



Native and non-native species for dryland afforestation: bridging ecosystem integrity and livelihood support

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Abstract

• **Key message** We propose a silvicultural-ecological, participatory-based, conceptual framework to optimize the socioeconomic-ecological services provided by dryland afforestation, i.e. addressing the limited resources in arid areas while minimizing the harm to the environment. The framework applies the following criteria to select multifunctional tree species: (a) drought resistance, (b) minimal disruption of ecosystem integrity, and (c) maximization of ecosystem services, including supporting community livelihoods.

• **Context** Dryland afforestation projects frequently aim to combine multiple ecological and economic benefits. Nevertheless, plant species for such projects are selected mainly to withstand aridity, while other important characteristics are neglected. This approach has resulted in planted forests that are drought-resistant, yet harm the natural ecosystem and provide inadequate ecosystem services for humans.

• **Aims** We suggest a comprehensive framework for species selection for dryland afforestation that would increase, rather than disrupt, ecological and socio-economic services.

• **Methods** To formulate a synthesis, we review and analyze past and current afforestation policies and the socio-ecological crises ensuing from them.

• **Results** To increase afforestation services and to support human-community needs, both native and non-native woody species should be considered. The framework suggests experimental testing of candidate species for their compliance with the suggested species selection criteria. Furthermore, regional stakeholders are involved in evaluating, ranking, and prioritizing the candidate species according to experimental results and stakeholders' values and needs. We exemplify our approach by describing our ongoing research project, aimed to evaluate several native and exotic *Ziziphus* species in the Middle East region.

• **Conclusion** The employment of our proposed framework forms a novel community of native and non-native woody species. We discuss the ecological context of this proposal.

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Contribution of the co-authors This article is a synthesis and each author contributed from her field of expertise. Orna Reisman-Berman contributed in the fields of forestry, dryland afforestation, and ecology. Tamar Keasar contributed in the fields of ecology, pollination ecology, and bee foraging. Noemi Tel-Zur contributed in the fields of reproductive biology and the biology of *Ziziphus* spp. All authors wrote and edited the manuscript.

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1 Introduction

Drylands are characterized by scarce water resources (<~ 500 mm) annual rainfall, Aridity Index > 0.65 (UNEP 1992)), with a high spatial and temporal variability in precipitation that increases with aridity. Arid areas cover 40% of the world's earth and support 30% of its population. Dryland inhabitants consist the majority of the world's poor population, and as many as 16% of the inhabitants of drylands live in chronic poverty (<http://drylandsystems.cgiar.org>). Unsustainable land use practices, adverse climatic conditions, and population increases in many dryland regions contribute to land degradation, which impairs ecosystem services and contributes to food insecurity, carbon

emissions, and social and political instability (Yirdaw et al. 2017). These predicaments have fueled the action of afforesting, i.e., the formation of a novel tree ecosystem as one of the restoration tools to combat land degradation and decreased ecosystem services (Harris et al. 2006; Chazdon 2008). Accordingly, the socio-ecological aims of many dryland afforestation projects include conservation and regeneration of soil and biodiversity, water conservation, groundwater recharge, dust and flood prevention, and maintenance of a cool micro-climate. Some of the projects aim to produce food and timber, to generate recreation areas and even to create jobs (Mander et al. 2007; Nassauer and Opdam 2008; Scherr and McNeely 2008; Lovell and Johnston 2009; Chirwa and Mahamane 2017). The Great Green Wall for the Sahara and the Sahel Initiative, aimed to restore forest landscapes and degraded lands in Africa, is a pertinent example (Berrahmouni et al. 2014). Unfortunately, these goals have been only partially achieved. One of the main reasons for the lack of complete success lies in the species composition of the planted forests, which is the key to their effect on humans and the environment. Tree species for dryland afforestation were and still are selected mainly for drought tolerance and rapid growth and establishment (e.g., Ben Dov et al. 2001). Such a selection strategy ignores other biological characteristics of the trees and has severely damaged the integrity of some natural ecosystems, mainly due to tree invasiveness and habitat fragmentation.

While the pros and cons of afforestation of arid areas are currently under active discussion, it continues to be widely practiced. Recently, Yosef et al. (2018) demonstrated that large-scale dryland afforestation in arid environments can potentially enhance carbon sequestration, moisture penetration, and precipitation, influencing areas larger than the actual afforestation. Those modeling results, which are based on previous empirical results (Rotenberg and Yakir 2010), produce a strong recommendation to further increase the scale of dryland afforestation.

This paper calls for a comprehensive framework for dryland afforestation that would increase, rather than disrupt, ecological as well as socio-economic services. We argue that afforestation in drylands should include species that sustainably support livelihoods, can be extensively managed, and pose minimal risk to the integrity of the natural ecosystem. Based on past and present socio-ecological crises, we suggest a process to select woody species for dryland afforestation that would form multi-functional forests. Similar agendas were recently suggested for large-scale forest restoration projects, aimed at climate change adaptation and mitigation worldwide, rather than rehabilitation of arid ecosystems (Stanturf et al. 2015, 2017).

