Flora 207 (2012) 693–700

Contents lists available at SciVerse ScienceDirect

Flora

journal homepage: www.elsevier.de/flora

Restoring degraded arid Mediterranean areas with exotic tree species: Influence of an age sequence of *Acacia salicina* on soil and vegetation dynamics

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ARTICLE INFO

Article history:
Received 21 February 2012
Accepted 22 June 2012

Keywords:
Acacia salicina
Exotic species
Soil characteristics
Vegetation dynamics
Age
Arid Mediterranean areas
Tunisia

ABSTRACT

Acacia salicina Lindl. is an Australian leguminous tree widely planted outside its original distribution as a multi-purpose species, and successfully establishes on degraded areas. The objective of this study was to determine how an age sequence of 3, 5, 9 and 13-year-old *A. salicina* plantations affected soil properties and flora diversity of an arid Mediterranean steppe. Soil samples were taken from the upper 10 cm of soil under tree canopies and in the associated open spaces. The results showed that tree establishment and development enhanced soil contents of total C, total N, available P and exchangeable K^+, Ca^{2+}, Mg^{2+} and Na^+. These trends increased significantly with increasing plantation age. At the same time *A. salicina* facilitated the colonization and development of understory vegetation. Indeed, the number of plant species, the total plant cover, the perennial species density, the plant biomass and the diversity all attained higher values under tree canopies and increased with increasing plantation age. The soil dynamic under 13-year-old *A. salicina* plantations reflected two phases in the restoration sequence, characterized by nominal changes during tree establishment (0–5 years) and showing marked and rapid changes associated with the start of canopy closure (5–13 years). *A. salicina* establishment could be an effective and applicable measure to restore soil and vegetation and control desertification in the Mediterranean arid steppes.

Introduction

In arid and semiarid Mediterranean degraded ecosystems, exotic fast growing species have been successfully used for restoration, as they provide ecosystem services that native species may be unable to supply (Dumancic and Le Houérou, 1980; Gasques and Garcia-Fayos, 2004; Jeddi et al., 2009; Tiedeman and Johnson, 1992). Several studies, which contribute to the debate on the suitability of using exotic species outside their natural range, have shown that these species frequently complicate restoration projects by changing ecosystem properties, reducing native biodiversity and altering hydrologic regimes (Mack and Lonsdale, 2002; Richardson et al., 2008; Wilson et al., 2011). However, other studies, have highlighted the relevant role of invasive exotic species in the restoration process (Ehrenfeld, 2003; Ewel and Putz, 2004; Geldenhuys, 1997; Jeddi et al., 2009; Levine et al., 2003; McNamara et al., 2006; Norisada et al., 2005; Reubens et al., 2011; Reubens, 2010). Some ecological traits of these species, including N fixation, fast growth, and resistance to stress, contribute to their ability for colonizing new areas (Vitousek, 1990; Yelenik et al., 2004; Yang et al., 2009). These traits can be of interest to foster ecological restoration as an alteration of ecological conditions and resources flow may promote the establishment of late successional species (Jeddi et al., 2009; Levine et al., 2003). Indeed, the introduction of exotic woody species to degraded areas has become increasingly important as a measure to enhance soil nutrient concentrations and to promote vegetation cover establishment (Chen, 2001; Levine et al., 2003). In some specific areas and communities, afforestation of exotics could also provide better conditions for the establishment of native shade tolerant species without threatening native forests, which have proven to be resilient to invasion by exotic species (Geldenhuys, 1996).

Acacia salicina Lindl. (Willow wattle) is an Australian leguminous shrub or small tree introduced to many regions as a multi-purpose species (Rehman et al., 1999), and successfully establishes on degraded areas (Grigg and Mulligan, 1999). Because of its high growth rate and tolerance to bare soil, this species has been commonly planted in degraded Mediterranean arid and semi-arid areas (Correal et al., 1988; Le Houérou, 1986). However, in these environments, the net effect of this exotic woody

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http://dx.doi.org/10.1016/j.flora.2012.07.002

species on soil properties and understory vegetation dynamics has received less attention (Jeddi et al., 2009; Jeddi, 2009). However, such knowledge is necessary to understand ecosystem function and community dynamics in order to develop sound management programs. Moreover, several studies have shown that characteristics of plant patches like species identity, age, and size strongly influence seed banks, seedling recruitment, composition of understory vegetation (Facelli and Brock, 2000; Maestre and Cortina, 2005), and can strongly influence community structure and dynamics (Pugnaire and Lázaro, 2000).

In this work, our objectives were to investigate changes in soil characteristics and plant communities following the establishment of an age sequence of artificial *A. salicina* plantations in an arid *Stipa tenacissima* steppe in southern Tunisia.

Materials and methods

Study approach

A common approach in studies of soil restoration in relation to plant cover is to monitor plant and soil changes occurring along a chronosequence developed on similar soils under similar climatic conditions (Bhojvaid and Timmer, 1998; Stevens and Walker, 1970; Su and Zhao, 2003). This approach was adopted in this study because of the availability of closely located (within 3 km) *A. salicina* plantations established 0-, 5-, 9-, and 13-year ago on a fenced *S. tenacissima* steppe with a common history of grazing use. Therefore, in spite of the short establishment period (13 years), the plantations (representing a chronosequence), provide a time gradient of tree occupancy on similar sites.

