of the CA territory should be the 24 terrestrial ecoregions from the 'Global 200' list (Olson & Dinerstein 1998) represented in the CA countries, as already mentioned. A finer subdivision based on vegetation types and edaphic differences can be useful for national-level planning (Fig. 2).

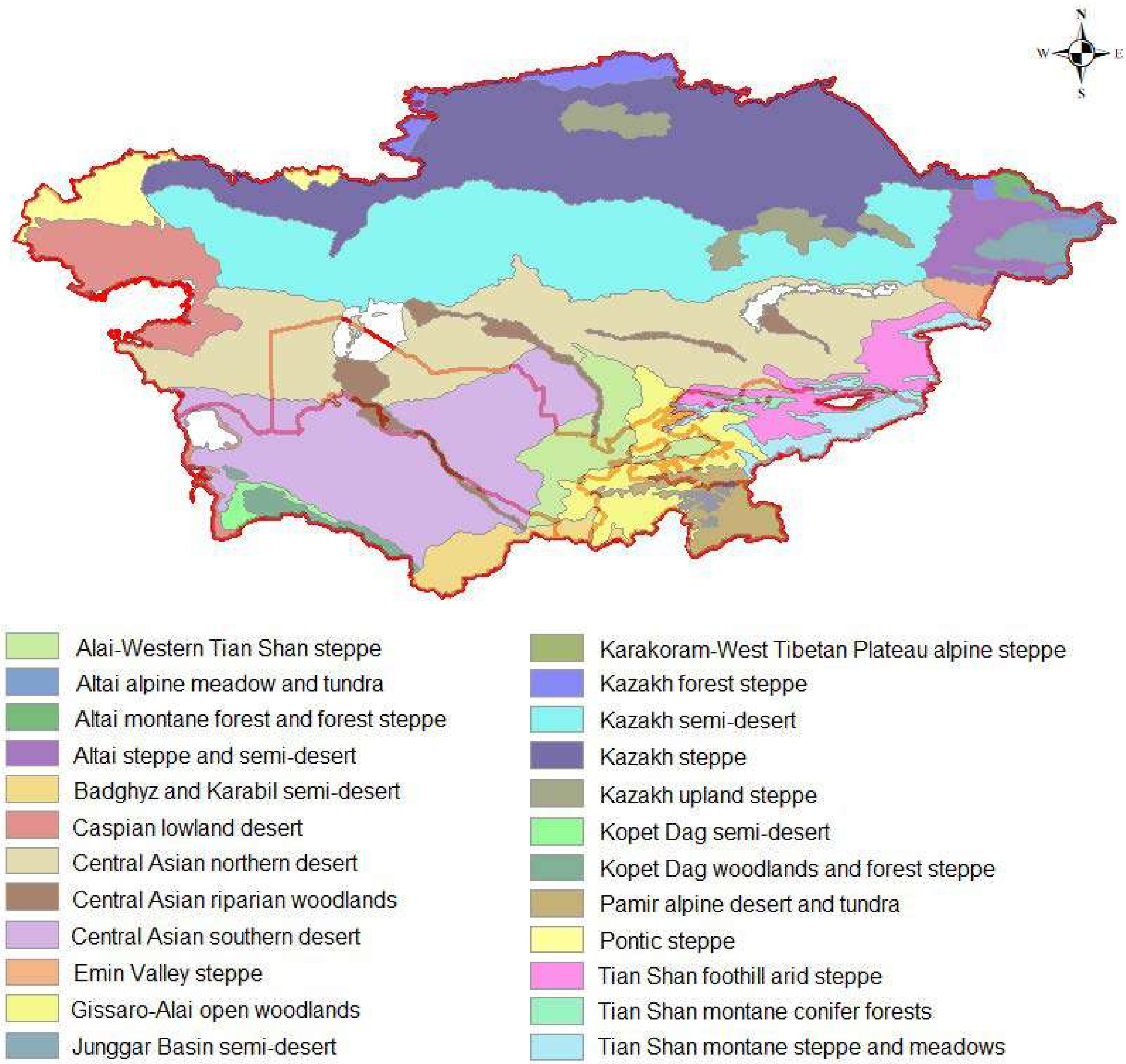

Fig. 1. The terrestrial ecoregions from the 'Global 200' list (Olson & Dinerstein 1998) represented in the CA countries. Red lines denote the country borders.

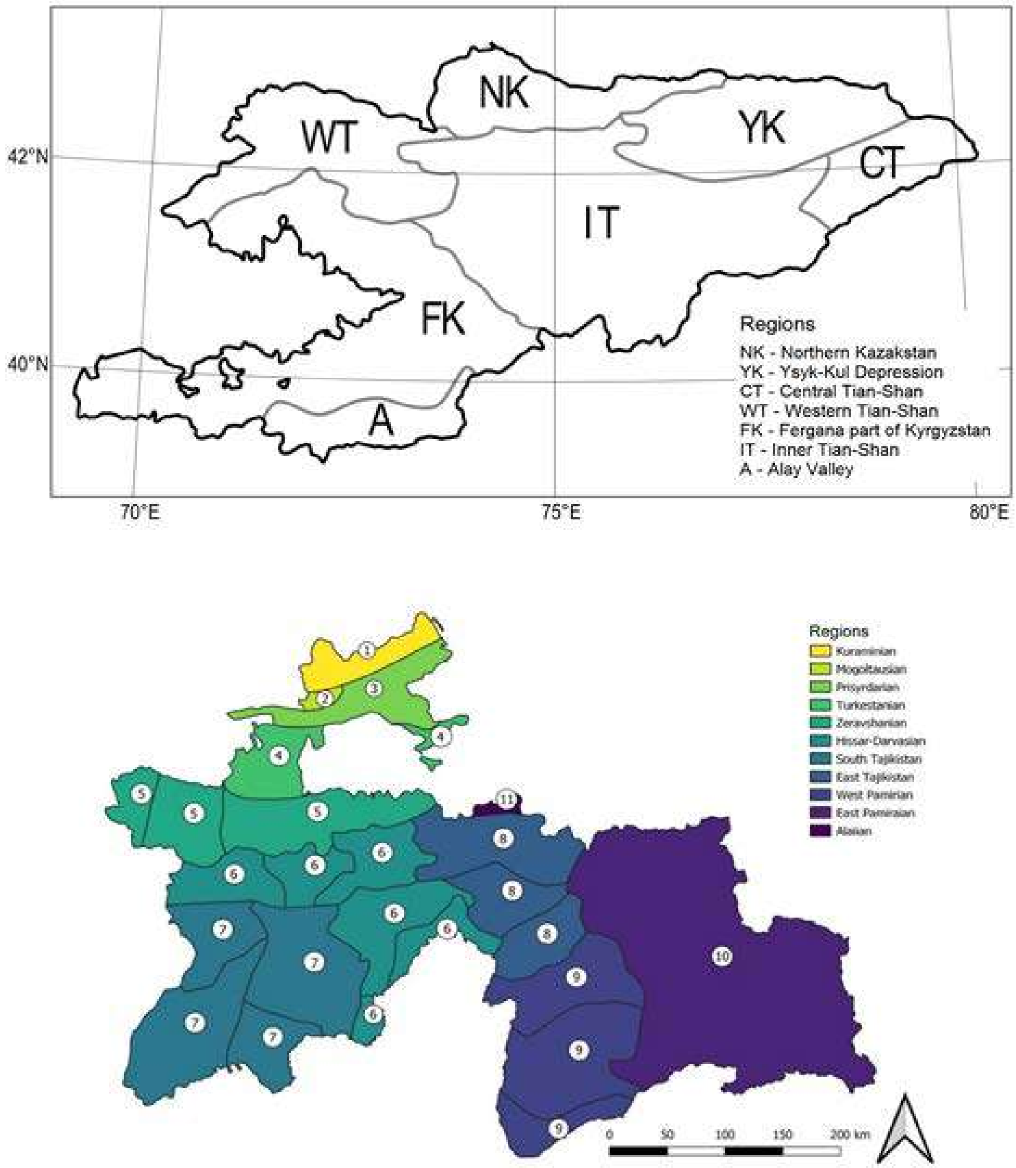

Fig. 2. Top. The biogeographical division of Kyrgyzstan (from Lazkov & Sennikov 2015). Bottom. Phytogeographical division of Tajikistan (from Nowak et al. 2020)

Species and population inventory

Spatially explicit data on biodiversity is fundamental to any conservation planning, and therefore conservation within an ecoregion always starts from a species and population inventory. The inventory, compiled from field observations and regional occurrence of species recorded in databases, must include all threatened species found in an area along with the approximate size of each population, or at least with the number of known locations (Fig. 3). The GPS coordinates of every known species should be deposited in the Global Biodiversity Information Facility (GBIF, http://www.gbif.org/).

Unfortunately, the importance of collecting and disseminating this information has been recognized as an important task only by a small fraction of CA botanists (e.g. Nowak et al. 2020). In the past this may have been due to the lack of necessary equipment (i.e. GPS receivers). Today, due to availability of free software that can be installed on a cell-phone, there is no need for expensive equipment to obtain precise coordinates. However, the information on species occurrence in CA is still very difficult to obtain. Many local botanists appear to be reluctant to provide their data on species occurrence to GBIF. Such attitudes should change toward making their data more open and publicly available.