2 Ecological challenges and opportunities in dryland afforestation—past and present

The need for careful land management in arid regions is exemplified by the history of afforestation in Middle Eastern

drylands. Travelers in the 19th century described the vegetation in the area as dwarf shrublands with a few forest patches of *Pinus halepensis* Miller and *Quercus calliprinos* Webb (Paz 1979). These landscapes resulted from a long history of economic pressures and heavy grazing, which depleted the woody species populations of the native woodland and shrubland ecosystems (Pignatti 1983). As a result, the physiognomy of the woodland changed, triggering soil erosion and ecosystem degradation (Bottema et al. 1990; Lespez 2003). The degraded landscapes of the Middle East have thus motivated inhabitants throughout the region to establish large-scale afforestation projects attempting to reclaim and restore the ecosystems. Some major ecological crises ensued, owing to the traits of the tree species selected for these afforestation efforts. These crises include:

1. Altering of native plant communities by the invasion of afforestation species

The first risk associated with planted forests is the invasive spread of the planted species (Richardson 1998). The invasion and spread of a species depend both on the susceptibility of the environment to invasion (invasibility) (Lonsdale 1999) and on the traits associated with the species' ability to become invasive (species' invasiveness) (Goodwin et al. 1999). Non-native introduced species with long-lived seed banks and/or shoot resprouting abilities are hypothesized to persist in harsh environments and to eventually become invasive (Lawes et al. 2006). A well-known example is the introduced Australian acacia, *Acacia saligna* (Labill.) H. Wendl., which has established new populations outside planted areas. This significantly altered arid and semi-arid ecosystems over a wide geographical range (Wilson et al. 2011). Other examples of introduced invasive species include *Eucalyptus* spp. and *Prosopis juliflora* (Sw.) Dc. These species were planted in dryland afforestation projects in Africa, in the Middle East, and in arid zones around the Mediterranean basin, and their distribution has since extended into a wide range of ecosystems (Henderson 2001; Dufour-Dror 2012).

More recently, the spontaneous expansions of native species beyond their historical distribution range, or dramatic increases in their populations, have also been recognized as invasions. Such invasions often arise in response to human interventions, such as wildfire suppression, urbanization, or climate change (Carey et al. 2012; Valéry et al. 2013). A recent example is *Pinus halepensis*, native to the Eastern Mediterranean, which has spread from afforestations into ecosystems that the species does not inhabit naturally, including semi-arid grasslands, shrublands, and woodlands. The distance from the source population, precipitation, substrate type, woody vegetation density, and grazing pressure best explain *P. halepensis*' expansion success into new habitats

(Osem et al. 2011; Sheffer et al. 2014). Fire, however, does not have a significant effect on pine invasions (Osem et al. 2011).

2. Contamination of the native gene pool

This is another aspect of human-mediated species movement that is only starting to receive attention (Rejmánek 2014). The introduction of alien species that are closely related to local ones may result in new geographic overlaps with the native species. In these cases, therefore, hybridization between native and non-native species could occur. Such hybridization events might introduce new alleles into the indigenous gene pool with further risks to the local genotypes and native populations (Wolf et al. 2001; Barbour et al. 2010). There are currently no clear policies or protocols to test for potential hybridizations between native local trees and introduced species that are close relatives (Flory and D'Antonio 2015).

3. Inhibited establishment of native species under the afforestation canopy

Drought-tolerant plants are traditionally considered suitable for dryland afforestation. Yet, some of these species, such as *Eucalyptus* spp. and *Prosopis* spp., pose an ecological risk because of their allelopathic traits. Such trees release phytotoxic compounds into the environment, e.g., volatile oils in *E. camaldulensis* (Verdeguer et al. 2009) or water-soluble phenols in *P. juliflora* (Al-Humaid and Warrag 1998). These metabolites can strongly inhibit the germination of other plant species (Inderjit and Callaway 2003) or even the germination and establishment of congeneric species (Inderjit et al. 2008). In Middle East afforestations planted with *Eucalyptus* spp. or *Prosopis* spp., the establishment and further recolonization of native species under the afforestation canopy are strongly inhibited (Reisman-Berman et al. 2011). At the landscape scale, allelopathy in planted forests could interrupt vegetation landscape connectivity by impeding the dispersal and spread of native plant species through afforestation patches. This fragmentation of natural habitat patches could, in turn, restrict gene flow of native species (Honnay et al. 2002).

4. Changes in vegetation physiognomy, with potential ecosystem-level effects

Alteration of vegetation physiognomy, from a typical arid shrubland with sparse trees to patches of tall trees, may drastically affect ecosystem resource cycling and ecological networks. For example, planting tall and spaced trees ('savannization') in a desert afforestation project in Israel provided avian predators with perching and scouting points and thus increased their foraging efficiency. This, in turn, caused a decrease in lizard populations (Hawlena and Bouskila 2006). The transformation from shrub-dominated to tree-dominated landscapes in a semi-arid area reduced both the productivity of the herbaceous

species and the soil fertility beneath the canopies of the trees, as compared to the productivity and soil fertility under the shrub canopy in a native shrubland ecosystem (Paz-Kagan et al. 2016). The above examples emphasize the multidimensional response of an ecosystem to converting a shrubland into a forest.