Study site

We conducted our study in the SE of Tunisia, approximately 20 km south-east of the town of Sfax (34°41′60″N, 10°30′22″E). The local name of the site is El Gonna. The climate is Mediterranean lower arid with temperate winters (Emberger, 1954). According to the soil survey reported by FAO (1990), the main soil class is Regosol with friable caliches at 10–25 cm depth and gypsum outcrops. Topography is hilly. The average annual rainfall is 196 mm, with more than 70% of precipitation between November and April. The average annual temperature is 18.6 °C. The study area has been exploited as a summer rangeland which mainly is grazed by sheep and goats in an extensive-grazing system. Since 1995, it was protected from grazing by the Forest Service of Sfax, and gradually planted by forest native species. The main objective of the exclosure and the plantations in El Gonna is to restore vegetation in areas that were heavily disturbed by intensive human use. The main species introduced was *Pinus halepensis*. However some areas were planted with the exotic species *A. salicina*. Average planting density was 300 trees ha^{-1}, resulting in an average tree spacing of about 6 m × 5.5 m.

The area is occupied by an overgrazed *Stipa tenacissima* L. steppe, showing some indicator species of the presence of gypsum such as *Lygeum spartum*, *Atractylis serratuloides*, *Gymnocarpos decander*, *Erodium glaucophyllum*, and *Helianthemum lippii* ssp. intricatum and some plants such as Artemisia herba-alba, reflecting vegetation recovery after grazing exclusion.

Experimental design

Three representative plantations (5-, 9-, and 13-year) and an adjacent site with no plantation (0 age) similar in physiographic and soil features, were selected as the experimental sites. In early April of 2008, we selected three 40 m × 40 m plots within the experimental sites for sampling. The plots were separated ca. 100 m from each other. In each plot, we randomly selected three *A. salicina* individuals (i.e. nine individuals in each age group) and associated open spaces. Three random locations were also selected in each plot within the control area. Morphologic traits of *A. salicina* in each age group are listed in Table 1.

Table 1
Morphologic traits of *Acacia salicina* plantations.

Traits	Plantation age (year)		
	5	9	13
Mean height (m)	3.5 ± 1.2	5.9 ± 0.7	6.3 ± 1.1
Crown diameter (m)	3.1 ± 0.61	4.3 ± 0.97	5 ± 0.82

Soil sampling

Soil samples were collected from the upper 10 cm of soil excluding litter and stones. We sampled in two sub-habitats for each tree: beneath the canopy (at 50% radius), and outside of the canopy (at 150% radius). All samples were sieved to pass a 2 mm mesh. Soil pH and electrical conductivity (EC) were determined (Saturated paw method, AFNOR, 1987) by pH meter and conductivity meter, respectively. Total nitrogen (N) was determined by Kjeldahl method and phosphorus (P) by the Olsen method (Olsen and Sommers, 1982). Oxidable soil organic matter (C) was determined by using the Walkley–Black method (Nelson and Sommers, 1982). Data were transformed to indicate organic carbon by using a transformed Van Bemmelen factor of 2 (Nelson and Sommers, 1982). Potassium (K^+), calcium (Ca^{2+}), magnesium (Mg^{2+}) and sodium (Na^+) were determined by atomic absorption spectrometry (AAS 6800, Shimadzu).

Vegetation sampling

Understory vegetation properties including floristic composition, perennial species density (calculated as the perennial plant number per m^{-2}), and total plant cover percentages were measured in four replicate quadrats, each of 50 cm × 50 cm, located at four aspects around each tree (N, S, E and W). The plant biomass of aboveground parts of vegetation (Bm) was assessed by the formula of Le Houérou (1987): Bm (kg ha^{-1}) = $r \times 43.1 \pm 3.6$ (r, represents the cover of perennial plant species). Diversity was calculated using the Shannon–Wiener index (H'). Plant nomenclature follows Chaieb and Boukhris (1998).

Data analysis

Data were statistically analysed using SPSS (11.0) for Windows. Soil and vegetation parameters were tested for significant differences among treatments in two-way ANOVA. Differences between means were tested using Tukey HSD post hoc test with a level of significance of $p < 0.05$.

In order to compare changes in soil properties under the canopy of *A. salicina* during phases of plantation development, relative differences (RD) were calculated by: RD = $(A/B) \times 100$, where B represents total change in soil parameters over 13-year period, and A denotes changes observed in particular phase.

Results

Soil properties

Total C, total N and available P under tree canopies were found to be significantly higher than in the open spaces (Fig. 1; Table 2).

Table 2
Results of analysis of variance for soil properties in *A. salicina* plantations of different age and microsite.

Soil properties	C (mg g^{-1})	N (mg g^{-1})	P (mg kg^{-1})	Ca^{2+} (mg kg^{-1})	K^+ (mg kg^{-1})	Na^+ (mg kg^{-1})	Mg^{2+} (mg kg^{-1})	pH	EC (ds m^{-1})
Plantation age effect	<0.001	<0.01	<0.05	<0.001	<0.001	<0.01	<0.05	NS	NS
Microsite effect	<0.001	<0.001	<0.001	<0.001	<0.001	0.055	<0.001	NS	NS
Age × Microsite	<0.001	<0.05	NS	NS	NS	<0.01	<0.05	NS	NS

NS, not significant.