	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R	S	T	U	V
1	ID	Species name according to Cherepanov 1995 with basic s	Status in the country	Habitat	...e form	Number of locations before 2000	Current number of locations (after 2000)	All known locations	Criterion IUCN 2017	Threat category IUCN 2017	Threat category according to Hissorev 2015	Threat category for Tajikistan	Global threat category	Number of occupied subregions	EOO [km²]	AOO [km²]/trend	...ed as medicine	...ed as forage	...ed as food	...ed in industry	...ed in household	...ed as ornamental plants
2	2054	Saponaria tadzhikistanica (Botsch.) V.A. Shultz	E	steppes, loose sandy screes	A	3	1	4	B2ab(ii)		VU	EN	EN	2	12370	<500						
3	1	Abutilon theophrasti Medik.		segetal, gardens and orchards, ruderal veg	A	17	15	32				LC	LC	5	28348					1		
4	2	Acalypha australis L.	A	ruderal vegetation	A	0	2	2				NE	NE	1	4054							
5	3	Acanthocephalus amplexifolius Kar. et Kir.		forbs, meadows, steppes, segetal	A	14	12	26				LC	LC	8	36154							
6	4	Acanthocephalus benthamianus Regel		forbs, meadows, steppes, segetal	A	77	24	101				LC	LC	20	1E+05							
7	5	Acantholepis orientalis Less.		semi-deserts, river beds, salt shrubs	A	4	0	4	B2ab(ii)			EN	LC	1	9114	<500						
8	6	Acantholimon afanassievii Lincz.	E	rocks	Ch	7	0	7	B1ab(1)			VU	VU	2	12370							
9	7	Acantholimon alberti Regel		steppes	Ch	1	1	2	B1ab(i)			EN	LC	1	3146							
10	8	Acantholimon alexeenkoanum Czerniak. ex Ikonn.	E	rocks, screes, river beds	Ch	10	1	11	According to IUCN 2017			NT	NT	4	43978							
11	9	Acantholimon anzobicum Lincz.	E	forbs, rocks, screes	Ch	16	2	18	According to IUCN 2017			NT	NT	3	11800							
12	10	Acantholimon auganum Bunge		steppes, screes	Ch	2	0	2	B1ab(i)+B2ab(ii)			EN	LC	1	2434	<500						
13	11	Acantholimon compactum Korov.	E	steppes	Ch	1	0	1	B1ab(i)+B2ab(ii,iv)			CR	CR	1	650	<10						
14	12	Acantholimon diapensioides Boiss.		alpine steppes, moraines and snow-beds,	Ch	48	25	73				LC	LC	3	41281							
15	13	Acantholimon erythraeum Bunge		xeric shrubs, forbs, steppes	Ch	28	0	28				LC	LC	13	50213	decline						
16	14	Acantholimon hedinii Ostenf.	SE	steppes	Ch	11	2	13	According to IUCN 2017			NT	LC	1	32568							
17	15	Acantholimon hilariae Ikonn.		alpine steppes, alpine meadows	Ch	21	8	29				LC	LC	4	45843							
18	16	Acantholimon kokandense Bunge	SE	forbs, steppes	Ch	7	1	8	B1ab(1)			VU	LC	3	13291							
19	17	Acantholimon komarovii Czerniak. ex Lincz.	E	rocks, screes, steppes	Ch	10	1	11	According to IUCN 2017		EN	NT	NT	2	9346							
20	18	Acantholimon korolkowii (Regel) Korov. ex Lincz.	SE	steppes, alpine steppes, juniper forests	Ch	12	1	13	According to IUCN 2017			NT	LC	6	25150							
21	19	Acantholimon kuramense Lincz.		steppes, screes	Ch	1	0	1	B1ab(i,iii,iv)+B2ab(ii,iv)			CR	LC	1	3146	<10						
22	20	Acantholimon lycopodioides (Girard) Boiss.		forbs, rocks, screes	Ch	17	1	18	According to IUCN 2017			NT	LC	4	15972							
23	21	Acantholimon pamiricum Czerniak. ex Lincz.		alpine steppes	Ch	35	2	37				LC	LC	6	51528	decline						
24	22	Acantholimon parviflorum Regel	E	steppes, forbs	Ch	28	2	30				LC	LC	8	24822							
25	23	Acantholimon raicoviae Czerniak. ex Lincz.	E	rocks	Ch	2	0	2	B2ab(ii)			EN	EN	2	7259	<500						
26	24	Acantholimon tataricum Boiss.	E	rocks, screes	Ch	29	10	39	According to IUCN 2017			NT	NT	5	17991							
27	25	Acantholimon tianschanicum Czerniak.		alpine steppes	Ch	2	5	7	B2ab(ii)			VU	LC	1	32568	<2000						
28	26	Acantholimon varivtzevae Czerniak. ex Lincz.	E	alpine steppes	Ch	6	0	6	B2ab(ii)		VU	VU	VU	3	40295	<2000						
29	27	Acantholimon velutinum Czerniak. ex Lincz.	E	steppes, juniper forests	Ch	22	4	26				LC	LC	5	51339							
30	28	Acantholimon virens Czerniak. ex Lincz.		steppes	Ch	12	0	12	According to IUCN 2017			NT	LC	2	9049							
31	29	Acantholimon zaprjagaevii Lincz.		rocks, screes	Ch	2	0	2	B2ab(ii)		VU	EN	LC	2	8191	<500						
32	31	Acanthophyllum adenophorum Freyn		thermophilous shrubs	Ch	17	3	20	According to IUCN 2017			NT	LC	2	11803							
33	30	Acanthophyllum albidum Schischk.	E	river beds, semi-deserts, steppes	Ch	14	0	14	According to IUCN 2017			NT	NT	4	9340							
34	32	Acanthophyllum brevicalycale Sosk.	E	screes, thermophilous shrubs, steppes	Ch	6	1	7	B1ab(1)			VU	VU	2	4835							
35	33	Acanthophyllum elatius Bunge		deserts, semi-deserts	Ch	5	0	5	B2ab(ii)			EN	LC	1	8316	<500						
36	34	Acanthophyllum glandulosum Bunge		screes, loose sandy screes	Ch	16	5	21	According to IUCN 2017			NT	LC	2	8713							
37	35	Acanthophyllum pulchrum Schischk.	E	steppes, loose sandy screes, screes	P	10	8	18	B1ab(1)			VU	VU	1	870	<500						
38	36	Acanthophyllum pungens (Bunge) Boiss.		screes, loose sandy screes	P	21	3	24				LC	LC	6	23130		1					
39	37	Acanthophyllum sarawschanicum Golenk.	E	steppes, screes	Ch	12	4	16	According to IUCN 2017			NT	NT	3	11138							
40	38	Acanthophyllum subglabrum Schischk.		steppes	Ch	24	2	26	According to IUCN 2017			NT	LC	2	11462							
41	39	Acer negundo L.	A	river beds	M	50	2	52				NE	NE	2	12834							
42	40	Acer pubescens Franch. [syn. A. ovchinnikovii V. Zapr., A. regelii Pax]		thermophilous shrubs	M	98	35	133				LC	LC	13	51282							1
43	41	Acer semenovii Regel & Herd.		juniper forests	M	13	2	15	According to IUCN 2017			NT	LC	2	3630							
44	42	Acer turkestanicum Pax		broad-leaved forests, xeric shrubs, forbs	M	144	55	199		DD		LC	LC	14	53759							1
45	43	Achillea biebersteinii Afan.		steppes, forbs, thermophilous shrubs	P	64	78	142				LC	LC	19	75848		1					
46	44	Achillea bucharica C. Winkl.		forbs, alpine meadows, moraines and snov	P	26	2	28				LC	LC	10	36348							
47	45	Achillea filipendulina Lam.		steppes, forbs, ruderal vegetation, screes, x	P	79	28	107				LC	LC	18	1E+05		1					
48	46	Achillea millefolium L.		forbs, steppes, meadows	B, P	56	24	80				LC	LC	16	62650		1	1	1			
49	47	Achillea nobilis L.		segetal	P	2	0	2	B2ab(ii)			EN	LC	2	5846	<500						
50	48	Achillea setacea Waldst. & Kit.		xeric shrubs, forbs, screes, loose sandy sc	P	6	2	8	B1ab(1)			VU	LC	3	12195							
51	49	Achillea wilhelmsii C. Koch		steppes, thermophilous shrubs, segetal	P	6	0	6	B2ab(ii)			VU	LC	4	20570	<2000	1					
52	2380	Achnatherum caragana (Trin.) Nevski [syn. Lasiagrostis caragana Trin. e		steppes, forbs, thermophilous shrubs, xeric	P	41	18	59				LC	LC	14	53373							
53	2382	Neotrinia splendens (Trin.) M. Nobis, P. Gudkowa, A. Nowak [syn. Achnath		steppes, loose sandy screes	P	33	11	44				LC	LC	7	57712			1				
54	2381	Achnatherum turcomanicum (Roshev.) Tzvel. [syn. Lasiagrostis longeari		xeric shrubs, steppes	P	34	0	34				LC	LC	9	42573	decline						
55	2872	Achoriphragma albidum (M. Pop.) Sojak [syn. Parrya albida M. Pop.	E	screes	Ch	2	0	2	B1ab(i)			EN	EN	1	3146							
56	2874	Achoriphragma darvasicum (Botsch. & Vved.) Sojak [syn. Parrya d	E	screes	Ch	7	6	13	According to IUCN 2017			NT	NT	2	10787							
57	2876	Achoriphragma fruticulosum (Regel & Schmalh.) Sojak [syn. Parrya	E	juniper forests, xeric shrubs, alpine steppes	Ch	8	3	11	According to IUCN 2017			NT	NT	3	11402							
58	2877	Achoriphragma kuramense (Botsch.) Sojak [syn. Parrya kuramens	E	screes	P	2	1	3	B1ab(i)			EN	EN	1	3146							
59	2879	Achoriphragma pinnatifidum (Kar. & Kir.) Sojak [syn. Parrya pinnatifida K		rocks	P	29	15	44				LC	LC	6	24335							
60	2880	Achoriphragma runcinatum (Regel & Schmalh.) Sojak [syn. Parrya	E	forbs, screes, alpine steppes, moraines an	Ch	49	1	50				LC	LC	11	69821							
61	2881	Achoriphragma schugnanum (Lipsch.) Sojak [syn. Parrya schugnar	E	rocks, screes, alpine steppes	P	30	2	32				LC	LC	9	73387							
62	2883	Achoriphragma turkestanicum (Korsh.) Sojak [syn. Parrya turkesta	E	screes, rocks, moraines and snow-beds, ic	P	61	3	64				LC	LC	12	75837							

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Fig. 3. An example of species occurrence database (from Nowak et al. 2020)

Demographic survey

Within each delimited region, every population of a threatened species must be visited for at least preliminary survey of the population demographic structure focusing on regeneration, i.e. production of seeds and presence of seedlings and juveniles. Even a single snapshot of the population demographic structure often can allow identification of the cause of a population's inviability (which may be due not only to direct anthropogenic impact, but, for example, unavailability of mates or pollinators for successful reproduction, seed predation or lack of microhabitats for seedling establishment) (Volis & Deng 2019). In small populations, the number of reproducing adults should be estimated. Currently, there is no such information even for the most critically endangered species of CA. The preliminary demographic survey ideally should be followed by properly organized long-term observations on population demography and reproductive phenology and study of the biology of each species. Knowledge of the population demographic structure, mode of pollination and the main pollinators, seed dispersal, breeding structure, presence of seed dormancy, age at maturity and pattern of fruit/seed production are important for working out the management plan. Knowledge of the reproductive phenology is also necessary for developing a seed collection calendar. Repeated observations and tracing the fate of marked plants over several years makes possible population viability analysis (PVA) (i.e. predictions about population size and structure over time). If, for example, the plants used in Akhmedov et al. (2021) for analysis of demographic structure (Fig. 4) were not only counted but marked with numbered tags and repeatedly visited, it would make PVA possible.