3 Social challenges of dryland afforestation

Dryland afforestation programs should be designed not only to minimize ecological risks but also to provide multiple long-term benefits for local inhabitants. Increasing aridity and tree logging for fuel are the main factors causing land and forest degradation in natural dryland areas. Forest loss in these areas has been estimated to exceed 40% within four decades (De Waroux and Lambin 2012). Logging pressures are expected to increase even further, as the global need for wood is estimated to triple or even quadruple by 2050 (Fund for Nature 2011). These considerations point firmly to the need to plant and protect new trees for human benefit. In particular, in deprived areas, forests can provide food in the form of fruit and can also serve as a hunting habitat (Sunderlin et al. 2005). The introduction of merely a few productive fruit trees into subsistence farms can dramatically enhance household incomes (Neupane and Thapa 2001). One of the best examples of a tree species suitable for the provision of multiple ecosystem goods and services in drylands is Argan [*Argania spinosa* (L.) Skeels], which provides cattle fodder, wood for fiber and fuel, and seeds for the production of high-quality oil in southwestern Morocco. However, due to intensive exploitation and consecutive droughts, the species is under threat in its natural habitats (De Waroux and Lambin 2012).

In more prosperous countries, the utilization of forests changed from timber production at the industrial stage to a post-industrial stage, at which the forest is perceived as a multi-purpose functioning ecosystem (Sunderlin et al. 2005). Multifunctional forests support sustainable food provision, along with creation of jobs and additional provisioning and regulating ecosystem services (Lovell and Johnston 2009).

There are strong relationships between multi-functionality of the landscape, the involvement of the community in forest management, and the actual conservation of natural resources. These interactions can be optimized through 'participatory planning', namely, public participation in forest planning (Buchy and Hoverman 2000). The power of participatory planning is in the integration of local knowledge with science-based approaches (Agrawal and Gibson 1999). It has been advocated as a necessary step for any forest restoration project (Hanson et al. 2015). In North America and Europe, it often aims to sustain the forest as a functioning ecosystem that conserves biodiversity (Aerts and Honnay

2011). In developing countries, participatory planning aims to manage and conserve natural stands and agroforests to provide basic food and regular income (German et al. 2006; Reij et al. 2009; Xu et al. 2012; Sacande and Berrahmouni 2016).

The advantages of participatory planning have been particularly well demonstrated in a study conducted in a rural semi-arid region of Spain (García-Llorente et al. 2012). This study showed that the more multifunctional the landscapes, the more conservation support they received from the local communities. Accordingly, the conversion of multifunctional landscapes to mono-functional ones disturbed the capacity of rural areas to provide diverse ecosystem services and hence, reduced community support for their conservation.

Today, more and more reforestation and afforestation projects are aimed at providing multi-purpose functioning ecosystems (Rudel et al. 2010). Nevertheless, in practice, multifunctional participatory planning is more commonly being integrated into afforestation projects in wealthy communities than in deprived populations and in disadvantaged areas of the world. Thus, multifunctional afforestation in drylands still remains a formidable challenge (Lambin and Meyfroidt 2010; Wolfersberger et al. 2015).

4 What is missing? A framework to maintain ecosystem integrity, increase afforestation services, and support community livelihood in drylands

To reduce the ecological risks of afforestation projects, The Convention for Biological Diversity (<http://www.cbd.int/forest/tools.shtml>) provided guidelines for forest planning and management. The guidelines recommend planting of native woody species, aiming at increasing the similarity between human-planted and native forests. We argue that in drylands, in particular, these guidelines are insufficient in the following aspects:

- (1) We argue that it is cardinal to develop agendas and protocols to test and select woody species for dryland afforestation based both on their contribution to the local human communities and on their impacts on nearby ecosystems. We suggest that species selection processes should be fed from (a) lessons learnt from past forest crises and (b) the results of experiments, specifically designed to test the contribution and impact of candidate afforestation species in a specific dryland region.
- (2) We claim that, in drylands, the exclusive use of native trees might decrease afforestation ecosystem services, in particular in supporting local community livelihoods, and that native species are not always optimal. Instead, we suggest that non-native species also be considered as afforestation candidates after they have been experimentally evaluated

for potential ecological risks and services. It may be argued that planting native species could preclude environmental crises caused by the introduction of alien species (Hobbs et al. 2006). However, there are several scenarios for which this supposition is not true. First, native species can expand their distribution with afforestation activities and outcompete other species, as discussed above regarding *Pinus halepensis*. Second, native species are well adapted to the local environment, but when used in dryland forestry, they are sometimes planted at the margins of their ecological distribution and in degraded habitats, which could possibly lead to tree mortality (Dorman et al. 2015) with no regeneration (Osem et al. 2013). Third, native species do not always offer all the characteristics that are required of a multifunctional forest, especially in terms of livelihood support. Non-native species that are closely related to the native ones may often be more suitable in that they are likely to have both similar ecological suitability and performance and similar aesthetic and cultural values (Hodder and Bullock 1997), but can potentially provide extra benefits. Thus, the introduction of closely related species from abroad may address the important need for additional ecosystem services. Exotic species, closely genetically related to the native ones, should be introduced only if it is possible to avoid any spontaneous hybridization with their native relatives and further genetic contamination (Barbour et al. 2010; Carey et al. 2012). Additional aspects of invasiveness, such as resprouting, vegetative spread, and spontaneous establishment, need to be tested in the native and alien candidate trees, before they are planted in large-scale forest stands.

