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Woody plant species richness, composition and structure in urban sacred sites, Grahamstown, South Africa

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Abstract Sacred sites are important not only for their traditional, spiritual or religious significance, but may also potentially be valuable for biodiversity conservation in human transformed landscapes. Yet, there has been little consideration of sacred sites in urban areas in this respect. Consequently, to better understand the ecosystem service and conservation value of urban sacred sites, inventories of their floral communities are needed. We examined the richness, composition and structure of the trees and shrubs in 35 urban churchyards and cemeteries in the City of Saints (Grahamstown). The combined area of urban sacred sites (38.7 ha) represented 2.2% of the city area and 13.6% of the public green space area. Species richness of woody plants was high, albeit dominated by non-native species. Levels of similarity among sites were low, indicating the effects of individual management regimens. There was no relationship between age of the site and measured attributes of the vegetation, nor were there any significant differences in vegetation among different religious denominations. However, the basal area and number of woody plants was significantly related to site size. These results indicate the significant heterogeneity of urban sacred sites as green spaces within the urban matrix. The significance of this heterogeneity in providing ecosystem services to users of sacred sites and the broader urban communities requires further investigation.

Keywords Age · Basal area · Cemeteries · Churchyards · Religious site · Size

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Introduction

The multiple functions and values of trees and green spaces in towns and cities are being increasingly recognised by urban ecologists and planners (Gómez-Baggethun and Barton 2013; Andersson et al. 2014; Livesley et al. 2016). Consequently, visually greener cities and suburbs are being developed, with trees and vegetation added to multiple public spaces such as parks, corridors, sidewalks and roadsides and around public service centres such as hospitals, libraries and schools. Valuation studies have indicated positive benefit to cost ratios from the planting of trees in public spaces (Tyrväinen 2001; Roy et al. 2012). However, many of the benefits are not directly monetised, meaning that some city officials view urban greening as an unnecessary luxury, or not of sufficient value to compete with other development needs (Gwedla and Shackleton 2015).

In justifying the importance and value of urban trees and green spaces, most work has focused on provisioning and regulating services, along with some on recreational values (Roy et al. 2012). However, cultural ecosystem services encompass more than just aesthetic or recreational benefits, and include a wide range of poorly studied dimensions such as educational, heritage and spiritual values (Andersson 2006). The latter have been examined in many rural areas of the world, and the importance of cultural and sacred forests and sites in the conservation of biodiversity and cultural beliefs, rites and traditions has been emphasised (Cocks and Wiersum 2003; Deil et al. 2005; Anthwal et al. 2010; Ormsby and Bhagwat 2010; Gokhale and Pala 2011; Ormsby 2013). Frequently, the species richness of plants is higher in rural sacred sites than in the surrounding matrix vegetation, whilst also providing refuge habitats for vertebrates and invertebrates (Byers et al. 2001; Deil et al. 2005). For example, 4 % of the total plant species found in Meghalaya are confined to sacred groves (Khan et al. 1997, 2008). This is also the case in Morocco, where certain forest types exist only in sacred sites (Frosch 2010).

In contrast to rural areas, understanding of the prevalence, nature, ecology and benefits of spiritual or sacred sites in urban areas is still in its infancy (Ishii et al. 2010). Green sacred sites are the formal or informal gardens, spaces, lands and trees associated with church buildings, or any place of worship (Cooper 1995), such as Christian churches, Muslim mosques or Hindu temples. Sacred sites such as churchyards and cemeteries are numerous, are found scattered throughout most urban settings, and may be managed by a number of different stakeholders (Cooper 2012). Often they contain a wide variety of plant species and even endemics (Laske 1994; Cooper 1995; Prober 1996). Indeed, urban sacred sites may be of significance to horticulturalists because they are less disturbed and often old, and thus may contain old-fashioned cultivar material (McBarron et al. 1988; Betz and Lamp 1992). Cemeteries harbour some of the oldest and largest trees within a region, thereby providing valuable habitat for birds (Lussenhop 1977) and acting as corridors for the dispersal of native species (Barrett and Barrett 2001).

Other than the people who maintain such sites, there are millions of people worldwide who regularly visit churchyards and cemeteries to tend to graves or to worship, as well as those who may only view these areas in passing yet benefit from their relative tranquillity and aesthetics (Cooper 1995). In recent times, mainstream religions have demonstrated an increased interest in environmental matters aiding in the protection of sacred sites (Awoyemi et al. 2012; Bhagwat et al. 2011; Dudley et al. 2009). It is possible that the different spiritual beliefs and practices can affect the structure, composition and functioning of such sites (Anderson et al. 2005).

Within sacred sites, trees may be worshiped according to formal belief systems. For example, sacred trees in Iran are related to different faiths and beliefs, thereby providing spiritual enlightenment to those who view them (Khaneghah 1998). The worship of trees and plants has been documented as part of Indian religious practice since the hunter-gatherer era (Chandrakanth et al. 1990) and plant species figure prominently in religious practice and particular species help facilitate worship (Chandrakanth et al. 1990). The very act of planting these species is seen as an act of worship (Chandrakanth et al. 1990).