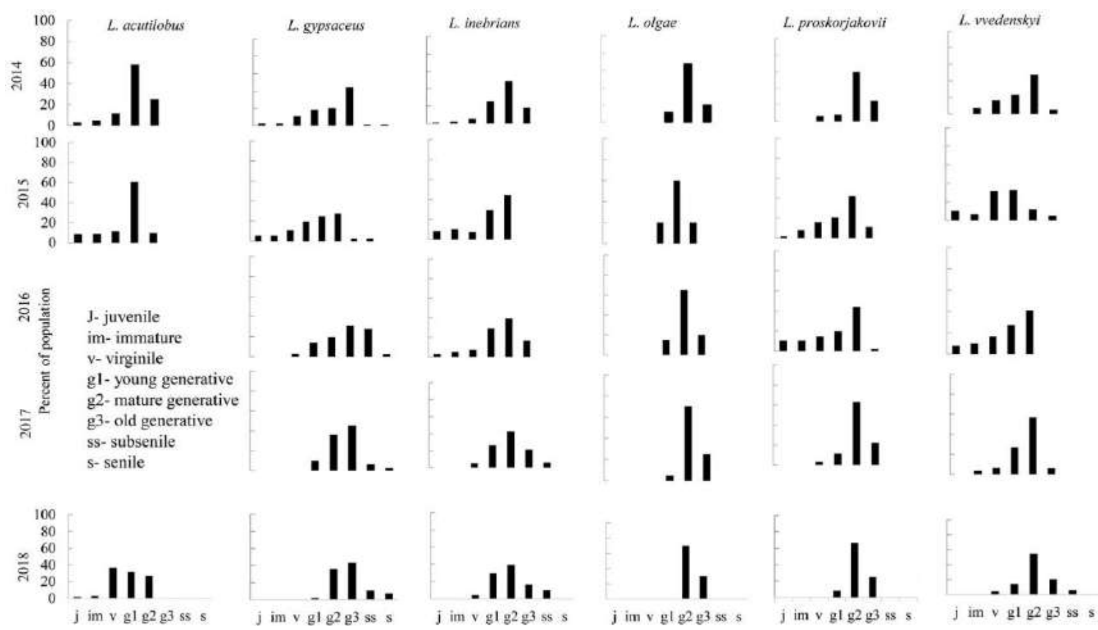

Fig. 4. Population demographic structure of six species of *Lagochilus* in Uzbekistan in 2014-2018 (from Akhmedov et al. 2021)

The focus of the demographic survey must be on species regeneration because many extant populations of threatened plant species, even in the PAs, do not regenerate naturally. A species regeneration niche can be destroyed, if, for example, altered soil or light conditions make germination impossible, or when extirpation of pollinators/seed dispersers make seed production/dispersal impossible. Although lack of recruitment is a common problem in threatened species, it has so far received limited attention in general, and in CA research in particular. Even if problems with natural regeneration are reported, recommendations usually do not go beyond trivial and obscure sentences like “to promote conservation of natural populations by removal of societal threats, e.g. through outreach activities, and by providing alternatives through cultivation of the most useful species for human use” (Akhmedov et al. 2021). What are these ‘societal threats’? How do they impact natural populations? And how will 'outreach activities' help? Who will provide 'alternatives' and by whom (and where) will 'the most useful' plant species be cultivated? The authors do not explain.

When problems with recruitment are detected, observations on flowering and fruiting, and introduction experiments to manipulate germination and growth conditions can reveal the causes of regeneration failure in populations. Depending on the cause, the possible solutions can be thinning of neighboring species, introduction of nurse plants or planting more individuals of the target species to increase the pollinator visitation rate. If there is no remedy for the observed regeneration failure, the ultimate solution may be translocation to a location where the species' requirements for regeneration are met.

Although analyses of population demography and regeneration niches are still scarce for CA threatened plant species, there has been certain progress in this direction. Orozumbekov et al. (2015) conducted demographic surveys and assessed the status of 10 threatened species of fruit and nut tree of Kyrgyzstan (Fig. 5) where field surveys were carried out across the entire range of walnut forests. Wilson et al. (2019) analyzed the population structure of *Malus niedzwetzkyana* in Kyrgyzstan, focusing on the capacity for species regeneration. Unfortunately, in both of those studies the most important stages for regeneration (seedlings and saplings) were skipped by the authors. In Uzbekistan, as already mentioned, Akhmedov et al. (2021) analyzed the population demography of six species of *Lagochilus*, four of which were red-listed ones. In Tajikistan, a survey of forest trees, including several threatened species, was conducted in Dashtijum Nature Reserve to study forest composition, tree population demography and assess the use of forest resources by local villagers (Pilkington et al. 2020). Those studies, although not without drawbacks, provided information important for developing conservation plans for those species and highlighted the utility of population demographic data for the latter. A young generation of CA conservation biologists should be encouraged to learn and implement population ecology analytical methods to tackle population regeneration problems.

Ranking threatened species by conservation value

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), or in some CA countries analogous categories inherited from the Soviet era. However, this categorization is simplistic and often misleading when applied to setting conservation priorities. Species having the same category can dramatically differ in many important attributes. For example, one of two species having the same category can be a regional endemic and the second one to be locally rare but common elsewhere. For example, *Platanus orientalis*, *Punica granatum*, *Ficus carica*, *Vitis vinifera*, *Diospyros lotus* are all listed in the Red Book of Uzbekistan (Khassanov 2019) but very common in many other places. Or one species can represent a monotypic genus and the second one a genus with several hundred species closely resembling each other. For example, there are 11 *Eremurus*, 19 *Tulipa*, 20 *Cousinia* and 34 *Astragalus* species in the Red Book of Uzbekistan (Khassanov 2019). Have any of them the same importance for conservation as the single representative of the genus *Ostrowskia*, *O. magnifica*, which is also Red Listed? In other words, regional conservation planning must have a more nuanced, while straightforward and easy to use, procedure to rank species. A solution was proposed by Freitag and colleagues (Freitag & Van Jaarsveld 1997; Freitag et al. 1997). They proposed to complement the IUCN categories with three additional criteria: relative endemicity (the proportion of a species' range in a region) (RE), regional occupancy (RO) and taxonomic distinctiveness (RTD), i.e. to combine several quantitative estimates of species rarity, uniqueness and vulnerability. These estimates can be calculated in different ways depending on spatial scale and quality of the species' occurrence data using either special units, area occupied or number of populations. A description of how to calculate these and two additional criteria can be found elsewhere (Volis 2017, 2019) and are not covered here

Regional systematic planning that utilizes a unified scoring system that combines several measures of species rarity, uniqueness and vulnerability will give the highest priority to the most endangered local endemics with regeneration problems. This should help in reserve design, monitoring and assessment of efficiency of a reserve network, and creation of a seed bank and living collections, as will be shown below.

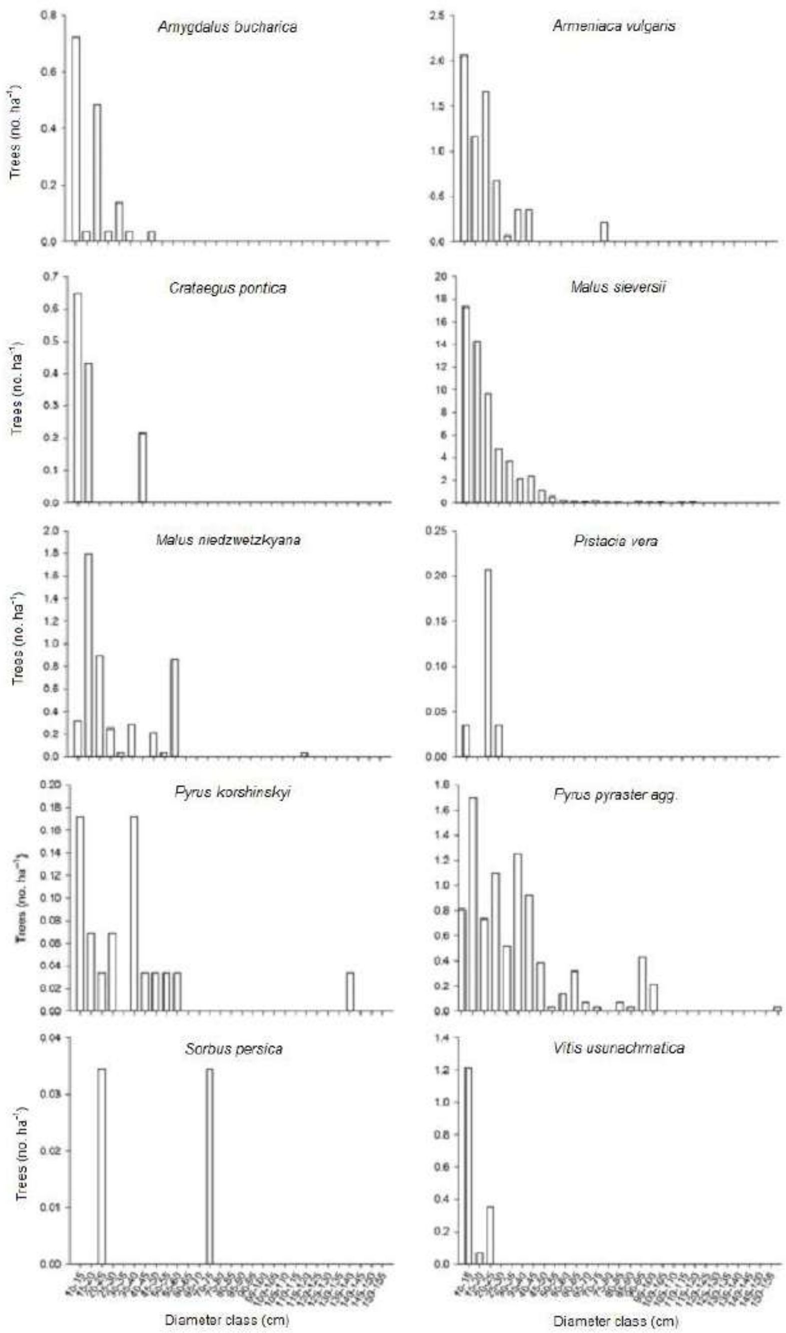

Fig. 5. Mean density and size structure for 10 tree species in walnut forests of Kyrgyzstan. Size classes start from 10 cm dbh and differ from each other by 5 cm (from Orozumbekov et al. 2015).

Ranking areas for protection

Legislated land protection is the core component of nature conservation. The major purpose of systematic conservation planning is to create a network of conservation areas covering those most important for preserving biodiverse natural territories (Margules & Pressey 2000). Designing such network requires candidate areas to be first assessed for their conservation value and then prioritized for inclusion in a network. This procedure called reserve selection is actually an optimization algorithm that searches for a compromise between resource allocation (land area) and conservation efficiency. In other words, the goal of this procedure is, while minimizing land area and other costs, to create a network of areas that will protect all conservation targets (species or habitats) and include as many elements of biodiversity as possible. Computer programs to select reserves divide the region into a set of spatial units (‘planning units’) either based on existing boundaries (e.g. administrative or ecological) or grid cells of a preset resolution, and assign each planning unit a conservation value estimated in one way or another depending on the focus, which can be on an individual species, habitat or ecosystem. The parameters for evaluation can be species occupancy (presence or absence), species abundance, habitat conditions etc. Since a desired solution is a set of sites whose assemblages of species complement each other and together embrace the largest species pool, selecting sites by species richness is inappropriate. Instead, selection algorithms use the concept of complementarity (Pressey et al. 1993), i.e. a measure of the degree to which a site contributes to the representation of biodiversity features that are not adequately represented in the existing set, and the concepts of irreplaceability and vulnerability. Sites with lower irreplaceability can be replaced by other, unselected sites, while sites with high irreplaceability cannot be replaced and therefore negotiated (Pressey et al. 1994). Sites with lower vulnerability have a lower probability of biodiversity loss due to current or imminent threatening processes (e.g. infrastructure development) (Pressey et al. 1996).