Non-native trees have been widely used to increase ecosystem services and support livelihoods, particularly in areas characterized by limited resources (Vilà and Hulme 2017). Their contribution to those aims has been lately re-acknowledged. Yet, it is also clear that forestry programs often fail to harness the advantages of introduced species to a win-win situation with no or little negative effect (Witt 2017).

This means that there is a delicate balance between the desired spontaneous forest regeneration required to maintain forest sustainability and the “out of control” regeneration that can turn the non-native species to invasive ones. Our suggested framework addresses this challenge.

- (3) We call for multifunctional afforestation. In the past few decades, planting of multifunctional woody species has slowly begun to receive attention within forestry projects that aim to enhance rural livelihoods (German et al. 2006; Chazdon 2008).
- (4) We stress the need to integrate participatory planning in afforestation. We think that past and present afforestation projects have not paid sufficient attention to the needs

and preferences of local inhabitants. The integration of stakeholders in planning of forest management has become a crucial element in modern silviculture, ecology, and other environmental disciplines (Lovell and Johnston 2009; Houehanou et al. 2011). Participatory planning of tree ecosystems has so far been applied to single-use forests, and a protocol for community involvement in the design of multi-functional afforestations is yet to be developed. The selection of tree species for multiple benefits, by a defined set of criteria, has been demonstrated only in a few instances and has not necessarily been performed for afforestation purposes but rather for genetic tree improvement (Franzel et al. 1996). Any program for participatory planning should be based on prioritization, a process in which multiple evaluation criteria are ranked and weighed (Barazani et al. 2008). A pioneering study on the selection of trees by a set of criteria was conducted by Reubens et al. (2011). In that study, trees were ranked for land rehabilitation by a multi-criteria decision-based approach. While an excellent starting point for multi-species afforestations, this study is limited in two respects: it is based on a meta-analysis of the scientific literature and therefore suffers from unavoidable knowledge gaps that have to be addressed experimentally. In addition, as the authors pointed out: “It is essential that such final assessment is made together with local stakeholders, particularly those user groups potentially vulnerable to land or tree use restrictions” (Reubens et al. 2011).

5 The proposed framework

To address the gaps described above, we suggest an integrative framework for selecting species assemblages for multifunctional and sustainable dryland afforestation (Fig. 1). First, we suggest three principal criteria for species selection. Those criteria are based on past and current socio-ecological silvicultural crises. Accordingly, the selected species should (a) withstand the arid environment, including compatibility with dryland soil conditions; (b) minimally risk and damage to ecosystem functions through invasiveness, cross hybridization, suppression of local species, and disruption of forest physiognomy; and (c) contribute to community livelihoods. Next, we suggest four steps that involve the selection and the evaluation of the species through a feedback process between the scientific, forest managerial, and the community stakeholders (Fig. 1):

Step 1: Surveying local trees and their closely genetically related non-local woody species over a wide eco-geographical region, to form a list of candidate species. These species should comply with the values,

needs, experience, knowledge, and expectations of land owners, managers, and other stakeholders. The candidate non-local woody species should add value and enhance ecological services provided by the local species that currently thrive in the region or that used to flourish there historically. This step crystalizes specific benchmarks of the community beyond the principle criteria. However, it also requires the constant examination of evoking local crises and risks that should be included in the list of criteria. This step involves the academia and the whole range of stakeholders from the local community, forest managers, and practitioners.

- Step 2:** Testing the candidate native and non-native woody species experimentally, over a wide range of aridity conditions, for their compliance with the main and specific local criteria. We believe that the experimental approach will reduce knowledge gaps and facilitate region- and site-specific decision-making regarding the list of desired species for local drylands. This step involves the scientific community.
- Step 3:** Prioritizing the examined species according to the experimental results (step 2), the principal criteria and the community needs and values. This step involves both local communities and scientists.
- Step 4:** Long-term monitoring of the afforestation and the surrounding ecosystem and in situ feedback aimed at interactive adaptive management, with on-going feeding of information to the community, forest managers, and decision makers.

Step 1 concurs with the United Nations FAO’s guidelines for restoration of degraded forests and landscapes in drylands, which stress the need to combine technical criteria with local preferences when selecting afforestation species (Berrahmouni et al. 2015a, b). Furthermore, the FAO guidelines emphasize assessment and monitoring of forest restoration projects as components of adaptive management, in agreement with step 4 of our framework.

The novelty of the proposed dryland afforestation framework is in developing a simple step-wise process that bridges scientific studies, social needs, and values shaped by stakeholders from the community, governmental organizations, and NGOs.