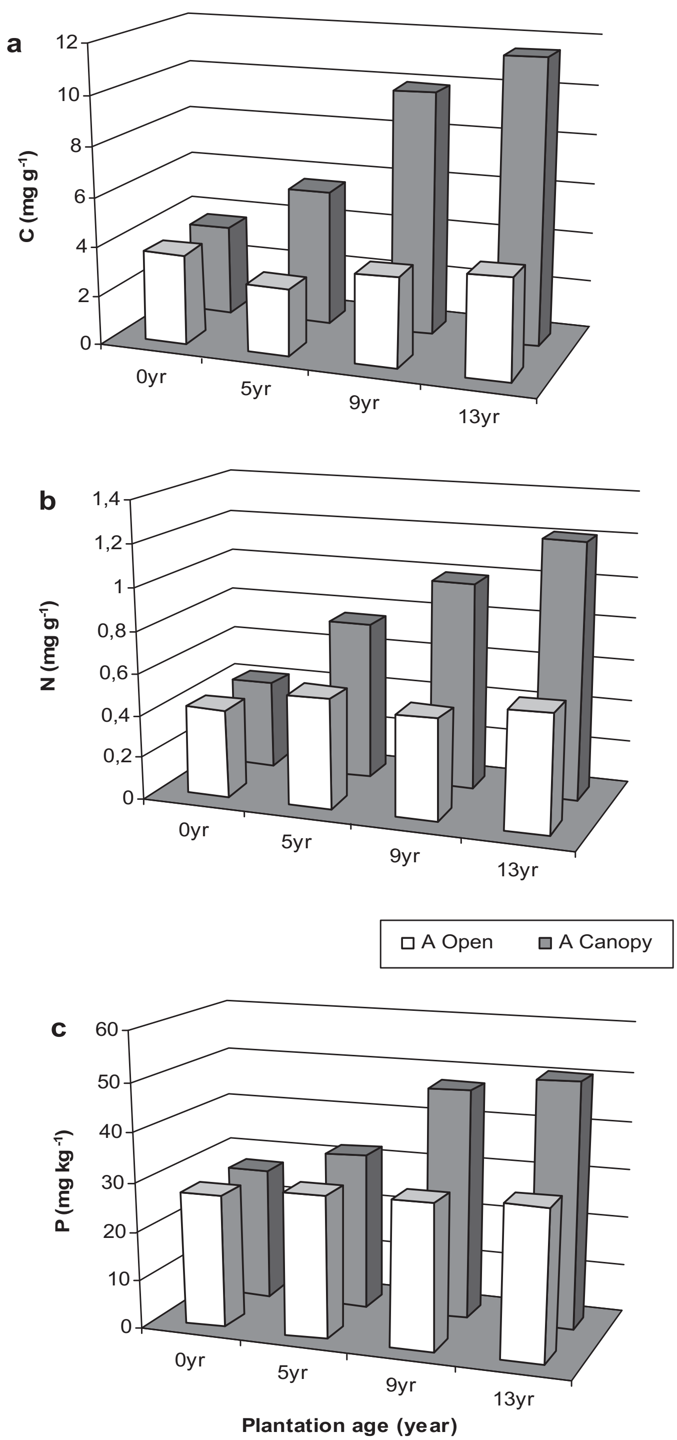

Fig. 1. Change of soil organic C (a), total N (b) and available P (c) with increased *A. salicina* plantation age.

There was a marked increment in these soil elements during the plantation period, from their original values. After 5, 9 and 13 year tree growth, soil carbon levels increased by 1.5, 2.7 and 3.2 times those in the control. Total N and available P accumulation followed the same pattern with respect to the plantation age with 3 and 2 times increased values, respectively, after 13-year tree growth. The establishment and development of the *Acacia* plantations also resulted in a significant enrichment of exchangeable K, Ca and Mg, but the values of increase were lower than those of C, N and P with, respectively, 1.7, 1.7 and 1.8-fold increase reached in the soil after 13 years of tree growth (Fig. 2a–c; Table 2). Conversely to other ions, Na^+ levels in soil showed a significant decrease with age, where values dropped from 16.2 to 8.9 mg kg^{-1} (Fig. 2d; Table 2).

Soil pH and EC where higher under tree canopies than in open spaces (Fig. 2e and f). However, differences between both microsites were not significant. Conversely to the previously mentioned other variables, EC and pH mean values were relatively constant with increasing plantation age (Table 2).

In Fig. 3 two phases are distinguished of the relative changes of specific soil properties that occurred early (0–5 years) and later (5–13 years) in the succession process. The maximum change was 46.9% (for soil K^+ and Mg^{2+} contents) and 75.6% (for soil carbon content) for the 0–5-year and the 5–13-year period, respectively, reflecting large and rather rapid changes that, however, became moderated in the second period.

Development of plant species

Vegetation parameters changed under the tree canopies more strongly than in the open spaces (Table 3). Except for the youngest *A. salicina* plantations (5 years existing), significant differences were found between the below and between the canopies situated microsites.

A vegetation composed by higher amounts of native plant species developed gradually in the *A. salicina* plantations. In addition to the increased total number of species present, also cover, perennial plant density, biomass and diversity increased with increasing plantations age (Tables 3 and 4). Initially (0–5 years), the herbaceous community developed slowly, with only a few annual (*Atractylis carduus*, *Erucaria pinnata*, *Coris monspeliensis*) and perennial (*Ajuga iva*) species colonizing. Species number increased from nine to twelve, vegetation cover from 24% to 27%, plant density from 14 plants m^{-2} to 18 plants m^{-2}, biomass from 1243 to 1548 kg ha^{-1} and diversity from 2.1 to 2.3. However, except for the biomass, none of the parameters differed significantly from each other in the early age classes (0 and 5-year). In the 9-year-old plantation, 20 species were invaded, the cover reached up to 52 per cent, density of perennials 24 plants m^{-2}, biomass 2240 kg ha^{-1} and diversity 2.9 but there were still more short-lived annual species than perennial species (11 and 9, respectively). In the 13-year-old plantations, all vegetation parameters increased further, and the number of species reached 31, including 18 perennial species such as *Argyrolobium uniflorum*, *Retama raetam*, *Deverra tortuosa*, *Atractylis serratuloides* and *Stipa tenacissima* (Table 4).