Current reserve selection algorithms produce either a set of sites that represent the maximum number of species in a given number of sites, or all species in the minimum number of sites (known as maximum coverage and minimum set problems), referred to as a ‘solution.’ A solution in the second option (implemented in the program, Zonation; Moilanen et al. 2009) is an area that includes the maximum possible proportion of distribution for each species. To rank sites over a landscape based on habitat quality for each species and land cost, Zonation considers the landscape as a grid of cells, and iteratively ranks cells, at each step removing cells whose loss results in the smallest marginal loss in the overall conservation value of the remaining landscape. The later in the process that cells are removed represents the highest priority they have. The least valuable cells receive the lowest ranks (close to 0), and those having the highest priority get the highest ranks (close to 1). The resulting solution is hierarchical, meaning that the top 2% is within the top 5%, which is within the top 10% and so on.

Zonation has two options for cell removal in finding an optimal solution: the additive benefit function (ABF) and the Core Area Zonation (CAZ). The CAZ emphasizes, in contrast to ABF, not species richness but rarity, and therefore tends to rank cells containing habitat for rarer species as highly irreplaceable, because habitat for those species is available in few or no other sites in the landscape. As a result, CAZ is the most appropriate algorithm to ensure high-quality locations for all species, regardless of whether these locations are in species-rich or species-poor areas (Moilanen et al. 2005, 2011; Lehtomäki & Moilanen 2013).

Zonation has an advantage over other programs that design reserves in that it allows weighting of species by their conservation values. Such weighting allows changing a focus in reserve network design from maximizing overall species protection to maximizing protection of the most threatened species. Species can be prioritized either by IUCN Red List Categories (Fiorella et al. 2010) or by a combination of criteria (Freitag et al. 1997; Arponen et al. 2005; Lannuzel et al. 2021). The regional priority scores described above can be particularly useful for this purpose.

Although it has been more than a decade since the use of decision support software tools in systematic conservation planning became a standard, in CA the prioritization and design of protected area networks is still based exclusively on expert opinions. Although expert knowledge is necessary for selecting regionally important habitats and locations vital for charismatic animal species (e.g. breeding places, migration routes), it is known that umbrella and flagship species are usually poor surrogates for regional biodiversity (Andelman & Fagan 2000), and habitat requirements of majority of rare species are not known even to the local experts. For those reasons the reserve selection algorithms based on occurrence records are much more efficient in working out the design of reserve networks. Volis and Tojibaev (2021), in their study of 50 critically endangered plant species of Uzbekistan, found that about half of the species not only did not have a record of occurrence within the existing PAs, but even their predicted suitable habitat (which covered a substantially larger territory than their known occurrence) was outside any PA. Using Zonation reserve design software, Volis and Tojibaev (2021) identified areas that harbored critical habitat of a disproportionally high number of critically endangered plant species. Those areas were covered very poorly by the existing PAs (Fig. 6). It is highly probable that the situation is the same in the other CA countries.

Although establishing new PAs is bureaucratically complicated and time-consuming, every possible effort for achieving this goal should be tried. Any proposal for a new PA must be justified using modern scientific tools, i.e. reserve selection software. Ideally, data for reserve design should be based on species presence/absence data. When a species is rare, but its distribution is thoroughly studied, it is possible to assign to every cell of a landscape a binary value 0 (absent) or 1 (present). However, for majority of CA endangered species this is not possible because the occurrence records are incomplete and often the only source of this information is herbarium specimens. A solution is to use as input data in reserve selection algorithms, instead of occurrence records, maps of SDM-predicted habitat suitability. The number of studies doing so is growing (e.g. Kremen et al. 2008; Cuesta et al. 2017; Taylor et al. 2017; Choe et al. 2018; Spiers et al. 2018; Manish & Pandit 2019), and convincingly showing that the integration of SDM and site-selection algorithms is a powerful tool for reserve design.

Ex situ: collecting and storing seeds.

Ex situ conservation (e.g. preserving a species outside its natural settings) until recently has been considered a cost-effective alternative to conservation *in situ*. This view has changed. Nowadays *ex situ* collections are seen as complementary rather than as an alternative to *in situ* conservation. Germplasm banks can contribute to plant recovery in several ways. First of all, they can buy time for threatened species, making them immune to habitat destruction, grazing and predation during storage. Second, they can provide vital knowledge about seed longevity, breaking dormancy, and germination requirements. And third, they can supply plant material (seeds and seedlings) to *in situ* recovery programs. One of the targets of the Global Strategy for Plant Conservation 2011–2020 is to make 20% of the germplasm bank collections available for in situ conservation actions (http://www.plants2020.net/gspc-targets/).

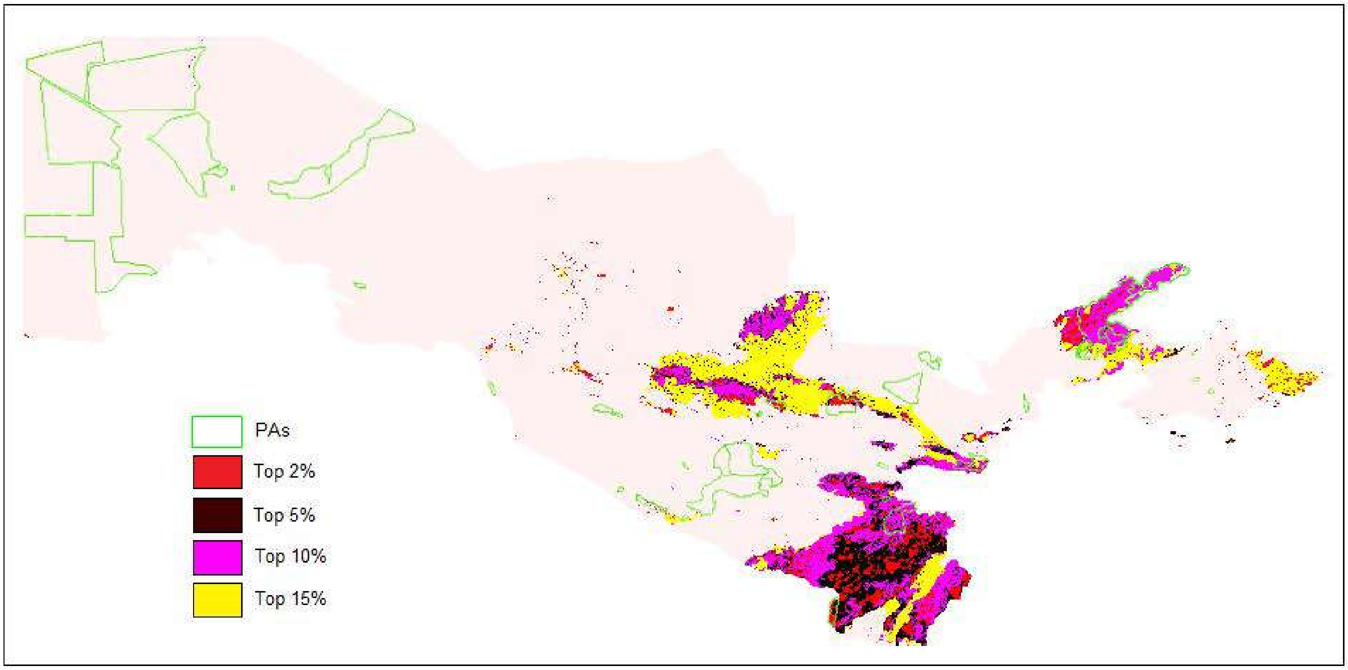

Fig. 6. Spatial distribution of conservation priorities produced by Zonation using the predicted critical habitats for 50 critically endangered plant species, and existing protected areas (PAs) in Uzbekistan (from Volis & Tojibaev 2021). Nested hierarchical ranking of conservation priorities is shown for four levels: top 2%, 5%, 10% and 15%.

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About 20-25% of known seed-bearing species cannot be subjected to conventional seed banking because their seeds are recalcitrant, i.e. cannot survive a drying process at 15% eRH, 15 °C) and prolonged storage at −20°C (Walters et al. 2013). In contrast, the non-recalcitrant (orthodox) seeds can remain viable under these conditions for decades, perhaps centuries (Walters et al. 2004). Fortunately, in CA the vast majority of higher plants produce orthodox seeds and for them seed banking is possible. Despite such an advantage, at present, there is only one seed bank with threatened plant species in its collection in CA. The Institute of Botany and Phytointroduction (Kazakhstan) maintains a collection of wild plant species in modern seed storage facilities (Fig. 7). Ideally, threatened species will constitute a core of this collection and analogous collections will be created in the other CA countries. There are no serious obstacles, either financial or logistical for establishing seed banks in botanical gardens or botanical institutes because the majority of non-recalcitrant seeds can be maintained for long periods of time under −20 ℃, a temperature that can be provided by a wide range of inexpensive freezers. Even small collections preserving tens of threatened species in freezers of conventional refrigerators can be far more important for plant conservation than huge seed storage facilities preserving thousands of common species. Obviously, those seeds must be available for *in situ* actions, or at least for the creation of living collections.

Fig. 7. Seed bank collection at the Institute of Botany and Phytointroduction (Kazakhstan)

As a part of systematic conservation planning, in all populations targeted for demographic surveys, seed collecting must be initiated. Sampling should be done repeatedly over several years to prevent harm to a population, or simply because seed production is too low. During collection, seeds from different individuals should be kept separately as maternal lines (families) and not mixed into a single bulk collection. More detailed guidelines for collecting seeds of threatened species can be found in Volis (2016d,c).

The right time for seed collecting is critical as there can be only a short window of a few weeks or even days between seed maturation and dispersal after which they are no longer available for collection. Therefore it is important to have for each target species a collection calendar with information about onset and duration of fruiting (Fig. 8).

Locating reproducing individuals of threatened species is a difficult task because many of them have small, sparsely distributed populations in remote places.