Alternatively, the trees and green spaces associated with urban sacred sites may be protected, not because they are directly worshipped but because they play an integral role in spirituality and are usually respected by all, including residents who are not religious or of an alternative denomination (Deil et al. 2005). For example, shrine/temple forests in Japan exist because both Shintoism and Buddhism have traditions of preserving vegetation in places of worship, with forests as objects of nature worship in Shinto shrines, and for aesthetic value and places of religious training in the case of Buddhist temples (Ishii et al. 2010). In the Maghreb region Moroccan Muslim societies are based on the appreciation of the spiritual authority of patron saints (Marabout or Marabut) expressed in collective pilgrimages, the moussem, to the saint's tombs, which are shadowed by trees (Deil et al. 2005). The trees are not cut or damaged as they provide shade to the deceased saint.

Given the dearth of understanding of the nature and biodiversity of urban sacred sites as part of urban green infrastructure, as opposed to rural ones, we sought to determine the abundance and composition of woody plants of urban sacred sites in Grahamstown, South Africa. The objective was to determine the nature and composition of woody vegetation within the urban sacred sites and whether these were related to site attributes. Key questions were (i) what is the abundance and composition of woody plants in urban sacred sites and (ii) how do the abundance and composition differ in relation to site factors such as age, size, religious denomination and soils?

Study area

Grahamstown (33°18′S; 26°32′E) is located 60 km inland between the two major cities of Port Elizabeth and East London, in the Eastern Cape province, South Africa. It has a population of approximately 70,000 (Integrated Development Plan 2011). Temperatures range between 0 °C and 40 °C, with a daytime average of 22.6^0 C. The hottest months are December to March, while the coldest months occur during winter (June–August). It receives, on average, 670 mm of rainfall annually (McConnachie et al. 2008), with bimodal peaks in October–November and again in March–April. The general altitude is 570 m.a.s.l.

Grahamstown is situated in the Sub-Tropical Thicket biome, specifically grassland thicket or xeric succulent thicket (McConnachie et al. 2008). South Coast Renosterveld can also be found in patches surrounding Grahamstown, along with small patches of Afromontane forest, Grassland and Nama Karoo. Grahamstown is situated in the eastern part of the Cape Fold Belt and is underlain by folded rocks of the Cape and Karoo supergroups (Jacob et al. 2004).

A large proportion of the population (42%) has no or only primary education (Integrated Development Plan 2011). Consequently, unemployment is high at 34% and approximately one-quarter of households subsist below the national poverty line. Grahamstown has many historical and contemporary churches, earning itself the nickname, "the City of Saints". Religious denominations included Anglican, Catholic, Apostolic, Methodist, Presbyterian, other Christian denominations and Eastern religions (Muslim and Hindu). Because of the high number of churches, it made an ideal setting for examination of the key questions posed in this study. Although the churchyards are private property, most of the gardens are open to members of the public and as such can be regarded as public green space.

Methods

Sacred site identification and woody plant inventory

Thirty churches and five cemeteries in Grahamstown were identified using Google Earth as well as obtaining information from the Grahamstown Historical Society. For each one the church official was contacted to obtain permission for the study. The perimeter of each site was measured and the area of green space calculated without any buildings or non-green areas (such as paved areas). The date that the site was established was obtained from interviews with parish elders. A full inventory of shrubs and trees taller than 1.5 m was done. Nomenclature followed Germishuizen and Meyer (2003). Perimeter hedges were not included in this inventory, however if a hedge occurred within the grounds it was recorded. The basal diameter (at approximately 0.3 m) was measured for each stem of all shrubs and trees using digital callipers or a diameter tape.

Each garden was then divided into four approximately equal-sized subsampling areas (which ranged between 0.02 ha to 0.15 ha in churchyards and approximately 1 ha to 3.5 ha in cemeteries). In any vegetated area (lawn, flower bed, tree or shrub area) closest to the centre of each of the four subsamples a soil sample was taken to a depth of 10 cm. If the centre point was under a tree or shrub the soil sample was taken midway between the stem and the canopy edge. The four soil samples were pooled and sent to a commercial laboratory (Bemlab, Somerset West) for analysis, including percentage organic matter, clay, silt, sand, stone, nitrogen, pH, electrical conductivity and cation exchange capacity. Texture analysis was via hydrometer and total nitrogen was determined by complete combustion using a Eurovector Euro EA Elemental Analyser. Soil pH was measured in 1:5 soil-KCl extract. Organic matter was determined using a modified Walkley-Black method (Chan et al. 2001). Cations were extracted in a 1:10 ammonium acetate solution using the centrifuge procedure (Thomas 1982), filtered and analysed by atomic absorption spectrometry. Electrical conductivity was measured in a 1:5 soil: deionised water solution using a conductivity probe.