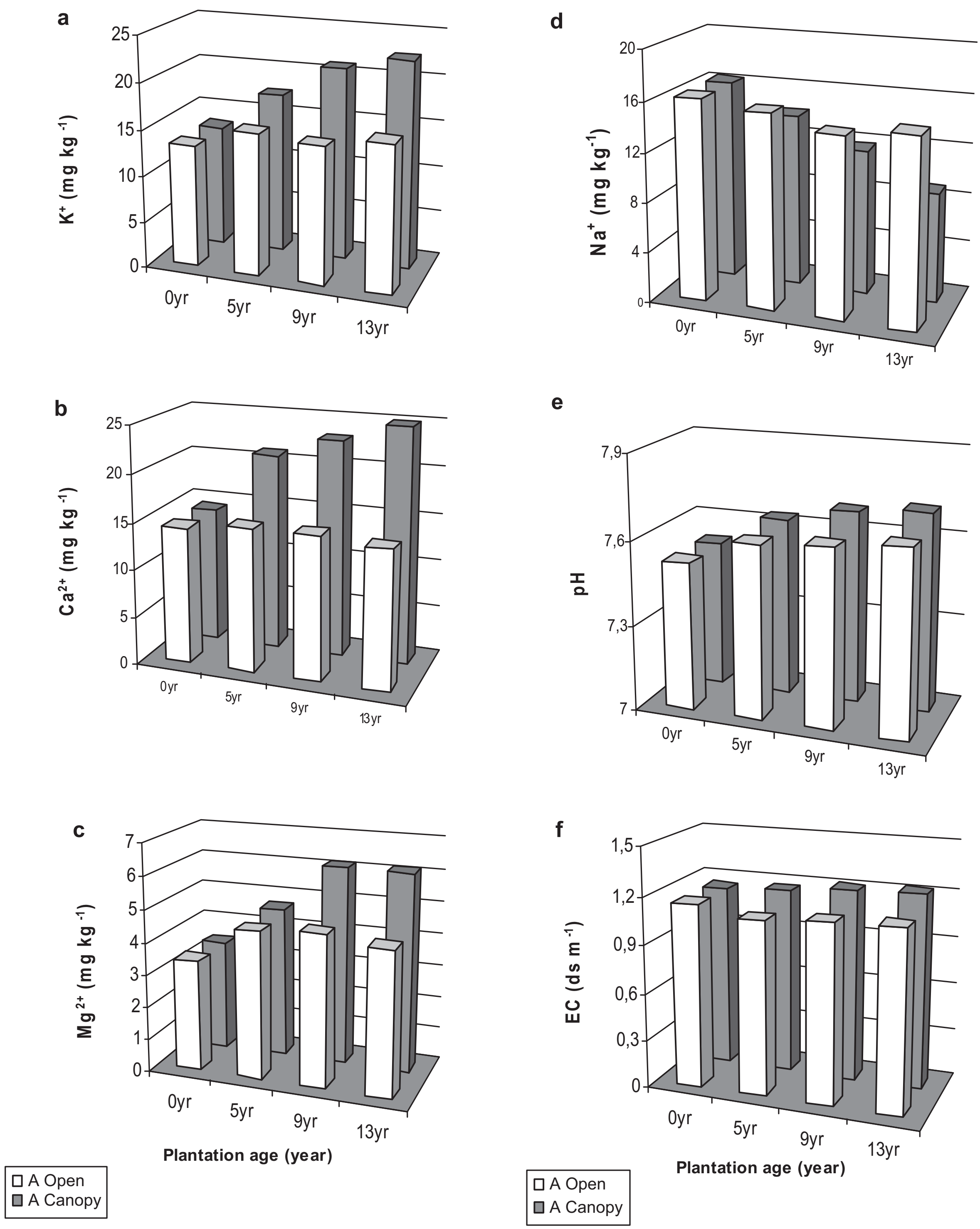

Fig. 2. Change of K^+ (a), Ca^{2+} (b), Mg^{2+} (c), Na^+ (d), soil pH (e) and EC (f), with increased *A. salicina* plantation age.

Discussion

Change of soil characteristics after Acacia salicina *establishment*

In the present study, establishment of the N-fixing *A. salicina* resulted in significant accumulations of organic C, total N, available P, exchangeable K, Ca and Mg under the canopies, producing "islands of fertility" surrounding each shrub. This is consistent with results elsewhere (Caravaca et al., 2003; Wezel et al., 2000). These positive effects increased with plantation age. The important changes in soil nutrient contents may be related to the high productivity and high aboveground and belowground inputs in these afforestation areas. Indeed, *Acacia* species establishment enhanced the rates of colonization and the growth of other plants, and paved the way for vegetation expansion by ameliorating stressful environmental conditions (Jeddi et al., 2009; Reubens et al., 2011; Yelenik et al., 2004). In southeast Asia, Kuusipalo et al. (1995) suggested to plant introduced *Acacia* ssp. into degraded lands. The rationale behind this recommendation was that the topsoil can

Table 3
Changes in the understory vegetation community after *A. salicina* establishment.

Vegetations parameters	Sampling location	Plantation age (years)			
		0	5	9	13
Total plant cover (%)	Canopy	24a	27a	52b	68c
	Open		23	30	29
	Microsite effect		NS	<0.01	<0.001
Perennial species density (number per m^2)	Canopy	14a	18ab	24b	29c
	Open		13	17	18
	Microsite effect		NS	<0.05	<0.01
Biomass (kg ha^{-1})	Canopy	1243a	1548b	2240c	2811c
	Open		1330	1620	1702
	Microsite effect		NS	<0.01	<0.001
H'	Canopy	2.1a	2.3ab	2.9b	3.2c
	Open		2.1	2	2.3
	Microsite effect		NS	<0.05	<0.01

Values are means. Values with same letter within columns are not significantly different at $p < 0.05$; NS, not significant.

be improved and the desired woody vegetation would then be favoured at the expense of more light-demanding grasses. The resulting more heterogeneous light environment would favour a higher degree of biodiversity. Also in other places where soil was eroded and the original vegetation had almost disappeared, *Acacia* species – which have high growth rates, tolerance to poor soil, and substantial nitrogen-fixation ability – have been widely used for afforestation (Nichols and Carpenter, 2006; Peng et al., 2005; Yang et al., 2009). Some species of the genus, like *Acacia auriculiformis* and *Acacia mangium*, were introduced to degraded tropical and subtropical regions to establish forest communities (McNamara et al., 2006; Norisada et al., 2005; Peng et al., 2005). These trees could not only improve soil nutrients, but by shielding against intense radiation and heat load, they could also facilitate colonization by other plants (Lugo, 2004; Parrotta et al., 1997). Therefore, they were considered as nurse plants for understory native species (Yang et al., 2009).