The innovation in the field of ecology resides in identifying candidate species that do not endanger ecosystem functioning and integrity. Such dangers may arise from the simple action of man-mediated species movement to form a novel or designed ecosystem. We distinguish between the risks of invasiveness, i.e., the spread of species from the afforestation to surrounding ecosystems and beyond, and the risks that are related to the afforestation as a novel ecosystem that interacts with its adjacent natural ecosystems. The latter category includes the risks of cross-pollination between the local and the planted species, allelopathy, and its effect on species movement through the

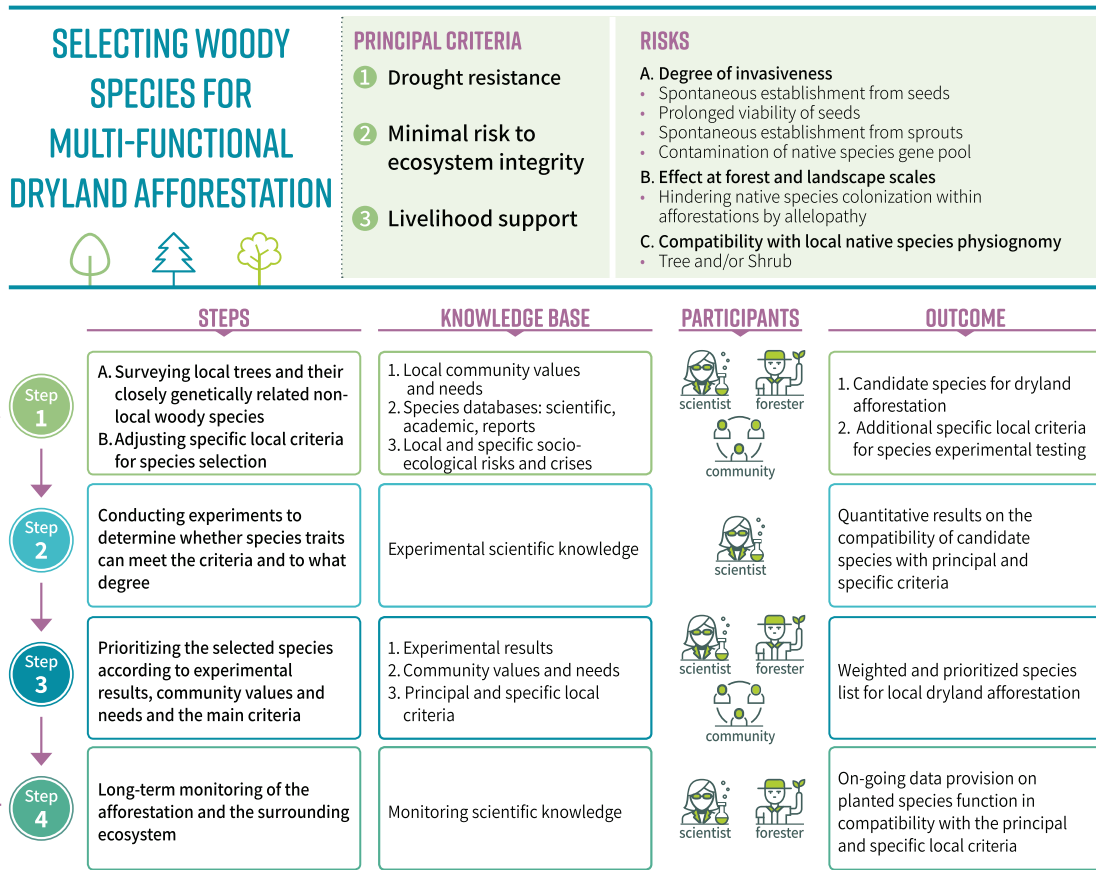


Fig. 1 Illustration of the suggested conceptual framework for selecting native and non-native woody species for multifunctional dryland afforestation. The framework suggests considering criteria and potential risks based on past and current socio-ecological silvicultural crises. The

process involves experimental testing and evaluation of the species through a feedback process between the scientific, forest managerial, and the community stakeholders

forest; and the changes in the natural vegetation physiognomy that may occur with the introduction of afforestation.

Hereby, we illustrate the proposed framework by suggesting five species of the genus *Ziziphus* for dryland afforestation in the Middle East. Presently, various *Ziziphus* species are being planted in some dryland afforestation projects that aim to integrate native species (Bozzano et al. 2014). In other projects, *Ziziphus* species are introduced because of their contribution to provisioning ecosystem services (Owens and Lund 2009). We exemplify the proposed use of native and non-native trees by considering three species that are native to the region of the Middle East and two that are non-native.