In each of the four subsamples the general appearance of the garden was recorded. This included a qualitative ranking of the dominant cover (lawn, shrubs/trees, flower beds, hard surface (paving /concrete/gravel) as well as taking note of any amenities (benches, water fountains, pathways, etc.) present in the garden. The total hours per week of garden maintenance and care that the site received was also documented, as reported by the parish elders.

Statistical analyses

The relationships between the church/cemetery age or area and woody plant density, total basal area, species richness, species density and total individuals were explored using regression analyses. When comparing the difference in abundance and species richness among religious denominations, a one-way ANOVA was used after checking that the data were normally distributed.

An ordination was performed using Primer 6 (Quest Research Limited, Auckland) to ascertain the degree of variation among sacred sites in terms of vegetation composition and to seek relationships between the vegetation composition and the measured site variables. The five sites that lacked woody species were omitted from the ordination, as were species that occurred in four or less of the sites. Non-metric multidimensional scaling was used to produce the ordination, while hierarchical clustering was used to produce the dendrogram.

Results

Species composition, structure and abundance

The 35 sacred sites in Grahamstown covered a total of 38.7 ha, representing 2.2% of the total area of the city. The average garden size was 1.1 ha (± 2.9 stdev) while the average woody stem density was 106.1 (± 131.5) per garden (Table 1). The mean woody plant species richness was 7.9 (± 9.9) per garden. The churches and cemeteries will be referred to by their abbreviated name in brackets (Table 1). STBAR and UNION had the greatest density of woody plants (454.5 and 519.3/ha, respectively), while five churches had no woody species at all (Table 1). The greatest total basal area was found in NEWCE and KINGS (177.4 and 40.5 m^2, respectively). However, due to the large size of many trees, the woody plant density was low at both of these sites (< 55.0/ha). The church garden with the greatest species richness was CHRCH (46). The highest number of woody plants occurred in the NEWCE and KINGS, containing 576 and 258 individuals, respectively (Table 1).

One hundred and thirty-nine different woody plant species were encountered comprising 1315 individuals. No species occurred at all of the sample sites. The most numerous species was *Cupressus sempervirens* L (Table 2), with 246 individuals, but it was encountered at only two sites (NEWCE and KINGS). This made up a total of 18.7% of the total plants sampled. The next two most numerous species were also *Cupressus*, namely *C. glabra* Sundw. and *C. macrocarpa* Hortw., with 186 and 155 individuals,

Table 1 Summary vegetation characteristics of sacred sites (30 churchyards and 5 cemeteries) in Grahamstown, South Africa

Church/Cemetery	Garden size (ha)	Total woody plants	Plant density (/ha)	Basal area (m^2) per garden	Species richness	No. invasive alien species
Churches						
African Congregational (AFRIC)	0.07	0	0	0	0	0
Apostolic Faith Mission (APOST)	0.09	0	0	0	0	0
Christ Church (CHRCH)	0.34	75	221.2	2.96	46	6
Christian Centre (CHRCN)	0.34	18	53.3	1.16	10	2
Dutch Reformed (DUTCH)	0.14	30	219.7	1.89	16	1
Ethiopian Episcopal (ETHIO)	0.04	6	169.0	1.04	4	2
Full Gospel (FULLG)	0.12	26	215.3	0.60	16	2
Hindu Mandir (HINDU)	0.51	34	66.7	1.09	12	0
Jesus Christ of Latter-day Saints (CHJCL)	0.14	52	362.6	1.68	29	1
Lesley Hewston (LESLE)	0.20	0	0	0	0	0
Living Star (LIVIN)	0.10	0	0	0	0	0
Methodist of South Africa (METHO)	0.09	3	34.4	0.74	1	2
Mosque (MOSQU)	0.04	6	142.1	0.20	4	0
New Apostolic (NEWAP)	0.09	14	164.6	0.43	5	3
Old Apostolic (OLDAP)	0.23	4	17.0	0.28	4	3
Presbyterian (PRESB)	0.08	0	0	0	0	0
Seventh Day Adventist (SEVEN)	0.17	12	69.4	1.23	7	4
Shaw Memorial (SHAWM)	0.55	21	38.4	1.15	7	0
St. Augustine (STAUG)	0.13	5	39.1	1.08	4	1
St. Bartholomew's (STBAR)	0.05	24	454.6	0.19	11	0
St. Clements's (STCLE)	0.07	11	151.0	0.86	6	1
St. Joseph's (STJOS)	0.10	3	29.0	0.03	3	0
St. Mary's (STMAR)	0.15	13	89.6	1.85	6	1
St. Patrick's (STPAT)	0.15	36	232.8	1.27	16	2
St. Peter Claver's (STPET)	0.20	4	19.8	2.08	2	1
St. Philips (STPHI)	0.03	1	32.9	0.01	1	1
Trinity Presbyterian (TRINI)	0.07	15	208.0	0.64	8	0
Twelve Stone (TWELV)	0.25	3	12.2	0.54	2	0
Union Congregation (UNION)	0.09	46	519.3	0.14	2	2
Wesley Methodist (WESLE)	0.08	4	48.7	0.36	4	0
Cemeteries						
Kingswood Cemetery (KINGS)	4.91	258	52.3	40.53	21	0
Mayfield Cemetery (MAYFI)	12.50	14	1.1	0.02	3	0
New Cemetery (NEWCE)	11.86	576	48.6	177.40	25	5
Tjanti Cemetery (TJANT)	2.19	1	0.5	1.63	1	0
Mean	1.1	37.6	106.1	6.9	7.9	1.2
Standard deviation	2.9	103.7	131.5	30.4	9.9	1.5