Species	J	F	M	A	M	J	J	A	S	O	N	D	Notes
	Bu = Buds; Fl = Flowering; Im = Immature Fruits; M = Mature Fruits												
Species 1		Bu	Fl	Fl	Fl	Im	M						Fruits turn black when ripe
Species 2	Bu	Fl	Fl			Im	Im	M					Rainy season triggers fruiting
Species 3	Bu/Fl	Fl	Im	M			Bu	Fl	Im	M			Fruits turn from green to red when ripe

Fig. 8. Seed collection calendar (from Hoffmann & Velazco 2014)

The common challenges of seed collecting of rare species are illustrated in Hoffmann et al. (2015). In a restoration project of the *Araucaria* Forest of southern Brazil, after compiling a list of threatened and rare species for collecting, and mapping all the known occurrence records, the authors produced a list of potential forest remnants containing at least one target species. Each of these remnants was surveyed intensively to identify healthy and reproducing mother trees. All attempts were made to have mother trees from at least three populations of each species and from at least 12 trees per population, or, when this was not possible, at least 20 trees per species in the whole study area. The selected trees of the same species were at least 50 m apart unless the trees grew in a single cluster. Information on the onset and duration of flowering and fruiting was verified during 52 field trips throughout the study area during four consecutive years and used to develop a seed collection calendar. Despite the high intensity of the search (68.7 km of trails were surveyed), more than a dozen mother trees in at least three remnants were located for only five species; for 23 of the target species, less than 20 mother trees were selected; and no individual could be found for 38 out of 71 targeted species. These disappointing results of very well planned and well performed collection missions convincingly show the extreme difficulties of collecting seeds of rare species. Therefore it is vital to engage experts in preparation and conducting seed collections, and to do them repeatedly over large areas.

Although ideally all threatened and rare species should be represented in seed bank collections, due to the challenges described above, and because financial resources are always limited, species prioritization is necessary. The species most threatened globally are a priority, but prioritization should also take into account regional and local concerns, for example some imminent threats that would warrant urgent seed collecting for some species. Over the course of time, when more and more accessions are collected, criteria for prioritization can change, and follow, for example, a decision tree for seed banking of Rivière et al. (2018) (Fig. 9).

In seed banks, seeds from each maternal plant should, whenever possible, be maintained separately for a number of reasons. In some species-specific genotypes may be needed to reinforce sexual reproduction. If the intermediate step of targeted breeding is needed, only collections organized by family can provide the information necessary for breeding designs. And for creation of quality reintroduction gene pools, the seed identity is necessary for maximizing family number and equalizing family size (Guerrant 1996; Guerrant & Pavlik 1997; Havens et al. 2004). If seeds are in storage for long time, control of their viability is necessary (Ferrando-Pardo et al. 2016).

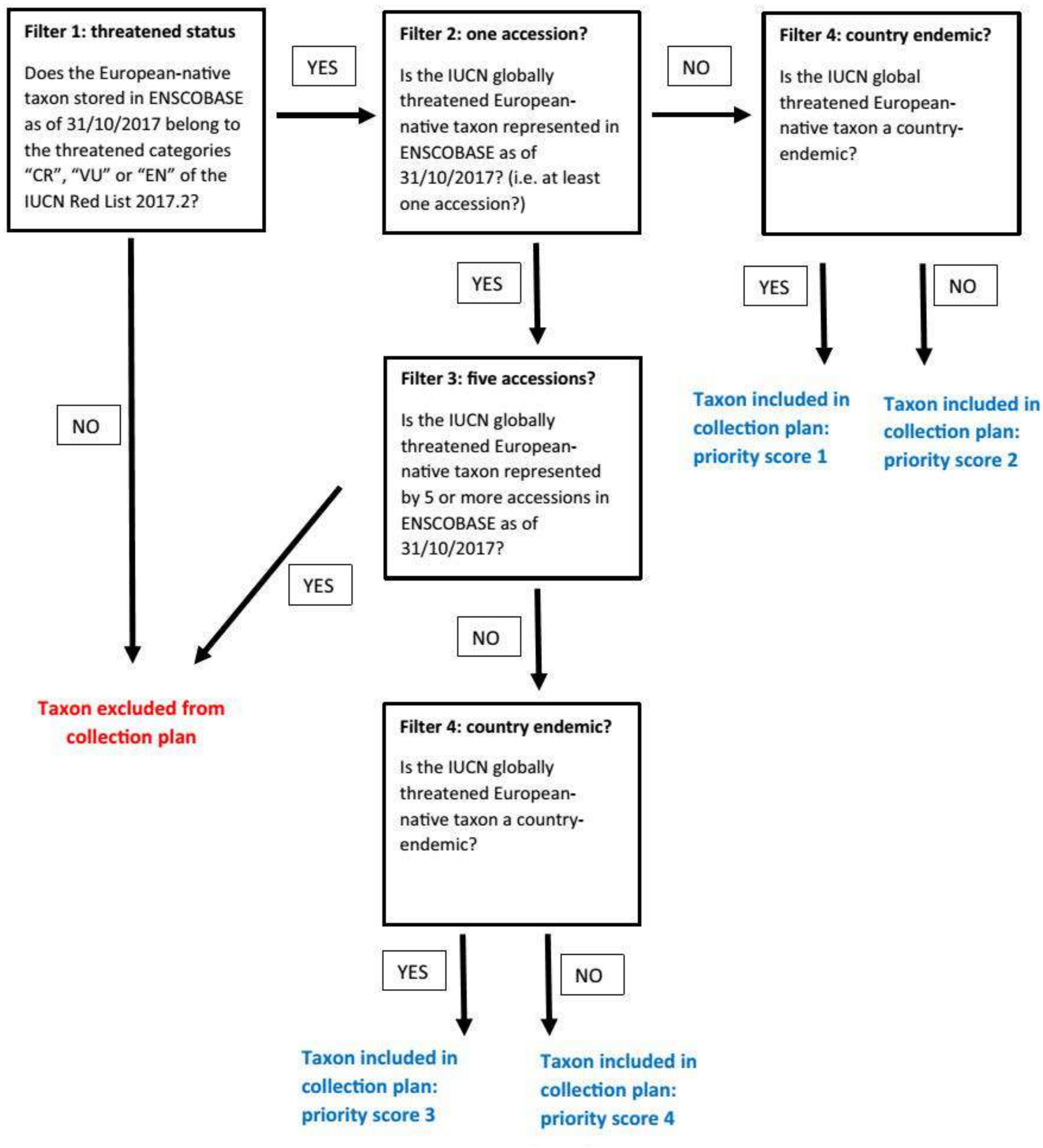

Fig. 9. Decision tree for priority-setting methodology of the collection plan: retrieval of all threatened European-native taxa from the combined checklist of threatened taxa, filtered by 'at least one accession', 'equal to or higher than five accessions' and 'country endemics' (from Rivière et al. 2018)

Ex situ: vegetative and micro-propagation

For many species producing too few or low-quality seeds or seeds damaged by pests and pathogens, vegetative propagation can be a solution. Besides, the use of vegetative propagation often can be a cheaper alternative even for producing healthy seed species. The common form of vegetative propagation is cuttings, which are typically branches or shoots 20–40 cm long from the previous season's growth that are about as thick as a pencil. Mature branches (harvested farther down a stem) usually establish more readily than apical cuttings, and leafy cuttings are more successful than leafless ones (Zahawi & Holl 2014). Most shrubs and trees, including highly threatened ones can be propagated from cuttings (e.g. *Wollemia nobilis*, Trueman & Peters 2006). For CA, beside arboreal species, vegetative propagation of threatened herbaceous plants (which constitute the vast majority of threatened CA species) is a very promising, but unfortunately so far overlooked in this region, possibility. Many plants with a soft, herbaceous growth habit can be propagated using soft tip cuttings. A soft tip cutting should include the first four nodes (two nodes for the shoot and two nodes for the roots) and have about half to two thirds of the leaves trimmed off. A cutting should be dipped into rooting hormone (the same used for rooting softwood cuttings) prior to placing it in a hole made in the rooting medium. The rooting medium generally consist of peat moss combined with vermiculite or perlite. Usual rooting time ranges from ten days to a few weeks. It is important to keep the planted cuttings damp at all times under a mist system or under waterproof sheeting. Some plant families produce roots from leaves. In this method it is important to let the wound at the leaf-base dry before planting the cuttings in coarse sand.

Unfortunately, some threatened species cannot be propagated by either seeds or cuttings. For those species, the only possibility (beside digging out the whole plant) is micropropagation. With the availability of funds, after establishing conventional seed banking, micropropagation facilities are a desirable option. Although micropropagation requires specialized equipment (e.g., transfer hoods) and staff training, the start-up and maintenance costs are affordable. For example, the entire collection of Lyon Arboretum Hawaiian Rare Plant Program (HRPP) at the University of Hawaii, Mānoa representing >200 native plant taxa is housed in a 75 m^2 room. The collection can be maintained by one fulltime staff member with the help of student interns (Werden et al. 2020). Micropropagation approaches can be an indispensable means for i) germinating immature seeds or increasing the germination rate of mature seeds when few seeds are available and/or when previous greenhouse germination attempts have failed; ii) cloning and multiplication of material when seeds are produced infrequently or when time to reproduction takes years; iii) protecting species from pathogens; and iv) maintaining as clones parental lines important for targeted breeding (Werden et al. 2020). A major step in micropropagation is induction and multiplication of shoots. Some successful multiplication strategies developed for a number of endangered Romanian plants, summarized in Table 1, can be useful for developing protocols for herbaceous CA species. However, *in vitro* multiplication makes little sense if is not followed by rooting. In this respect it is worth mentioning the experience of a Latvian research team that successfully germinated seeds (6 gl^{1} agar-solidified half-diluted MS medium with pH 5.8), obtained shoots from the seedlings (0.1 to 0.5 mgl^{-1} BAP, 0.1 to 0.5 mgl^{-1} Kin, 0.1 to 0.5 mgl^{-1} IAA) and achieved rooting on half-diluted MS hormone-free medium with low sucrose level for *Alyssum gmelinii*, *Dianthus arenarius*, *Galium schultesii*, *Galium tinctorium*, *Gypsophila paniculata*, *Juncus balticus*, *Linaria loeselii*, *Scrophularia umbrosa*, *Serratula tinctoria*, *Spergularia salina*, *Trifolium fragiferum*, *Tripolium vulgare*, *Veronica montana* (Dubova et al 2010).

Table 1. Multiplication strategies developed for some Romanian endangered species having threatened congeners in CA (from Paunescu 2009)

Species	Explant type	Multiplication media
Astragalus pseudopurpureus	leaf cuttings	MS; BAP - 1 mgl^{-1}, ANA - 0.1 mgl^{-1}
Astragalus p étersii	stem, leaf and floral cuttings	MS; BAP - 0.5 mgl^{-1}, ANA - 0.3 mgl^{-1}
Artemisia tschernieviana	stem and leaf cuttings	MS; BAP – 1 mgl^{-1}, Kin - 1 mgl^{-1}, ANA - 0.25 mgl^{-1}
Centaurea reichenbachii	nodal cuttings	MS; BAP - 1 mgl^{-1}, ANA - 0.1 mgl^{-1}
Cerastium transilvanicum	nodal cuttings from in vitro germinated plantlets	ditto
Dianthus callizonus	nodal cuttings	ditto
Dianthus simonkaianus	floral buds and nodal segments	ditto
Dianthus spiculifolius	apical buds and nodal segments	ditto
Dianthus tenuifolius	nodal segments	ditto
Ephedra distachya	stem cuttings	MS; Kin - 1 mgl^{-1}, 2,4D - 1 mgl^{-1}
Hieracium pojoritense	leaf cuttings	MS; BAP - 1 mgl^{-1}, ANA 0.1 mgl^{-1}
Leontopodium alpinum	floral buds	MS; BAP - 2 mgl^{-1}, IAA - 2 mgl^{-1}
Marsilea quadrifolia	buds excised from rhizome	½ MS without growing regulators

BAP – 6-benzylaminopurine; ANA – alpha-naphthyl acetic acid; IAA – Indole-3-acetic acid; Kin – Kinetin; MS – basal culture media, Murashige & Skoog, 1962; ½ MS – basal media with half strength minerals; 2,4 D - 2,4-dichlorophenoxyacetic acid.