6 *Ziziphus* as a case study—biology and ecology of the candidate genus

Species of the genus *Ziziphus* Mill. (Rhamnaceae) are spiny shrubs and small trees distributed in warm-temperate and subtropical regions throughout the Old and New Worlds (Islam and Simmons 2006). Several *Ziziphus* species are native to arid and semiarid ecosystems and are intrinsically adapted to

dry and hot climates. Therefore, they have excellent potential for dryland afforestation (Mizrahi and Nerd 1996). We focus on five promising species: *Z. mauritiana* Lam., *Z. jujuba* Mill., *Z. spina-christi* (L.) Desf., *Z. lotus* (L.) Lam., and *Z. nummularia* (Burm.). These five species provide multiple and complementary benefits (Outlaw et al. 2002; Saied et al. 2008; Pandey et al. 2010), as shown in Table 1. They have the potential to satisfy some of the suggested criteria for dryland afforestation: *Z. spina-christi*, *Z. lotus*, and *Z. nummularia* are native to the region. *Z. lotus* is a shrub and therefore provides an additional advantage by fitting into the physiognomy of dryland shrublands. *Z. mauritiana* and *Z. jujuba* are non-native to the region, originating in the Far East and central Asia, respectively. However, they provide additional livelihood-supporting resources, which include attractive edible fruits and complementary foraging resources for honeybees throughout the dry season (Vashishtha 1997; Pareek 2001; Outlaw et al. 2002; Azam-Ali et al. 2006; Haddad et al. 2008; Pandey et al. 2010). The berry fruits falling on the ground also contribute to soil fertility, as was demonstrated in *Z. mauritiana* (Singh et al. 2012) and *Z. spina-christi* (Tessema and Belay 2017) in drylands savannas. Improved

Table 1 Origin, classification, and traditional uses of five *Ziziphus* species. Summarized from: Aekerman 1961; Singh 1963; Vashishtha 1997; Danin 1992, 2001; Mirzahi and Nerd 1996; Pareek 2001; Outlaw et al. 2002; Abu-Hamdah et al. 2005; Dafni et al. 2005; Azam-Ali et al. 2006; Saied et al. 2008; Keasar and Shmida 2009; Tel-Zur and Schneider 2009; Pandey et al. 2010; Maraghi et al. 2014; <http://www.cabi.org/isc>; <http://www.worldagroforestry.org>

Species	Common name	Native geographical distribution, chorotype	Life form and phenology	Tolerance	Species specific traditional uses and livelihood support
<i>Z. jujuba</i>	Jujube, Chinese date	West China, Central and South-West Asia. Chorotype: Euro-Syberian	Small tree Deciduous. Flowering ^a : March-May	Extreme low and high temperatures	
<i>Z. mauritiana</i>	Ber	South and Central Asia, China. Introduced to the Middle East and North Africa in pre-historic times. Chorotype: Pluri-regional, Sudanian.	Shrub or small tree. Evergreen Flowering ^a : June-October.	High temperatures and severe drought	Leaves for fodder. In India leaves are gathered to feed tasar silkworms. Timber for house construction, firewood.
<i>Z. spina-christi</i>	Christ's thorn	Mauritania through the Sahara, Sahelian zones of west Africa to the Red Sea and to Northern Israel and Syria. Chorotype: Sudanian	Shrub or small tree Evergreen Flowering ^a : March-December	Drought and moderate tolerance to frost	Traditional medicine. Leaves for livestock fodder. Spiritual use of the tree crown among Muslim communities.
<i>Z. lotus</i>	Sedra, rubeida	Around the Mediterranean Basin, including dune areas of the Sahara in Morocco, Tunisia and Spain. Chorotype: Mediterranean-Saharo-Arabian	Small shrub. Deciduous. Flowering ^a May-June	Drought	Traditional medicine. Spiritual use of the tree crown among Muslim communities. Dune stabilization
<i>Z. nummularia</i>	Jharber	Iran to India. Chorotype: Sudanian	Tall shrub. Evergreen, shed leaves in extreme high temperatures Flowering ^a : June-July.	Drought, salinity and extremely high temperatures	Sweet fruit for home use. Honey plant. Excellent fodder from leaves. Traditional medicines extracted from leaves and fruits. Wood for home use.

^a Flowering time in the species' natural habitat

soil fertility may, in turn, result in increased herbaceous biomass and species richness under the canopy (Tessema and Belay 2017). Thus, planting these species mixed with other dryland afforestation species is expected to promote multi-functionality by combining drought resistance with livelihood support.

7 A framework to integrate *Ziziphus* into a multi-functional afforestation

To exemplify the framework, we consider the three principal criteria and risks, starting with an already defined list of species (step 1, Fig. 1). Therefore, we focus here on step 2—the experimental step of our suggested framework. This step tests the woody species' traits according to a set of socio-ecological criteria (Fig. 1). We pose a series of questions and experiments to address them, to be investigated for each criterion (Table 2). The timeframe of the experiments ranges 1–3 years, followed by in situ long-term monitoring at the afforestation site.

7.1 Criterion 1: drought resistance

Drought resistance is a complex of mechanisms and is difficult to assess by a single measure. Rather, it requires a set of developmental and physiological experiments that reveal the response of a species under specific conditions. To date, some studies have shown drought tolerance in *Z. mauritiana* (Clifford et al. 1998) and *Z. jujuba* (Cruz et al. 2012). Our suggested research questions include: which species develop and reproduce best under drought conditions? Does increased water availability improve the performance of the species and how is plant performance related to its drought tolerance? Is there a tradeoff between drought tolerance and the provisioning traits? To answer these questions candidate *Ziziphus* species should be grown in experimental plots, where their survival, development, and physiological performance are monitored.