respectively. *Cupressus* made up 44.6% of the plants sampled, and the 48.6 of all plants recorded were from the Cupressaceae family. The top 11 most common species accounted for 65.9% of the plants enumerated. Just over half of the species (56%) were non-native, 32% were indigenous and the remainder unidentified (and therefore most likely to be introduced). Only five of the top 20 species were indigenous to South Africa. Five species are declared invasive species in South Africa, including *Ligustrum lucidum* W.T.Aiton which was the most frequent species in terms of site presence (13 sites).

Most of the stems (69%) fell into the smallest size class (stem diameter of 0–15 cm) (Fig. 1). This could be attributed to the large number of shrubs found in the study sites. Large

Table 2 The twenty most common woody plant species in 35 sacred sites in Grahamstown

Species	Designation	No. of sites located in (n = 35)	Mean no. per site (n = 35)
Cupressus sempervirens	Alien	2	7.0
Cupressus glabra	Alien	3	5.3
Cupressus macrocarpa	Alien	2	4.4
Ligustrum lucidum	Invasive	13	2.5
Thuja orientalis	Alien	7	1.5
Jacaranda mimosifolia	Alien	6	1.5
Aloe ferox	Indigenous	4	0.7
Cestrum laevigatum	Invasive	9	0.6
Schinus molle	alien	1	0.6
Acacia mearnsii	Invasive	3	0.5
Olea europaea subsp. africana	Indigenous	5	0.5
Eucalyptus camaldulensis	Invasive	4	0.4
Cussonia spicata	Indigenous	4	0.5
Euphorbia pulcherrima	Alien	1	0.4
Acacia baileyana	Alien	2	0.3
Celtis africana	Indigenous	6	0.3
Grevillea robusta	Alien	2	0.3
Schinus terebinthifolius	Invasive	5	0.3
Acacia karroo	Indigenous	2	0.3
Nandina domestica	Alien	3	0.3

individuals of 90 cm or larger diameter only made up 0.8% of all stems measured.

Influence of site age, size, denomination and care on species richness and density

A significant relationship was observed between sacred site size and total basal area ($r^2 = 0.46$, $p < 0.001$) and similarly between garden or cemetery size and total number of woody plant individuals ($r^2 = 0.44$, $p < 0.001$) (Table 3). The removal of two outliers (NEWCE and KINGS) made the latter relationship not significant ($r^2 = 0.01$, $p = 0.84$). In some cases cemetery data were omitted from the analyses to see if their large area had an effect on the overall results. However, in the case of both woody plant density ($r^2 = 0.49$, $p < 0.001$) and species richness ($r^2 = 0.50$, $p < 0.001$) it made little difference. There was no significant difference in tree density (F = 0.69, df = 6, 23, $p = 0.66$), or in species richness (F = 0.59, df = 6, 23, $p = 0.73$) among the different denominations.

CHRCH had the greatest woody plant species richness (46), which may be partially attributed to the long hours of maintenance that it receives, a great deal more than other gardens. It had a garden committee of three people, as well as a gardener that spent three days a week caring for the garden. The mean for the other gardens was 6.5 ± 5.8 h per week.

Community composition of church/cemetery gardens

There was generally a low level of similarity among the gardens as evidenced by the low degree of overlap between sites depicted in Fig. 2 and low similarity percentages in Fig. 3. Four sites displayed less than 20% level of similarity with all of the other sites (TWELV, METHO, STBAR and TJANT). The hierarchical clustering (Fig. 3) showed that the two sites with the greatest level of similarity are NEWAP and SEVEN, at 59.6%. This is followed by NEWCE and KINGS at 52.0% and between STPAT and STCLE (51.8%) (Fig. 3). There are four broad groups differentiated at approximately the 20% level of similarity. Namely, group one (NEWCE, KINGS, HINDU, CHRCH, CHJCL), two (STMAR, STJOS, CHRCN, DUTCH, FULLG), three (ETHIO, SHAWM, STPAT, STCLE, STPHI, UNION, NEWAP, SEVEN) and four (TRINI, MOSQU, WESLE, MAYFI). There were also outliers with very low similarity to any other garden, represented by METHO, TWELVE, TJANT, STPET, STAUG and