With increasing plantation age of such *Acacia* afforestations, the development of understory herbs and the corresponding accumulation of foliage and litter cover provides an important input of C and N, but also a significant decrease of soil salinity (Whitford, 2002). Also in the here investigated site soil fertility increased with plantation age, demonstrating the progressive development of the restoration process. Two distinct phases could be distinguished in the chronosequence: an initial establishment phase (years 0–5) and a following enrichment phase (years 5–13). Since the initial establishment phase occurs before canopy closure, it is characterized by a harsh microclimate of the stands, with large temperature amplitudes (Bhojvaid and Timmer, 1998). Examining the restoration effects of tree plantations, Sanchez (1987) and Bhojvaid and Timmer (1998) described such succession patterns on degraded sub-humid sites. The beginning of the canopy closure is the start of

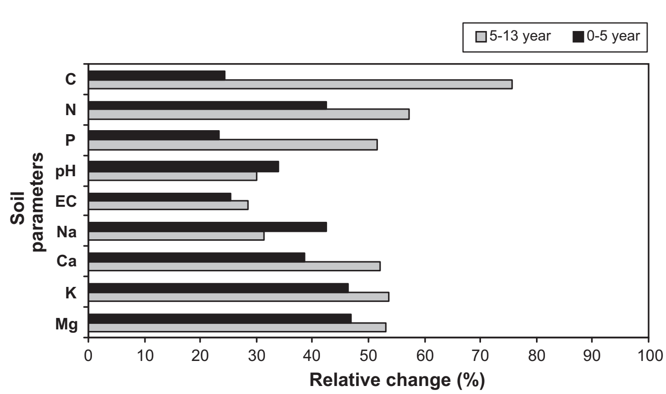

Fig. 3. Relative differences (RD) in changes of soil properties under the canopy of *A. salicina* during two phases of plantation development: 0–5 years and 5–13 years.

the faster enrichment phase, which coincides with increased litter production and tighter nutrient recycling because of an expanding root system (Fisher, 1995). Also in the present investigation site, this phase was marked by significant reductions in soil salinity and a build-up of organic carbon content and macronutrients in soil as compared to soil status in the initial tree-establishment phase.

Understory colonization and development of the floristic diversity

Acacia salicina had a strong positive influence on native understory vegetation. In this study, the magnitude of total plant cover and biomass obtained under *A. salicina* plantations (68% and 2811 kg ha^{-1}, respectively) is higher than that reported under the indigenous *P. halepensis* plantations developed at the same study site (65.3 and 2240 kg ha^{-1}, respectively; Jeddi and Chaieb, 2009) and to that reported under the indigenous *A. raddiana* developed in Bou Hedma National Park in the south of Tunisia (64.7% and 1546 kg ha^{-1}, respectively; Abdallah et al., 2008). Appreciation of the capacity of exotic *Acacia* species to positively affect native community composition and ecosystem function in semiarid lands is controversial (D'Antonio and Meyerson, 2002; Wilson et al., 2011). Derbel et al. (2009) reported that this species could have either a positive or a negative effect, depending on the balance between positive and negative interactions with autochthonous neighbours. Dense *Acacia* plantations did negatively affect the seed bank of native species in the South African fynbos and hampered the establishment of native seedlings (Holmes, 2002). An increased fertility of soil in *Acacia* stands may foster post-disturbance weed colonization (Yelenik et al., 2004). Competitive exclusion effects by the exotic tree species may be less likely in arid areas than in areas where climatic conditions are closer to the species' optimum (i.e. average rainfall > 350 mm – Duke, 1983).

The increased soil fertility, mainly total N, associated with *Acacia* plantation establishment may favour the presence of some nitrophilous species such as *Polygonum equisetiforme*, and some leguminous species such as *Retama raetam* and *Argyrolobium uniflorum*, which use (and in terms of nitrogen contribute also to) the amount of soil nutrients under the tree canopy. The improvement of soil fertility under the *A. salicina* canopy, and particularly the accretion of organic matter, is apparently the primary source of understory vegetation growth (Pugnaire et al., 2004). More importantly, the results of the present study indicate that *A. salicina* has been acting as a nurse plant providing favourable environmental conditions for the establishment of native species, such as *Rhus tripartita* and *Stipa tenacissima*. Soil protection and shade can be considered the main factors responsible for the establishment of these native species. No exotic species were found in addition to *A.*

Table 4
List of plant species present under the canopy of *Acaciq salicina* plantations of different age. Family, presence (1) and absence (0) of each species are shown. P: perennial, A: annual.