Given the type of material collected (seeds, cuttings or the whole plants), a decision tree from Fig. 10 can help to decide which *ex situ* method(s) are feasible, with the goal of producing the germplasm for outplanting programs.

When the number of plants acting as parents for seed production is small, and fruit/seed set is low, managed breeding is necessary. Usually managed breeding is done in greenhouse facilities and requires vegetative and/or micropropagation. The following example shows how managed breeding can make possible production of large amount of outplants. The Hawaiian endemic *Argyroxiphium kauense* was represented by two remnant populations, having 381 and 6 individuals, respectively, when the species restoration program started. Those two populations served as source populations for reintroduction efforts. To produce the outplants for reintroduction, the flowering wild plants and those cultivated at the Hawaiian Volcano Rare Plant Facility were pollinated with pollen collected from all other plants flowering at the same time, and thoroughly mixed (Fig. 11). The bulk pollen served to increase the genetic diversity of offspring for each maternal founder. In the 21,000 offspring produced, all 169 maternal founders were represented, with no founder accounting for > 2.5% of the total number of seedlings, thereby avoiding over-representation (Robichaux et al. 2017).

The experience gained in national programs of *ex situ* plant conservation in South Africa (Nichols 2005), Australia (Offord & Meagher 2009) and Hawaii (Werden et al. 2020) must be

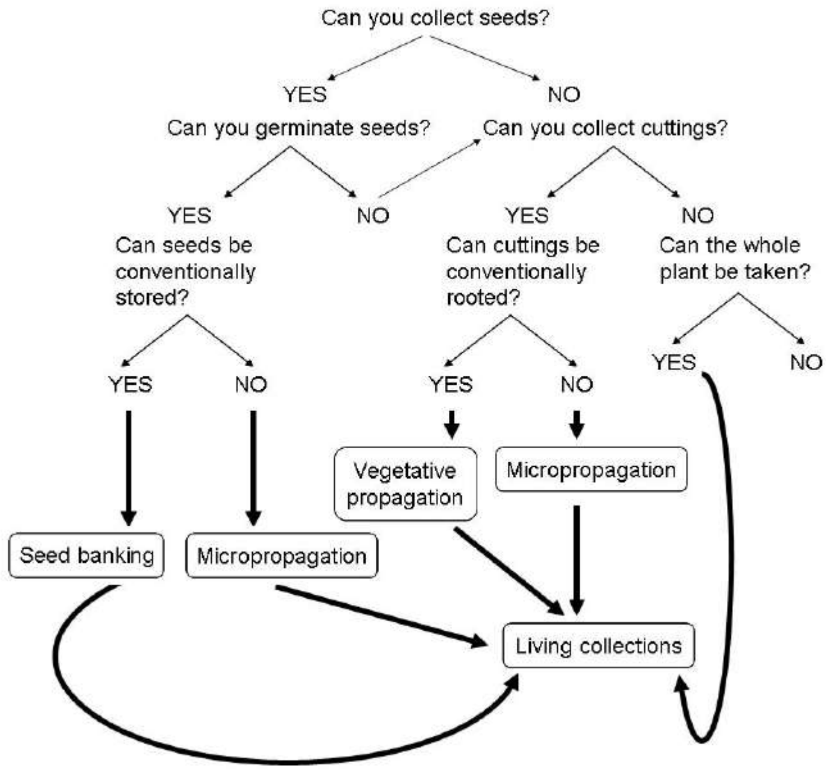

Fig. 10. Decision tree for *ex situ* plant conservation (modified from Werden et al. 2020)

Fig. 11. Managed breeding program for critically endangered Hawaiian endemic *Argyroxiphium kauense*. The photos show the plants cultivated at the Volcano Rare Plant Facility that acted as both mother plants and pollen donors; hand pollination of a mother plant; seedlings produced for outplanting at the Kahuku site in Hawai'i Volcanoes National Park; and the reintroduced plants. Photos by Robert Robichaux

learned and adopted in the CA countries. Availability of facilities for seed banking, vegetative propagation and micropropagation should make it possible to implement ambitious national projects for threatened species conservation.

Ex situ: living collections.

Botanical living collections are another ex situ conservation approach, with these collections being important for the “recovery and rehabilitation of threatened species and for their reintroduction into their natural habitats” (Wyse Jackson 1997). Many botanical gardens around the world maintain threatened species in their living collections, study their biology, develop germination and propagation protocols, and provide seeds and plantlets for various reintroduction and restoration programs (reviewed in Volis 2017). However, botanical gardens have numerous limitations in their utility for maintaining living collections as described elsewhere (Volis 2016c, 2017; Abeli et al. 2020), with the major weakness being their capacity. Traditional guidelines for *ex situ* conservation (e.g. 10–50 individual plants from five natural populations) is impossible to meet for a majority of the botanical gardens.

In CA, a gardens’ collection of threatened species, if it exists, is small, maintained primarily for research or education, and has never provided material for *in situ* conservation programs. Investments in conservation capacity building is certainly warranted, but only as an integral part of a complex conservation approach that includes facilities and qualified personal for seed storage, production of plantlets, and planting and monitoring *in situ*.

Integration of ex situ and in situ

As discussed above, the study of Hoffmann et al. (2015) demonstrated the difficulties in finding a sufficient number of mother plants in a fragmented and degraded habitat. On the other hand, having a sufficient number of mother plants in botanic gardens is impossible due to space and logistical limitations. A solution has been proposed by the *quasi in situ* concept (Volis & Blecher 2010; Volis 2015). The *quasi in situ* concept proposes creation of living collections of needed capacity under natural, semi-natural or artificial conditions. The quasi in situ site maintains a living collection of individuals from populations sharing the same climatic zone and biotic/abiotic environment, has natural or semi-natural conditions and is legally protected. The detailed guidelines for choice of material, planting and management of the quasi in situ living collections are provided in Volis & Blecher 2010; Volis 2016d). These collections resembling natural populations can be useful not only as germplasm depositories. The problems discussed above, i.e. space limitations of botanical gardens, problems with storing non-orthodox seeds, negative impacts of seed harvesting on local population dynamics and scarcity of mother plants *in situ* can be resolved by quasi in situ living collections. The plants maintained in these collections can be a reliable source of seeds to be used *in situ* (Fig. 12).

Fig. 12. Two *quasi in situ* living collections for endangered *Iris atrofusca* in Israel. Note large number of fruits produced by the plant

Beside seeds, living collections can supply cuttings. *Hibiscus liliiflorus*, endemic to the Mascarenes islands, at the beginning of the 1990s was represented by only three individuals. Cuttings taken from these plants were used to create a living collection. When the trees in this collection reached maturity, they, in turn, started to be used as a source of cuttings. The seeds and cuttings supplied by these plants have been and continue to be an important source of material for propagation and *in situ* reintroduction (Jhangeer-Khan et al. 2018).

Living collections can also be important for managed breeding. In the example of the Hawaiian *A. kauense*, breeding was done both in the wild and in a facility. However, when a wild population is not within easy reach and the adult plants are large, living collections suit managed breeding much better. For example, seed orchards helped to produce several hundred genetically diverse seedlings of the critically endangered Sicilian fir, *Abies nebrodensis* (Ducci 2014). This was achieved by replicating the 30 remaining wild adult trees through clonal propagation, grafting and planting the mother trees in a single-tree, random design to enable efficient pollen exchange among the genotypes.

Quasi in situ collections have numerous advantages over ex situ collections, but not ubiquitously better than the latter. The ex situ collections are a better option if the plants can be easily maintained for many years, do not require much space, and produce numerous seeds that can be easily germinated and grown to the required plant size/age. Ex situ cultivation is especially advantageous if seed production requires assistance (e.g., hand pollination), or when quasi in situ collections are not within easy reach. Thus, both ex situ and quasi in situ cultivation should be intensively used for the conservation of local flora, and a decision tree (Fig. 13) should help in choosing the appropriate type of a living collection for a given species.

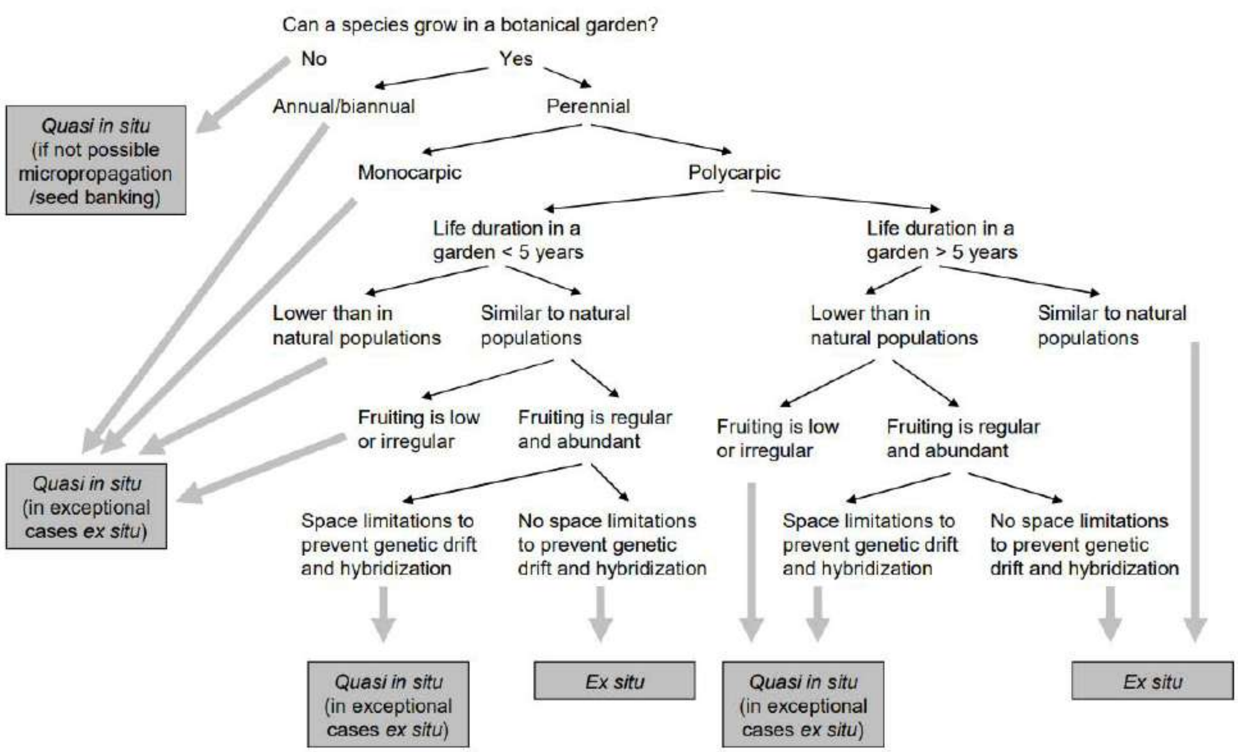

Fig. 13. A decision tree for creation of *ex situ* vs. *quasi in situ* living collections (from Volis 2023)