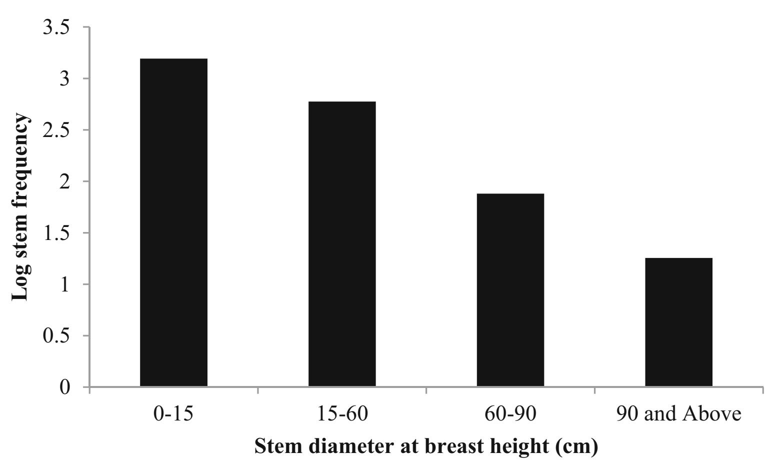

Fig. 1 Size class distribution of all woody plants (>1.5 m tall) sampled in 35 sacred sites (churchyards and cemeteries) in Grahamstown, South Africa

Table 3 Regression results between sacred site age or size and vegetation characteristics in Grahamstown, South Africa (significant results in bold)

Variables	r^2	*p* value
Church/cemetery age and:		
•Woody plant density (/ha)	0.01	0.45
•Total basal area (m^2)	0.08	0.09
•Species richness	0.08	0.09
•Species richness/area	0.03	0.30
•Total number of woody plants	0.06	0.15
Church/cemetery size and:		
•Woody plant density (/ha)	0.04	0.21
•Woody plant density (/ha) (excl. Cemetery)	0.04	0.28
•Total basal area (m^2)	**0.46**	**< 0.001**
•Species richness	0.03	0.29
•Total number of woody plants	**0.44**	**< 0.001**

STBAR. There were no significant differences among the groups with regards to garden age, woody plant density, site size and total basal area. Group one was significantly different from the other four groups in terms of species richness ($F = 10.71$, df = 4, 23, $p < 0.001$). The same was found between group one and all other groups when comparing total number of individuals ($F = 3.86$, df = 4, 23, $p = 0.01$).

The regression looking at the influence of abiotic variables on plant assemblages showed that p was optimized (at 0.22) for the seven variables of age, electrical conductivity, clay, silt, stone, nitrogen, and CEC. Yet, statistical testing of the results of this relationship between biotic and abiotic variables, indicated that the different plant assemblages had no specific associations with any of the abiotic (environmental) variables ($p = 0.20$). Thus, the abiotic variables were not useful in characterising the plant assemblages at a particular site or vice versa.

Discussion

Sacred sites are typically refugia for biodiversity in both rural and urban settings, being maintained by different cultures throughout the world (Malhotra et al. 2007; Sheridan and Nyamweru 2007). However, urban sacred sites have been poorly considered in debates on urban green spaces, cultural ecosystem services, as well as biodiversity and conservation of sacred areas. While there has been examination of church forests in Ethiopia (Wassie et al. 2005, 2009; Aerts 2007) there is limited information pertaining to the vegetation of sacred sites within the urban environment. This study sought to (i) characterize the woody plant structure and composition of sacred sites within Grahamstown, South Africa, and (ii) assess the factors that might influence the abundance and composition of woody plants in such sites.

There was a large range in vegetation abundance and composition across the sample sites, echoing work on urban green spaces globally. This is likely to be partially a result of the pervasive effects of human influence through management and introductions of non-native species (Kunick 1987; Chandrashekara and Sankar 1998; Cook et al. 2011; Nagendra and Gopal 2011). In addition to these human mediated influences are site factors such as size of the site (Ishii et al. 2010), soil characteristics (Betz and Lamp 1992) and the conditions of the site prior to major urbanization impacts