Family	Species	P/A	0-year	5-year	9-year	13-year
Anacardiaceae	*Rhus tripartita*	P	0	1	0	1
Apiaceae	*Deverra tortuosa*	P	0	0	0	1
Asteraceae	*Artemisia campestris*	P	0	0	1	1
	Artemisia herba alba	P	1	0	0	1
	Atractylis carduus	A	0	1	0	1
	Atractylis serratuloides	P	0	0	0	1
	Launea resedifolia	P	0	0	1	1
	Scorzonera undulata	P	0	1	1	1
Borraginaceae	*Echium humile*	P	0	0	0	1
Brassicaceae	*Diplotaxis harra*	A	1	1	1	1
	Erucaria pinnata	A	0	1	1	1
Caryophyllaceae	*Gymnocarpos decander*	P	0	0	0	0
Cistaceae	*Helianthemum kahiricum*	P	0	0	1	0
Dipsaceae	*Scabiosa rizantha*	A	0	0	1	1
Fabaceae	*Argyrolobium uniflorum*	P	0	1	1	1
	Medicago minima	A	0	0	1	1
	Retama raetam	P	0	0	0	1
	Hippocrepis areolata	A	0	0	1	1
Geraniaceae	*Erodium glaucophyllum*	P	0	0	1	1
	Erodium laciniatum	A	0	0	1	1
Lamiaceae	*Salvia aegyptiaca*	P	1	0	0	1
	Ajuga iva	P	0	1	0	0
	Teucrium polium	P	1	1	1	1
Plantaginaceae	*Plantago amplexicaulis*	P	0	0	0	1
Poaceae	*Brachypodium pinnatum*	A	1	1	1	1
	Bromus madritensis	A	0	0	1	1
	Cynodon dactylon	P	1	0	1	0
	Hyparrhenia hirta	P	1	0	0	0
	Koleria pubescens	A	1	0	0	1
	Lygeum spartum	P	0	1	0	0
	Stipa capensis	A	1	1	1	1
	Stipa tenacissima	P	0	0	0	1
Polygonaceae	*Polygonum equiseteforme*	P	0	0	1	1
Primulaceae	*Anagallis arvensis*	A	0	0	1	1
	Coris monspeliensis	A	0	1	1	1
Scrophulariaceae	*Kickxia aegyptiaca*	P	0	0	0	1
Zygophyllaceae	*Fagonia cretica*	P	0	0	0	1
Number of perennial species		**24**	**5**	**6**	**9**	**18**
Number of annual species		**13**	**4**	**6**	**11**	**13**
Total number of species		**37**	**9**	**12**	**20**	**31**

salicina. More notably, *A. salicina* did not regenerate in the experimental plots nor showed active colonization by suckers.

Similar ecological functions by *Acacia* species have been reported in other degraded areas. In south China, Yang et al. (2009), reported that *Acacia* trees can act as nurse plants for native understory species by increasing soil nitrogen and soil organic matter, thus improving conditions for photosynthesis, and also buffering air and soil temperatures. The shading provided by the tree species reduces photoinhibition in understory plants, which, in tunr, is beneficial to accumulate more biomass from carbon assimilation.

In present investigation, the Shannon–Wiener index (*H*') was greater under the canopies than in the open. This result agrees with those by Akpo and Grouzis (2004), who found a positive effect of trees on understory vegetation diversity in Senegal. According to these authors, the distribution of species was regular enough in this environment (i.e. under canopies), and, therefore, the ecological stability was higher than elsewhere.

Species composition and coverage increased gradually with increasing plantation age. These results can be paralleled with those of Su and Zhao (2003), in the semiarid northern China, who noted that this effect was mostly due to the soil conditions improvements. Pugnaire and Lázaro (2000) suggest also that in such vegetation patches the effect of the canopy changes with shrub age are due to both the physical protection offered by the canopy and the increase with time of soil organic matter and nutrients. Interaction between these soil resources and microclimate promote the formation of different niches that, in turn, increase in number and availability as canopy size increases (Pugnaire et al., 1996). This increased versatility of available habitats, together with the amelioration of the harsh climatic conditions typically found in semi-arid environments, favour facilitative interactions between plants (Callaway, 1995; Maestre and Cortina, 2005; Sanjerehei et al., 2011). The perennial species appearance and their density increase, with increasing plantation age in the current study, favour soil stability and improve the water balance and therefore, promote the establishment of other species (Jauffret, 2001). The high perennial species number and density at higher plantation age gives an idea on the ability of the ecosystem to regenerate (Aronson et al., 1993).

Conclusion

Using exotic woody species as a restoration tool can be an important strategy to combat desertification and degradation in Mediterranean arid and semi-arid areas. Their use in restoration must be carefully evaluated, however, and consider their effects on microhabitats. Further, their role as potential weeds must be carefully balanced with their positive effect on soils and plants.

We have shown that *Acacia salicina* can improve soil fertility and facilitate the establishment and growth of the understory vegetation, that has reached an appreciable diversity 13 years after plantation of the trees. Thus, in degraded sites of the North African rangelands it seems to be a useful restoration tool, suitable to create islands of resources and to foster succession. We may note, however, that our study site has been protected from grazing since the area was afforested. The impact of the studied species on soil properties and, especially, on accompanying understory vegetation would be lower if the area was grazed.

The use of this *Acacia* species can be important because it is used for fodder production, fuel, furniture construction and tannin production, apart from its positive effects on soil and vegetation. However, outside its natural range, careful control is needed in order to avoid negative effects on community composition and ecosystem functions. Indeed in other, more humid, Mediterranean areas the species can behave as an aggressive colonizer. Thus, spread during wet years, and into suitable areas cannot be ruled out, and this should be taken into account when integrating all advantages and disadvantages of the use of *Acacia* species. When native species are a priority, their use represents a good alternative, despite their lower ability to fulfil some of the objectives of restoration.

References

Abdallah, F., Noumi, Z., Touzard, B., Ouled Belgacemc, A., Neffati, M., Chaieb, M., 2008. The influence of *Acacia tortilis* (Forssk.) subsp. *raddiana* (Savi) and livestock grazing on grass species composition, yield and soil nutrients in arid environments of South Tunisia. Flora 203, 116–125.

AFNOR, 1987. Recueil de normes francaises, qualite des sols, methodes d'analyses. 1. edit. Association francaise de normalisation (Afnor), pp. 19–30.

Akpo, L.E., Grouzis, M., 2004. Interactions arbre/herbe en bioclimat semi-aride: influence du pâturage. Sécheresse 15, 253–261.

Aronson, J., Floret, C., Le Floc'h, E., Ovalle, C., Pontanier, R., 1993. Restoration and rehabilitation of degraded ecosystems in arid and semi-arid lands. A view from the South. Restor. Ecol. 1, 8–17.