In reintroduction/restoration projects, seedlings are preferred over seeds because of high mortality and low establishment of the latter (Guerrant & Kaye 2007; Godefroid et al. 2011). Obtaining seedlings from the *quasi in situ* produced seeds can be done either in botanical gardens or in specialized nurseries. Detailed and simple-to-follow guidelines on designing and setting up a nursery as well as planning and managing nursery operations can be found in Stott & Gill (2014), Hoffmann & Velazco (2014) and Wilkinson et al. (2014). To run a nursery efficiently, it is important to keep track of the development of each species summarized on a nursery production calendar (Fig. 14). Containers are used to grow and transport seedlings to the introduction site. Larger containers allow better plant development and thus a higher chance of establishment, but have increased handling and transportation costs. Choice of an appropriate container shape (Fig. 15A) depends on the root system morphology: shallow root systems, characteristic for most forbs, grow better in shorter containers (Fig. 15B), while for long taproots, characteristic of many types of trees, taller containers are preferred (Fig. 15C). Plants with multiple, thick, fleshy roots or rhizomes grow better in wide containers (Fig. 15D). In containers, native topsoil should constitute a substantial part of the soil mix used as the growing medium (Michaelis & Diekmann 2018; Doyle et al. 2021). Seedlings should not be kept in containers too long, i.e. beyond the time when root deformation starts, and 'hardened' before planting to reduce the susceptibility to heat, cold or water stress in the field. Hardening is usually done by reducing watering, starting approximately 2 months before planting, and gradually increasing exposure to the sun. Care must be taken during seedling transportation to the introduction site to avoid damage from excessive transpiration and exposure to wind. A heavy watering of plants is generally recommended before they leave the nursery and a minimum delay between delivery and planting.

Plant Development Record			
Species	Species A	Batch no.	00023
Seed source			
Date of collection	10/02/2015	Number collected	750
Place of Collection	Rare Tree Nature Reserve	Estimate altitude	1000-1200 m
Mother tree ID	SA009	Collected by	James Diaz
Germination			
Date sown	01/05/2015	Number Sown	600
Pre-sowing treatment	Yes - Scarified	Seedbeds or Seed trays	Seed trays
Germination medium	Coir-Sand medium 50:50 ratio		
1st Germination Date	20/05/2015	% Germinated	75%
Nursery Care			
Date seeds transplanted into larger pots	01/06/2015 – 08/06/2015	Number of seeds transplanted to larger pots	450
Potting medium	Sandy soil mixed with compost	Shading	Yes
Fertilizer applied?	Yes	Dates fertilizer applied?	11/07/2015
Pest / disease control	No	Dates applied	NA
Hardening off and Dispatch			
Hardening date started	October 1st	Number survived before hardening off	412
Dispatches	**Date**	**Number**	**Where**
Dispatch 1	02/11/2015	200	Rare Tree Nature Reserve buffer zone
Dispatch 2	03/12/2015	212	Rare Tree Nature Reserve

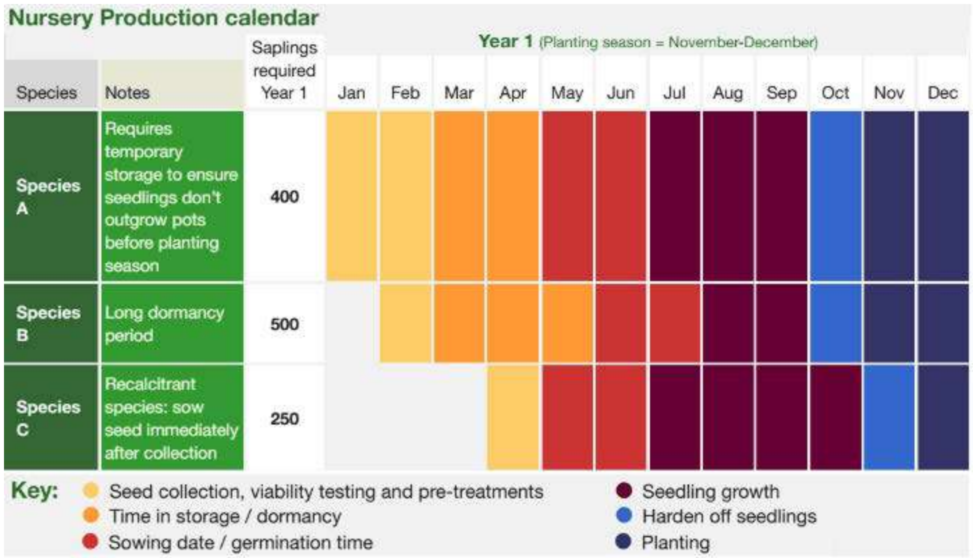

Fig. 14. Plant development records and nursery production calendar (from Stott & Gill 2014)

Fig. 15. Nurseries use a variety of plastic bags (A) and containers (B) to grow plants. Some plants, including most forbs, grow best in shorter containers (C) whereas taprooted species do better in taller ones (D) and fleshy-rooted plants should be grown in short, wide containers (E) (modified from Landis et al. 2014).

In situ: plant introductions

As was proposed in Volis (2022), the sad realities of the regional biodiversity crisis require multi-species introductions of threatened species into existing NRs as the only way to prevent their extinction. Species introductions are increasingly used to reinforce declining populations, recreate extirpated populations and create new populations in order to restore imperiled species or degraded communities (Falk et al. 1996; Maschinski & Haskins 2012; Liu et al. 2015; Silcock et al. 2019; Ren 2020). Introductions are vital for rescuing many imperiled species: analysis of 181 Recovery Plans for endangered plant species (Hoekstra et al. 2002) revealed that 72% of species require some form of plant introductions. For a long time, conservation biologists were critical about introductions outside the historic range of a species (reviewed in Hoegh-Guldberg et al. 2008), but this criticism has been gradually replaced by a more balanced view favoring such introductions in a variety of situations (Millar & Brubaker 2006; Vitt et al. 2010, 2016; Thomas 2011; Volis 2016b; Butterfield et al. 2017). Volis (2016a,b, 2019) provided a set of recommendations for identification of suitable multi-species assemblages prior to introduction, planting design and addressing inter-specific interactions.

The introduction of poorly adapted plants to local conditions should be avoided for two reasons. First, those genotypes will have a poor establishment record. Second, such genotypes can negatively affect adjacent native populations through gene flow by introducing maladapted genes (McKay et al. 2005). This means that plant germplasm must be adapted to the local environment, yet also possess sufficient genetic diversity. A solution is to use ecoregion-based seed transfer zones and quasi in-situ living collections.

In general, higher chances of success result from introductions conducted over multiple years (van Andel 1998; Kirchner et al. 2006; Kaye 2008) and carefully choosing favorable (micro)sites that match the species' niche (Jusaitis 2005; Menges 2008; Knight 2012; Wendelberger & Maschinski 2016). It is always better to use seedlings or saplings rather than sow seeds (Guerrant 1996; Jusaitis et al. 2004; Guerrant & Kaye 2007; Menges 2008; Godefroid et al. 2011; Albrecht & Maschinski 2012), to use growing rather than dormant plants (Batty et al. 2006; Smith et al. 2009), to use larger and more mature individuals (DeMauro 1994; Wendelberger et al. 2008; Smith et al. 2009; Brzosko et al. 2018) and genetically more variable seed sources (Dalrymple et al. 2011, Dalrymple et al. 2012; Maschinski & Albrecht 2017). There is growing evidence that source material from mixed population leads to greater success in translocation than do single population sources (Vergeer et al. 2005; Godefroid et al. 2011; Maschinski et al. 2013). A created population should be larger than 50 individuals whenever possible (Albrecht & Maschinski 2012) to overcome the inevitable loss of some individuals due to transplant shock and demographic and genetic constraints associated with small population size (Guerrant 1996; Frankham et al. 2014). The planting density should allow efficient wind or animal pollination (Abeli et al. 2016, 2018). Introductions of long-lived species should preferably use outplants of different ages/life stages (i.e. both seedlings and saplings) if the environmental conditions at the introduction site vary spatially or temporally and the variation differentially affects plants of different ages or life stages (Albrecht & Maschinski 2012; Wendelberger & Maschinski 2016). In many cases, pre- or post-planting management intervention may be necessary for establishing outplants, e.g. soil preparation and mulching (Sinclair & Catling 2003; Maschinski et al. 2004; Monks et al. 2012), caging or fencing (Monks & Coates 2002; Maschinski et al. 2004; Daws & Koch 2015; Fenu et al. 2016), weeding and burning (Pavlik et al. 1993; Bowles et al. 1998), or supplemental watering (DeMauro 1993).

In plant introduction projects, it is essential to have planting maps, GPS locations, and unique identification tags with sequential numbers for the introduced plants. This is necessary for evaluating the success of the introductions through monitoring of the created populations and, ideally, assessing the likelihood of their persistence through population viability analysis (PVA). Although survival of the introduced plants and their reproduction are important demographic measures in the created population, they are insufficient to conclude that the new population is viable (Bell et al. 2003). A crucial parameter for long-term population persistence is recruitment of subsequent generations (Volis & Blecher 2021), and extent of the recruitment must allow attaining and maintaining minimum viable population (MVP) size as defined by a 95% probability of survival after 100 years. The most desirable outcome in an introduction project is colonization of adjacent areas by the recruits with appearance of new satellite populations.

When it is allowed logistically and by availability of planting material, introductions should be designed experiments with several treatments (e.g. plant densities, levels of shade). This will make possible adaptive management with continuous protocol improvements when poorly performing methods are discarded and more effective methods are utilized more widely.

Now let's have a look at what has been done in CA to date. The first in situ introduction of plants project in CA dates back to 2005 when adult individuals of a threatened relict shrub, Otostegia bucharica, were rescued from a site of imminent habitat destruction and transplanted into a nearby natural population of the species (Tojibaev et al. 2019). The next introduction study reported in the literature took place more than a decade later, in 2017. The greenhouse-grown seedlings of the red-listed herbaceous perennial *Taraxacum kok-saghyz* Rodin, known as kok-saghyz, was planted in several locations adjacent to existing populations of the species in Kazakhstan (Uteulin 2021). Two translocation projects are more than modest achievement considering the number of critically endangered species in the region. For comparison, 1001 conservation translocations involving 376 plant taxa have been performed in Australia (Silcock et al. 2019), and 222 translocations involving 154 species in China (Liu et al. 2015).