7.2 Criterion 2: minimizing the risk to ecosystem integrity

Here, we focus on four features of ecosystem integrity disruption: invasion potential, hybridization and gene-pool contamination, allelopathy, and interference with the physiognomy of the local native vegetation. Up-to-date observations and monitoring of *Z. mauritiana* show that the species is invasive in some hot and humid countries but not in dry climate countries (CABI <https://www.cabi.org/isc/datasheet/57556>). We ask what is the ecosystem invasibility, i.e., to what extent are drylands sensitive to *Ziziphus* invasion? Simultaneously, we ask whether *Ziziphus* can become invasive? Are *Ziziphus* seeds viable for long periods, generating risks of invasion by spontaneous establishment, in particular in arid environments? Can establishment from resprouting shoots be limited under drought conditions? Can accumulated runoff in specific areas promote sprout establishment? These questions can be answered by an array of experiments that include seed burial trials in natural dryland ecosystems, tests of seed-bank viability, spontaneous establishment from seeds and resprouting, and general monitoring of vegetative development.

An additional set of experiments should evaluate the potential of the tree species to hybridize with local native species. We hypothesize that there might be also some risk of hybridization among the candidate *Ziziphus* species, as there is some overlap in the flowering seasons and spatial distributions of *Z. spina-christi*, *Z. jujuba* and *Z. mauritiana* (Asatryan and Tel-Zur 2014). Reciprocal interspecific artificial crossing between *Z. spina-christi* and *Z. mauritiana*, as well as between *Z. jujuba* and *Z. spina-christi*, resulted in the formation of viable seeds (Asatryan and Tel-Zur 2014). *Z. nummularia* also overlaps with these three species in blooming season (Feinbrun-Dothan and Danin 1991), hence could potentially hybridize with them if planted together. The major experimental question is whether any gene flow occurs among species, resulting in gene contamination and/or the establishment of interspecific hybrids. Experiments that involve hand-cross pollination among species, monitoring of fruit setting, and embryo viability are needed to test this question.

Table 2 Examples for experiments to evaluate *Ziziphus* spp. suitability for dryland afforestation

Criterion	Experimental manipulation	Measure category
Drought resistance	Water availability	Plant survival and growth rate, flower and fruit production, morphological and physiological signs of wilting
Minimizing ecological risk	Seed burial experiments	Seed viability, germinability
	Time from seed dispersal	Seed viability, germinability
	Water availability	Resprouting, germinability
	Cross-pollination	Fruit set, embryo viability, hybrid seed germination
Livelihood support	Introduction of honeybees	Honey production, foraging preferences, fruit yield

In testing for allelopathy, we ask: Do the leaves and fruits of the candidate species inhibit the germination, growth and survival of dryland species? Do they affect soil chemistry? Are allelopathic effects influenced by precipitation levels? Here simple bioassay experiments can be conducted under greenhouse and field conditions.

Finally, the changes in plant physiognomy as a result of afforestation and their ecosystem-level effects are the most difficult to measure. Using *Ziziphus* species as a case study, we ask: How will afforestation with *Ziziphus* trees affect ecosystem integrity, compared to afforestation with *Ziziphus* shrubs (*Z. lotus*), in particular where the natural vegetation of the arid environment has a shrubland physiognomy? This requires long-term monitoring of plots in afforestations and in the surrounding local ecosystem. Various measures of ecosystem functions should be recorded, including species biodiversity, food web structure and mineral cycling. Experiments of this kind are long-term, and thus unlikely to provide shorter-term insights for forest planning. Nevertheless, we believe that such experiments are essential and must be accompanied by adaptive management. To the best of our knowledge, compatibility of the architecture of introduced plant species with the physiognomy of local species has not been comprehensively studied in an ecological context.

7.3 Criterion 3: livelihood support

Ziziphus spp. flowers are an important source of nectar for high-quality honey production, and the fruits of some species are edible, providing potential sources of income for rural people (Table 1). Experiments should therefore test whether honeybees have preferences among *Ziziphus* species, and whether *Ziziphus spp.* nectar and pollen can adequately support the bees. Other experiments should focus on fruit yield and leaf biomass under dryland conditions.

8 Discussion: wider implications

8.1 What constitutes a good species assemblage in planted forests?

The use of non-native species to complement native trees in dryland afforestation is a contentious aspect of our proposed framework. The issue of non-local species becomes particularly thorny when considering afforestation projects aimed at ecosystem restoration and reclamation, i.e., forming plantations that highly resemble natural forests or the native plant community (Barlow et al. 2007). In this section, we briefly discuss a large body of recent ecological research, which maintains that human modifications of natural plant communities are inevitable and sometimes even ecologically beneficial. These ideas can provide a theoretical justification for

combining local and alien trees in sustainable planted desert forests.

Views on plant species compositions have developed from a rigid structure of plant communities (Clements 1916) to a random species assemblage, presented by Hubbell (2001) as the Neutral Theory that unifies theories of biodiversity and biogeography. According to this view, species establishment is driven by random processes, and the diversity and distribution of plant communities represent the outcome of random birth, death, dispersal, and speciation events. Currently, the neutral theory is playing an increasingly central role in the planning of species assemblages and in predicting restoration outcomes within the discipline of restoration ecology (Fukami et al. 2005; Hector et al. 2011; Barnes et al. 2014). This theory allows for considerable variability in the composition of species planted for habitat restoration, as would be expected in a natural community governed by stochastic processes.