Bhojvaid, P.P., Timmer, V.R., 1998. Soil dynamics in an age sequence of *Prosopis juliflora* planted for sodic soil restoration in India. Forest Ecol. Manage. 106, 181–193.

Callaway, R.M., 1995. Positive interactions among plants. Bot. Rev. 61, 306–349.

Caravaca, F., Figueroa, D., Barea, J.M., Azcon-Aguilar, C., Plenzuela, J., Roldan, A., 2003. The role of relict vegetation in maintaining physical, chemical and biological properties in an abondoned *Stipa*-grass agroecosystem. Arid Land Res. Manage. 17, 103–111.

Chaieb, M., Boukhris, M., 1998. Flore succincte et illustrée des zones arides et sahariennes de Tunisie. Association pour la Protection de la Nature et de l'Environnement, L'Or du temps, Sfax, Tunisie.

Chen, L.Y., 2001. Cost savings from properly managing endangered species habitats. Nat. Areas J. 21, 197–203.

Correal, E., Sanchez-Gomez, P., Alcaraz, F., 1988. Les espèces ligneuses à usages multiples des zones arides Méditerranéennes. In: Agrimed Research Programme. Commission of European Communities Brussels Belgium.

D'Antonio, C., Meyerson, L.A., 2002. Exotic plant species as problems and solutions in ecological restoration: a synthesis. Restor. Ecol. 10, 703–713.

Derbel, S., Cortina, J., Chaieb, M., 2009. *Acacia saligna* plantation impact on soil surface properties and vascular plant species composition in central Tunisia. Arid Land Res. Manage. 23, 1–18.

Duke, J.A., 1983. Handbook of Energy Crops. Unpublished. Available online at <www.hort.purdue.edu/newcrop/duke_energy/dukeindex.html> (accessed 03.04.08).

Dumancic, D., Le Houérou, H.N., 1980. Acacia cyanophylla Lindl. as supplementary feed for small stock in Lybia. In: Le Houérou, H.N. (Ed.), Browse in Africa. The Current State of Knowledge. Internat.Livestock Center for Africa, Addis Ababa.

Ehrenfeld, J.G., 2003. Effect of exotic plant invasions on soil nutrient cycling processes. Ecosystems 6, 503–523.

Emberger, L., 1954. Une classification biologique des climats. Recueil des travaux du Laboratoire de Botanique, Série botanique 7, 3–43.

Ewel, J.J., Putz, F.E., 2004. A place for alien species in ecosystem restoration. Front. Ecol. Environ. 2, 354–360.

Facelli, J.M., Brock, D.J., 2000. Patch dynamics in arid lands: localized effects of *Acacia papyrocarpa* on soils and vegetation of open woodlands of South Australia. Ecography 23, 479–491.

FAO, 1990. Soil Map of the World. Revised Legend. World Soil Resources Report 60.

Fisher, R.F., 1995. Amelioration of degraded rain forest soils by plantations of native trees. Soil Sci. Soc. Am. J. 59, 544–549.

Gasques, M., Garcia-Fayos, P., 2004. Interaction between *Stipa tenacissima* and *Pinus halepensis*: consequences for reforestation and the dynamics of grass steppes in semi-arid Mediterranean areas. Forest Ecol. Manage. 189, 251–261.

Geldenhuys, C.J., 1996. The Blakwood Group system, its relevance for sustainable forest management in the southern Cape. S. Afr. For. J. 178, 15–24.

Geldenhuys, C.J., 1997. Native forest regeneration in pine and eucalypt plantations in Northern Province, South Africa. Forest Ecol. Manage. 99, 101–115.

Grigg, A.H., Mulligan, D.R., 1999. Biometric relationships for estimating standing biomass litterfall and litter accumulation of *Acacia salicina* on mined land in central Queensland. Aust. J. Bot. 47, 807–816.

Holmes, P.M., 2002. Depth distribution and composition of seed-banks in alien-invaded and uninvaded fynbos vegetation. Aust. Ecol. 27, 110–120.

Jauffret, S., 2001. Validation et comparaison de divers indicateurs des changements à long terme dans les écosystèmes méditerranéens arides: applications au suivi de la désertification dans le sud tunisien. Thèse de Doctorat. Université des Sciences d'Aix-Marseille, 360 pp.

Jeddi, K., 2009. La restauration écologique en milieu aride de Tunisie: Cas de la protection et du reboisement du périmètre d'El Gonna, Sfax. Thèse de Doctorat, Université des Sciences de Sfax, 142 pp.

Jeddi, K., Chaieb, M., 2009. Using trees as a restoration tool in Tunisian arid areas: effects on understory vegetation and soil nutrients. Rangeland J. 31, 377–384.

Jeddi, K., Cortina, J., Chaieb, M., 2009. *Acacia salicina*, *Pinus halepensis* and *Eucalyptus occidentalis* improve soil surface conditions in arid southern Tunisia. J. Arid Environ. 73, 1005–1013.

Kuusipalo, J., Adjers, G., Jafarsidik, Y., Otasamo, A., Tuomela, K., Vuokko, R., 1995. Restoration of natural vegetation in degraded *Imperata cylindrica* grassland: understorey development in forest plantations. J. Veg. Sci. 6, 205–210.

Le Houérou, H.N., 1986. Salt tolerant plants of economic value in the Mediterranean basin. Reclamation Revegetation Res. 5, 319–341.

Le Houérou, H.N., 1987. Aspects Météorologiques de la Croissance et du Développement Végétal Dans les Déserts et Les Zones Menacées de Désertification. PNUE, Nairobi et OMM, Geneve.

Levine, J.M., Vilà, M., D'Antonio, C.M., Dukes, J.S., Grigulis, K., Lavorel, S., 2003. Mechanisms underlying the impacts of exotic plant invasions. Proc. R. Soc. Lond. B 270, 775–781.