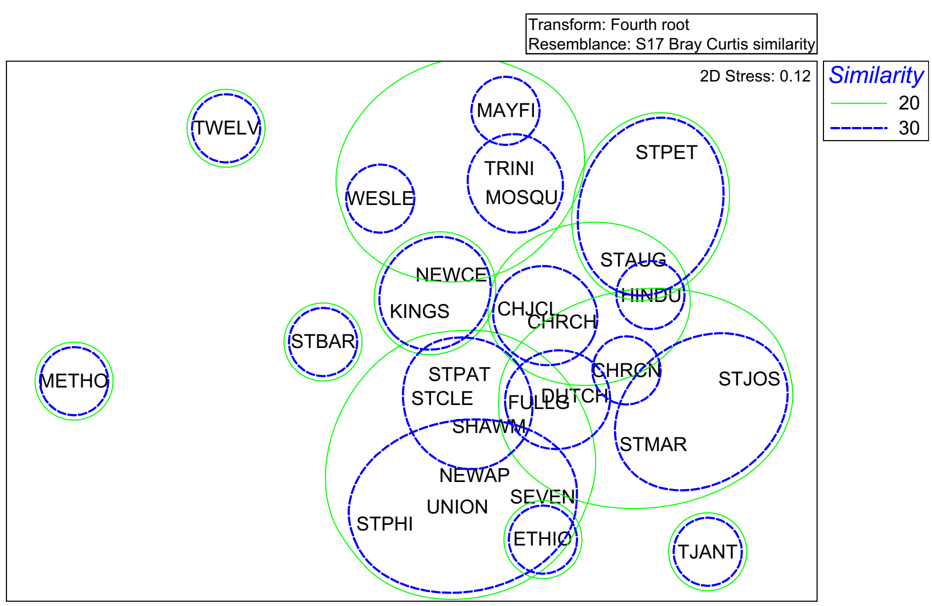

Fig. 2 Ordination plot of the 28 churchyards and cemeteries in Grahamstown indicating the degree of overlap at two levels of plant assemblage similarity (20 and 30%) (site abbreviations correspond to those in Table 1; *n* = 28; sites without gardens were excluded)

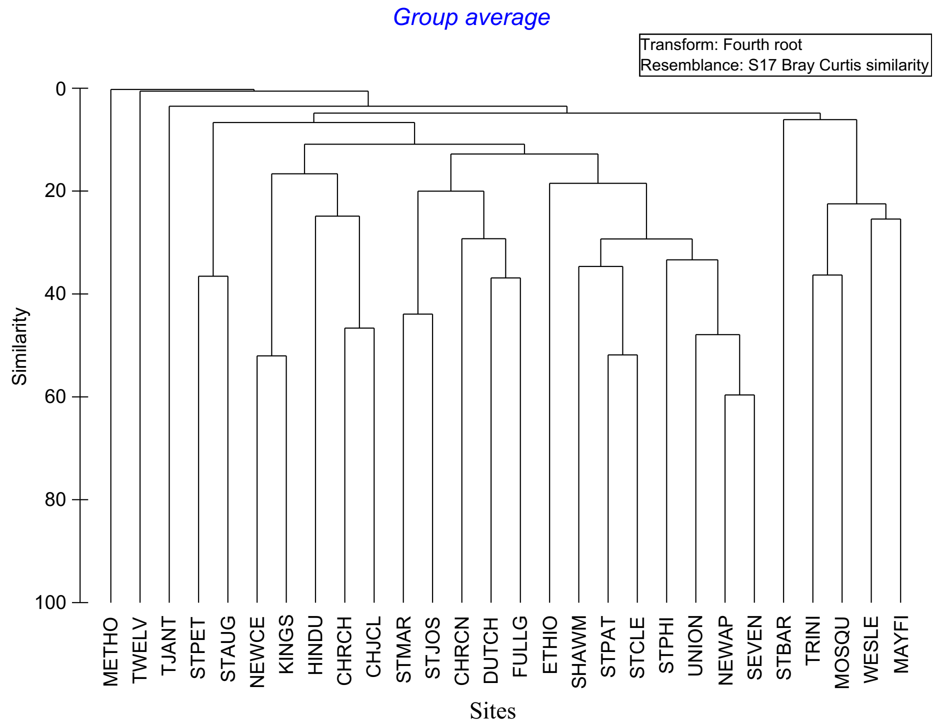

Fig. 3 Dendrogram of the plant communities in Grahamstown church gardens and cemeteries ($n = 28$; sites without gardens were excluded)

(Nowak et al. 1996), which have been shown to influence vegetation composition and abundance.

Characterization of urban sacred site vegetation

Five of the sites had no woody plant species over 1.5 m tall. All of these sites were located in Grahamstown east, a part of Grahamstown that remains impoverished due to the previous racially discriminatory apartheid regime which led to under-development and neglect of urban suburbs previously designated for black South Africans (Donaldson-Selby et al. 2007). These poorer areas have less urban public green space and higher housing density than that of the more affluent western parts of Grahamstown (McConnachie and Shackleton 2010). Street tree prevalence in Grahamstown also favours Grahamstown west, with the eastern parts containing less than 5 % of the city's street trees but over 80 % of the human population (Kuruneri-Chitepo and Shackleton 2011). The five sacred spaces lacking any woody vegetation are all, however, privately managed. The absence of gardens at these sites was attributed to some being rented properties and therefore the congregants had no influence on the appearance of the grounds, limited disposable income for the development of a church garden, and high prevalence of livestock in Grahamstown east which damaged nascent attempts at tree planting and beautification (Richardson and Shackleton 2014). The relative contributions of these require further elucidation. The potential relationship with relative poverty status corresponds to the oft invoked 'luxury effect' of wealthier householders typically having more species-rich domestic gardens (Hope et al. 2003) and with higher vegetation cover (Jenerette et al. 2007), although the two response curves may not be correlated and also vary in relation to age of the suburb (Kendal et al. 2012).