Additional views on the desirable assemblage of species in restored habitats include Rosenzweig's (2003) reconciliation ecology, which sees an opportunity in redesigning anthropogenic habitats, allowing the coexistence of broad assemblages of native and exotic species. Along similar lines, Hobbs et al. (2011) have stressed that restoration projects cannot turn the clock back to reconstruct pristine ecosystems, and therefore carefully thought out interventions are needed. This point of view can be particularly relevant to dryland forests, which have often been managed and exploited by humans during millennia for various provisioning services such as wood, fodder and fruit (Yirdaw et al. 2017; Berrahmouni et al. 2015a, b). Defining a "natural" state in such forest or savanna ecosystems might be misleading. The need for a thorough experimental approach, embedded in adaptive management, is termed "intervention ecology" (Hobbs et al. 2011), an approach that fits well with our proposed framework. Within this field we stress both the needs for an experimental approach and for adaptive management. Moreover, we follow the essence of this approach, which suggests that effective interventions should take into account the social and cultural history of the sites to be managed, together with current societal needs (Hobbs et al. 2011). We acknowledge that our framework also carries a price tag—the extra time and expenses needed for experiments, adaptive management, and stakeholder involvement.

We must thus ask: What are the implications of these theories for the selection of species assemblages for afforestation? Forest ecologists still do not know whether the selection of species assemblages that are comprised of the natural species community would recreate an ecosystem that maximizes ecosystem services. This uncertainty is reflected in the ongoing debate on "assisted migration," namely, translocation of trees outside their natural geographic distribution to reduce extinction risks due to climate change (Hewitt et al. 2011; Pedlar et al. 2012). Opponents point to risks of invasions (Mueller and Hellmann 2008) and the lack of sufficient

understanding of the impacts of the introduced species (Ricciardi and Simberloff 2009). Supporters of this strategy, on the other hand, argue that native ecological communities cannot be conserved or maintained in the face of rapid anthropogenic change, thereby justifying active translocation of climate-endangered plants (Thomas 2011). Both proponents and critics agree that the sharp distinction between natural and novel ecosystems is becoming blurred in the era of global warming. By analogy, sustainable forestry in human-dominated drylands may require species assemblages that differ from those of the native local woody species communities.

A few studies have investigated the role of natural vs. planted forest stands as providers of ecosystem services. Overall, studies on restoration projects have demonstrated a strong correlation between the increase in biodiversity and the gain of ecosystem services (Benayas et al. 2009). Barlow et al. (2007) have shown that primary forests have irreplaceable value in supporting biodiversity, although exotic tree plantations can provide complementary conservation services to those provided by natural stands. These findings were supported by Bremer and Farley (2010), who suggested that plantations best conserve biodiversity when they are established on degraded lands, rather than in natural ecosystems such as forests, grasslands, or shrublands. This is because man-made afforestations support biodiversity more effectively than degraded areas, but less well than natural ecosystems. Other studies have demonstrated that even systems based solely on exotic trees, such as agroforestry systems, maintain high species diversity when properly managed. Such afforestations can therefore play an important role in biodiversity conservation in human-dominated landscapes (Bhagwat and Willis 2008). It has also been recommended that the usefulness of diverse species assemblages with various traits be tested as a sustainable forest management strategy (Garnier and Navas 2012). The underlying rationale is that mixed diverse forests are more resistant to disturbances than single-species forests (Vilà et al. 2007). This approach complies with the restoration recommendations developed by Harris et al. (2006). These authors warn that by insisting on the exclusive use of local plant material (species/population/provenance), we may be consigning restoration projects to a genetic dead end, which does not allow for rapid adaptation to changing environments in the era of climate change.

To conclude, taken together, the above-described studies and concepts form a framework that applies ecological principles to arid zone afforestation. This framework is based on the conventional wisdom that the species assemblages chosen should resemble the natural plant communities. However, the details of the construction of the species assemblages may vary along a gradient from 'natural' to 'no-analog' (Jackson and Hobbs 2009) or 'novel' assemblages (Hobbs et al. 2006), with a decreasing affinity for the historical geographical distribution of the species (Seddon 2010). In practice, this conceptual gradient widens the range of selected species to a more

diverse one that includes both native and non-native species. Integrating alien species in dryland afforestation projects, after careful experimental testing for ecological compatibility, may indeed complement the ecosystem services provided by native species. We can envisage a continuum starting at no intervention in the natural ecosystem (wild) through restored ecosystems, via afforestation, through more intensive agroforestry systems up to an extensive coppice, industrial forest, or an orchard ecosystem. Our framework offers an ecosystem designed to be an extensively managed, livelihood-supporting afforestation, somewhere between afforestation and agroforestry along this continuum.

Adapting a new framework for afforestation in drylands is a mandate in the era of climate change, desertification and increased poverty in arid areas. All of these call for an integrative afforestation framework that would bridge between ecosystem integrity and livelihoods.

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Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

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