Lugo, A.E., 2004. The outcome of alien tree invasions in Puerto Rico. Front. Ecol. Environ. 2, 265–273.

Mack, R.N., Lonsdale, W.M., 2002. Eradicating invasive plants: hard-won lessons for islands. In: Veitch, C.R., Clout, M.N. (Eds.), Turning the Tide: The Eradication of Invasive Species. SSC Invasive Species Specialist Group, IUCN, Gland, Switzerland, pp. 164–172.

Maestre, F.T., Cortina, J., 2005. Remnant shrubs in Mediterranean semi-arid steppes: effects of shrub size, abiotic factors and species identity on understorey richness and occurrence. Acta Oecol. 27, 161–169.

McNamara, S., Viet Tinh, D., Erskine, P.D., Lamb, D., Yates, D., Brown, S., 2006. Rehabilitating degraded forest land in central Vietnam with mixed native species plantings. Forest Ecol. Manage. 233, 358–365.

Nelson, D.W., Sommers, L.E., 1982. Total carbon, organic carbon and organic matter. In: Page, A.L. (Ed.), Methods of Soil Analysis, 2. American Society of Agronomy, Soil Science Society of America, Madison, WI, pp. 539–579.

Nichols, J.D., Carpenter, F.L., 2006. Interplanting *Inga edulis* yields nitrogen benefits to *Terminalia amazonia*. Forest Ecol. Manage. 233, 344–351.

Norisada, M., et al., 2005. *Acacia mangium*, a nurse tree candidate for reforestation on degraded sandy soils in the Malay Peninsula. For. Sci. 51, 498–510 (10 authors).

Olsen, S.R., Sommers, L.E., 1982. Phosphorus. In: Page, A.L. (Ed.), Methods of Soil Analysis, 2. American Society of Agronomy, Soil Science Society of America, Madison, WI, pp. 403–430.

Parrotta, J.A., Turnbull, J.W., Jones, N., 1997. Introduction—catalyzing native forest regeneration on degraded tropical lands. Forest Ecol. Manage. 99, 1–7.

Peng, S.L., Liu, J., Lu, H.F., 2005. Characteristics and role of *Acacia auriculiformis* on vegetation restoration in lower subtropics of China. J. Trop. For. Sci. 17, 508–525.

Pugnaire, F., Lázaro, R., 2000. Seed bank and understory species composition in a semi-arid environment: the effect of shrub age and rainfall. Ann. Bot. 86, 807–813.

Pugnaire, F., Armas, C., Valladares, F., 2004. Soil as a mediator in plant–plant interactions in a semi-arid community. J. Veg. Sci. 15, 85–92.

Pugnaire, F., Haase, P., Incoll, L.D., Clark, S.C., 1996. Response of the tussock grass *Stipa tenacissima* to watering in a semi-arid environment. Funct. Ecol. 10, 265–274.

Rehman, S., Loescher, R.N., Harris, P.J., 1999. Dormancy breaking and germination of *Acacia salicina* Lindl. seeds. Seed Sci. Technol. 27, 553–557.

Reubens, B., 2010. Woody vegetation for Gully rehabilitation in Northern Ethiopia. Species suitability, root structure and seedling establishment, growth and management. Thesis. Katholieke Universiteit Leuven, 254 pp.

Reubens, B., Moeremans, C., Poesen, J., Nyssen, J., Tewoldeberhan, S., Franzel, S., Deckers, J., Orwa, C., Muys, B., 2011. Tree species selection for land rehabilitation in Ethiopia: from fragmented knowledge to an integrated multi-criteria decision approach. Agrofor. Syst. 82, 303–330.

Richardson, D.M., Van Wilgen, B.W., Nunez, M.A., 2008. Alien conifer invasions in South America: short fuse burning? Biol. Invas. 10, 573–577.

Sanchez, P.A., 1987. Soil productivity and sustainability in agroforestry systems. In: Steppler, H.A., Nair, P.K.R. (Eds.), Agroforestry: A Decade of Development. ICRAF, Nairobi, Kenya, pp. 205–223.

Sanjerehei, M.M., Jafari, M., Mataji, A., Meybodi, N.B., Bihamta, M.R., 2011. Facilitative and competitive interactions between plant species (an example from Nodushan rangelands, Iran). Flora 206, 631–637.

Stevens, P.R., Walker, T.W., 1970. The chronosequence concept and soil formation. Quart. Rev. Biol. 45, 333–350.

Su, Y.Z., Zhao, H.L., 2003. Soil properties and plant species in an age sequence of *Caragana microphylla* plantations in the Horqin Sandy Land, north China. Ecol. Eng. 20, 223–235.

Tiedeman, J.A., Johnson, D.E., 1992. *Acacia cyanophylla* for forage and fuelwood in North Africa. Agrofor. Syst. 17, 169–180.

Vitousek, P.M., 1990. Biological invasions and ecosystem processes: towards an integration of population biology and ecosystem studies. Oikos 57, 7–13.

Wezel, A., Rajot, J.L., Herbrig, C., 2000. Influence of shrubs on soil characteristics and their function in Sahelian agro-ecosystems in semi-arid Niger. J. Arid Environ. 44, 383–398.
Whitford, W.G., 2002. Ecology of Desert Systems. Academic Press, London.
Wilson, J.R.U., et al., 2011. Risk assessment, eradication, and biological control: global efforts to limit Australian acacia invasions. Divers. Distrib. 17, 1030–1046 (22 authors).
Yang, L., Liu, N., Ren, H., Wang, J., 2009. Facilitation by two exotic Acacia: *Acacia auriculiformis* and *Acacia mangium* as nurse plants in South China. Forest Ecol. Manage. 257, 1786–1793.
Yelenik, S.G., Stock, W.D., Richardson, D.M., 2004. Ecosystem level impacts of invasive *Acacia saligna* in the South African fynbos. Restor. Ecol. 12, 44–51.