In situ: gaps in the existing PA network

As I already mentioned in this paper, the target 11 of the 2011-2020 Strategic Plan for Biodiversity is that “at least 17% of terrestrial and inland water areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas”. WDPA (https://www.protectedplanet.net/en/thematic-areas/wdpa?tab=WDPA) reports the following number of PAs and their area of coverage for five CA countries: 129 (10.0%), 35 (6.7%), 26 (22.3%), 32 (3.2%) and 37 (5.8%) (Kazakhstan, Kyrgyzstan, Tajikistan, Turkmenistan and Uzbekistan, respectively). Among these PAs, categories Ia and Ib represent 10, 6, 4, 8 and 8 PAs, and category II comprise 15, 6, 2, 0 and 3 PAs, respectively. The total areas for PAs from the categories I and II, calculated from the number reported by WDPA are 42,728 and 42,457 km^2, respectively, which is 0.90 and 1.06% of the CA territory. These numbers speak for themselves and need no further comment.

The majority of NRs (i.e. categories Ia and Ib PAs) in CA were established long ago based on expert knowledge. That their number and covered area are too small for efficient protection of biodiversity was not clear in the 1970s when the last large cluster of reserves was established in the CA republics of the

Soviet Union. This is clear today, and it is important to use systematic conservation planning for identification of imbalances and gaps in the current PA network. Areas of high conservation value rapidly disappeared in the region, but, fortunately, still exist. The last remaining patches of relatively undisturbed natural landscapes which are home to threatened species must become NPs (i.e. category II PAs) or, preferably, NRs. In addition, the existing network of PAs should be reinforced by restoration of those landscapes where changes can be reverted (e.g. degraded forests and pastures) and which have a strategic position connecting protected areas and allowing migration of animals and seeds / pollen to flow between them.

In situ: PA policies

It is interesting and quite helpful to compare the CA and Chinese PA policies and the major problems with their management because in both regions a unique and rich endemic flora is facing a crisis with hundreds of species being on the verge of extinction. In both regions, on paper, the situation is under control and numerous NRs do their job. In reality, they do not, and though some of the reasons are different, the major one is the same. This major reason is that neither the Chinese nor the modern CA NRs comply with the three-zone (core, buffer, and experimental zones) biosphere reserve model of UNESCO (Batisse 1982), although formally the NRs in both regions were managed on the basis of this model. In the three zone model, prohibition of visitation and activities in core zone surrounded by a buffer zone allowed only scientific research and ecological monitoring, and then by an experimental zone that not only allowed but supported economic development for the benefit of local residents while maintaining a ‘healthy environment’. However, these regulations are violated in the majority of NRs of both regions. In both regions, the major threat to PAs comes from the poor rural population. Certain damage by the villagers in both the buffer and core zones is ignored by the rangers. In China this is the collecting of medicinal plants and planting profitable cash crops in the forest understory (Volis 2018b). CA it is grazing. In China, the core zones of some NRs include villages and even towns (Zhang et al. 2017). In CA, they do not; but the villagers allow their herds to graze in core zones on a daily basis, and there is no real legislative mechanism to prevent them from doing so. As a result, the NRs’ core zone virtually does not differ from the buffer zone.

Solving this problem must focus on two aspects. One is addressing the steady growth of the population in rural settlements surrounding the PAs. This can be done by stimulating, e.g. by providing fellowships to the young generations to leave instead of staying in the villages neighboring PAs. The older village residents should be offered an attractive opportunity to purchase land and resettle in areas farther from the PAs. There is usually a direct positive relationship between the number of residents in the villages neighboring the PAs, overall size of their domestic stock and a degree of vegetation degradation in a PA core zone area within a herd’s daily grazing distance (ca. 5 km for goats and sheep). This relationship can be violated in cases when, in addition to local herds, there are also herds brought from remote places for seasonal grazing in areas around PAs. As areas available for herds as pastures become devoid of vegetation due to overgrazing, the practice of using the PAs as pastures for livestock becomes more and more common. Since rangers usually have close family and other relationships with the villagers, they turn a blind eye to this kind of activity. Documented cases of herd intrusion with their owners being punished according to the law virtually do not exist. The fines are rather symbolic, with usually no system controlling their payment. To address this problem, there must be a strong legislative enforcement of the core zone status as a zone with entrance for any unauthorized visitors (and especially livestock) strictly prohibited and strongly punished. Additionally, a 5 km zone around a core zone (regardless whether this is a buffer zone) must be defined as a zone within which livestock graze freely is banned. The possible measures against general growth of the domestic stock in settlements neighboring PAs is imposing taxes on cattle, sheep and goats.

If the above problem of controlling the domestic livestock intrusion into PAs is not solved legislatively, the only solution will be the physical prevention of their intrusion by fencing the most valuable areas within PAs, thereby creating a 'double core' zone.

Given de facto exploitation of not only buffer zones but also core zones in virtually every PA, they all, including those formally defined as I-II categories, can be classified as category IV or even VI only. An overhaul of the existing system of PAs is absolutely necessary before it is too late. A new system must comply with the IUCN rules in designating PA categories, and be supported by clear laws and a mechanism of their enforcement. Sufficient budgets must be allocated to provide rangers the appropriate wages and social guarantees, and to cover the operational conservation costs, such as ecosystem monitoring, patrolling and active interventions when needed. Every protected area must have a comprehensive monitoring program to trace the ecosystem changes over time. In addition to estimating every year the population size of the target species (which usually are flagship or most-threatened species), the abundance of a specific indicator species for a given ecosystem should be used for this purpose. The changes in population size of the target species, abundance of indicator species, degree to which the ecosystem is intact or the percentage of the PA area that is intact should be used to evaluate the effectiveness of a PA (and not the amount of time and money spent on monitoring and patrolling).

Role of botanical research

Herbaria and botanical research are the basic source of information for plant conservation (Rivers et al. 2011; López & Sassone 2019; Marsico et al. 2020). The herbarium collections are indispensable for determining the endangerment status and critical habitats of species. The knowledge provided by herbaria and botanical research, summarized in the form of Red Lists and proposals for land protection is used by governmental organizations responsible for nature protection in decision-making.

In general, botanical research pursued in academic institutions and botanic gardens of CA is a story of many impressive achievements which are beyond the scope of this paper. Unfortunately, those achievements are predominantly in such fields as taxonomy and vegetation science. Organismal biology, in contrast, is an undeveloped field. The environmental requirements and life history have been studied in situ for only a limited number of CA species, mostly trees and shrubs of economic importance (e.g. Zapryagaeva 1964; Nechaeva et al. 1973). For non-arboreal species, studies of life history, phenology and seed germination have been conducted exclusively ex situ (i.e. in botanical gardens). The data obtained in those studies are useful for maintaining species ex situ and their propagation, but only partly can be extrapolated to natural populations. Studies of population demography in situ exist, but use outdated methodology that do not allow predictions (e.g. Orozumbekov et al. 2015; Akhmedov et al. 2021). Plant-animal interactions (e.g. pollination, fruit and seed dispersal, seed predation) is a neglected field in CA botanical research, which is a pity because knowledge of plant-animal interactions can be crucial for working out the plans for threatened species conservation. For example, in the critically endangered *Otostegia buharica*, seed production is virtually absent. Examination of fruits revealed problems with fruit development and cases of seed predation by beetles (Tojibaev, personal communication), but more detailed research on seed production problems in this species has never been conducted.

Botanical research in CA must be significantly improved in the listed fields. This can be done by providing a certain number of fellowships each year for studying abroad in the specific and narrowly defined disciplines listed above.

Coordination between the CA countries

As was stressed earlier in several studies (Wilson et al. 2021; Volis 2022) it is important that the planning and implementation of conservation actions is done at the regional rather than national level because the CA countries share watersheds and mountain ranges and many endangered species are distributed across the borders of several countries. For animals, such efforts exist, for example identification of the transboundary hotspots and obstacles for migration of migratory wild animals in CA (Yakusheva 2017; Anonymous 2019, 2020). Analogous programs should be created for critically endangered plant species (e.g. those listed in Eastwood et al. 2009).

Coordinated efforts in both *ex situ* and *in situ* plant conservation will help to solve many problems that any conservation program is facing. Those efforts can include joint expeditions, sharing of not only knowledge on species occurrence, biology, propagation protocols etc. but also seeds, and easy access for researchers to the PAs.

Conservation research and information sharing

There is a strong imbalance in subjects of the regional conservation-related research. Field research having explicit conservation implications started to appear only in the last decade (all these papers are cited in Volis 2021). The experimental conservation field work (e.g. experimentally manipulated germination or seedling growth conditions, reintroduction/translocation experiments) is still absent. What predominates is list-making of endangered species or priority areas. This situation has to be changed. Efficient conservation in the region will be impossible without studies of the population demography of the species and environmental requirements, experimental reintroductions, creation of *ex situ* gene banks etc.

Interactive learning is a prerequisite for efficient conservation. Therefore, availability of the information on active or completed conservation programs is important for both learning and coordination of conservation efforts. Inadequate system of disseminating/obtaining information can have a strong negative effect on efficiency of conservation (Meek et al. 2015).

In CA, the majority of conservation-related literature is published in in books, local journals and reports in the Russian language. Because of a lack of an interaction with peers from the international conservation community, these studies are often inadequately described and analyzed. Even papers that are published in English suffer from usage of outdated methodology developed as long ago as in the 1960s and 1970s, are poorly connected to the practice, and emphasize planning rather than implementation.

Hopefully, this situation will change. A new journal, *Plant Diversity of Central Asia*, was created as an outlet for conservation field research and monitoring - topics that are underrepresented in conservation literature. In addition to original research papers, the journal will publish reviews analyzing the outcomes of conservation projects, summarize new conservation tools and present detailed methodologies that can be useful for managers of protected areas and other practitioners. The journal’s editorial board comprises experts in all aspects of conservation biology who will help young researchers to properly analyze and describe their data before communicating their research to the international audience.

Conclusions

The conservation strategy proposed for CA can be summarized as having the following crucial components:

- conservation planning and implementation has an ecoregional basis.
- regional systematic planning utilizes a unified scoring system that gives the highest priority to the most endangered local endemics with regeneration problems; this system is used in designing reserves, monitoring and assessment

of efficiency of a reserve network, and creation of seed banks and living collections.
- population demography and presence of naturally occurring regeneration are the key criteria for defining the status of species conservation and the major focus of the species recovery plan.
- seeds collected in natural populations are used for creation of multi-species living collections in semi-natural or natural environments (e.g. NR buffer zones)
- the living collections provide material for *in situ* actions such as reinforcement, reintroduction and translocation.

To successfully adopt and implement this strategy in CA requires stricter protection of NRs and NPs than is currently practiced and tighter coordination in the development and implementation of the conservation plan.

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