The site with the highest number of woody plant species (CHRCH) was the one that received the most maintenance (five times more than the average for the other sites) as well as being a large site. The effects of different maintenance regimes on tree species composition and vegetation structure of sacred groves of India revealed that each regime yielded different outcomes (Chandrashekara and Sankar 1998). Areas managed by individual families were more disturbed than those maintained by groups of families or statutory agencies (Chandrashekara and Sankar 1998). LaPaix and Freedman (2010) considered the effect of anthropogenic stressors on compositional and structural indicators of vegetation within urban parks of Nova Scotia, Canada. They found that historical use and edge influences were significantly related to variation in vegetation composition within semi-natural forests, while natural disturbance (a hurricane in this case) strongly influenced plant communities (LaPaix and Freedman 2010). The semi-arid climate of Grahamstown requires a great deal of management and care to have an urban garden that could support a variety of woody plant species. CHRCH's 24 h of maintenance per week would therefore likely influence the vegetation markedly.

The number and type of alien species found in cemeteries in Illinois and Indiana was dependent on the past management of the cemetery (among other things), being affected by practices such as mowing, grazing by livestock and the number of plants that had been planted for decorative purposes (Betz and Lamp 1992). The use of herbicides and clearing of vegetation can greatly reduce the rare or 'infrequent' species that may also be found in some urban sacred sites such as cemeteries (McBarron et al. 1988). The management techniques of different generations also have an effect on the species composition (Nagendra and Gopal 2011). Older parks will have experienced many different managers, and so are more likely to have a greater variety of species than newer parks (Nagendra and Gopal 2011). The preferences of current park managers were also seen to have shifted from large canopied species to smaller trees, reducing the available habitats for birds and the removal of air particulate pollutants (Nagendra and Gopal 2011). Smith et al. (2006) found that a number of factors influenced domestic garden vegetation of individual households, but that the preferences of garden owners had a much greater role in determining floral richness than did environmental variables.

The total area of churchyards and cemeteries was 38.7 ha which increases McConnachie et al. (2008) estimate of public green space in Grahamstown (246.6 ha) by more than 15%. We recorded 139 different woody plant species, 56% of which were introduced, 32% indigenous and the remainder unidentified (albeit assumed to be introduced). Thompson et al. (2004) found that in 60 domestic gardens in Birmingham, UK, 67% of the floral species were alien. Similarly, of the 61 urban, domestic gardens surveyed in Sheffield, UK, 70% of plants were alien (Smith et al. 2006). In the case of Mexico City, 750 plant species were sampled and 70% were alien (Smith et al. 2006). In parks of Bangalore, India, of the 80 tree species that Nagendra and Gopal (2011) encountered 66% were introduced and 34% were native. These high ratios in favour of alien species further emphasise the significant role of human actions in determining the species composition of urban green spaces, including sacred sites. Indeed, ornamental plants comprise more than 40% of widespread invasive plant species globally, exceeding any other reason for plants being introduced (Smith et al. 2006).

The three most abundant species comprised almost half of the total sample population (45%), while the top 11 species made up 66%. This was similar to findings in India where the top five tree species in parks in Bangalore made up close to half the population (Nagendra and Gopal 2011). No species occurred at all of the sample sites of our study, something that was similar in the cemeteries of Campbelltown, where only one species was recorded at every site (McBarron et al. 1988). The four sacred groves examined in the Pondicherry region also had only two species occurring in each grove (Ramanujam and Cyril 2003). The domestic gardens of Sheffield also showed this phenomenon, with only 2.7% of the species occurring in more than half of the gardens (Smith et al. 2006).

Although no control sites were designated during this work, comparison to the findings of McConnachie et al. (2008), who reported on the woody vegetation attributes of 128 formal and informal public green spaces in Grahamstown (but did not include sacred sites), is instructive. Firstly, the mean size of sacred sites is a lot smaller (1.1 ha) than the public green spaces (1.9 ha). Secondly, they contain almost five times higher densities of woody plants (106/ha) than the public green spaces (23/ha), thus they are much more vegetated. The mean species richness per site was similar (7.9 and 7.7 for sacred and non-scared sites, respectively), but given the smaller mean size of the sacred sites, then woody plant species density was approximately double in sacred sites relative to non-sacred ones. Lastly, the proportion of woody plants that were indigenous was relatively similar, being 56% in the sacred sites and 51% in the public green spaces. Thus, overall, the sacred sites are smaller, more vegetated and richer in woody species than other public green spaces in the city. The higher species richness and density are likely to underpin a greater range of ecosystem services (Gokhale and Pala 2011) and emphasises the importance of urban sacred sites for ecosystem service benefits for residents and neighbouring landscapes (Bhagwat 2009). It is likely that the higher woody plant density and species richness will echo natural systems in supporting a greater diversity of vertebrate and invertebrate fauna, which has also been shown in some urban systems (e.g. Goertzen and Suhling 2013; Ferenc et al. 2014; Shackleton 2016), but this needs to be assessed for sacred sites.

Millward and Sabir (2010) suggest that to promote biodiversity and resilience 40% of the urban tree population should be in a size class with a diameter at breast height range of 0–15 cm, 30% 15–60 cm, 25% 60–90 cm and 5 % ≥ 90 cm. The majority (69%) of stems recorded in this study occurred in the smallest diameter size class. This occurred because many of the woody plants encountered were shrubs. The proposed classes would allow managers to allocate annual maintenance costs uniformly over many years and reduce establishment-related mortality (Millward and Sabir 2010). Diversity of the urban forest is important to reduce risk from pests and diseases, climate change, as well as supply a variety of ecosystem services (Kendal et al. 2014). This has led to the widespread acknowledgement of the 10/20/30 rule of Santamour (1990), which suggests that urban forests should not comprise of more than 10 % of any one species, 20% of any genus, and 30% of any particular family. The most common species (*Cupressus sempervirens*, 18.7%), genus (*Cupressus*, 44.6%), and family (*Cupressaceae*, 48%) in this study surpassed the suggested ratios. Kendal et al. (2014) also suggest generally these ratios are much higher at species level, but more comparable at the genus or family scale, however this is once again not the case in this study. This may have occurred because of the large majority of these specimens being located in the large cemeteries of Grahamstown.

Hierarchical clustering revealed that there was a low level of similarity across the sample sites, with the highest being 60% (NEWAP and SEVEN), while there was a 52% similarity between NEWCE and KINGS, which could be explained by the abundance of conifers at the latter two sites. The low levels of similarity found throughout the sample sites was not surprising, considering that no species occurred at all of the sites. Just as the sacred areas of the Maghreb countries are influenced by human uses (Deil et al. 2005), so too are sacred sites of Grahamstown, suiting the needs and preferences of the congregation. Such low levels indicate that, based on woody plants alone, it is not possible to construct a robust typology of urban sacred sites which could promote management approaches or strategies to integrate them into wider green space initiatives or plans. On the one hand it indicates that each site is relatively unique, which enhances both species and beta diversity at the city scale. On the other hand much of the diversity is based on alien species. Precisely how urban residents and users of urban sacred sites respond to this requires elucidation (De Lacy 2014).

Relationships between vegetation and site attributes

Although we found no significant relationship between age of the site and species composition, some studies have found otherwise. Age could influence the effect of historical uses over time (through management regimes) and therefore have an influence on variation in vegetation composition within semi-natural forests of Nova Scotia (LaPaix and Freedman 2010). Salick et al. (2007) found that due to the restrictions placed on timber extractions in Tibetan sacred sites, they have significantly larger trees and greater tree cover than non-sacred sites, making them effective in protecting old growth trees. Age had no significant effect on neighbourhood species in Halifax, Nova Scotia (Turner et al. 2005). In contrast, Nagendra and Gopal (2011) reported that older parks (those established prior to 1970) had fewer trees than younger parks, however, the trees in older parks were significantly larger than those in newer parks. There was a significantly greater size class diversity with increasing site age, as found by Nagendra and Gopal (2011).

In our study there were no significant relationships between the size of the urban sacred site and woody plant density or species richness. This finding contrasts with Smith et al. (2006) who found that domestic garden area significantly influenced plant species richness. This was, however, largely due to the increase in the size of lawns when gardens increased in size, which is positively related to richness of lawns (Smith et al. 2006). It was also found that the species density of gardens decreased with garden size (Smith et al. 2006). In sacred shrine/temple forests of Japan species richness was found to decrease with decreasing area (Ishii et al. 2010). It was also noted that smaller forests still play a vital role in conserving species diversity in urban environments as they often store rare and infrequent species (Ishii et al. 2010). One of the smallest sample sites, STBAR, stored a large variety of species as well as a high density of woody plants, showing that small sites do have the potential to store a great deal of plant diversity. Angold et al. (2006) found a positive relationship between site area and species richness. We speculate that the absence of a relationship in our data could be because many of the smaller gardens were well looked after and had high values for most of the garden attributes (for example STBAR and TRINI), while some of the larger sites had low woody plant species richness because of the large expanse of lawns (SHAWM), or areas reserved for parking (OLDAP), although not paved for such.

There was a significant, positive relationship between site size and total basal area and total number of individuals. This conforms with studies in natural and urban ecosystems (e.g. Echeverria et al. 2007). However, since urban sacred sites are not formally part of the municipal green space system, it is unlikely that planning or zoning processes would heed any suggestions to increase the size of such sites for biodiversity purposes. The most important ecosystem services from urban sacred sites are cultural and spiritual ones, and any biodiversity or conservation benefits occur more by chance as a result of optimising the cultural and spiritual benefits. The relationship between the appearance or type and abundance of vegetation of such sites and the spiritual satisfaction they provide requires investigation (De Lacy, 2014).

There was no significant effect of religious denomination on woody plant density or species richness. This differs from Anderson et al. (2005) who found that religious beliefs affected the ecology of sacred sites and that sacred sites differed in useful species and endemic species composition when compared to non-sacred sites. In Japan Shinto shrines are seen as the object of nature worship, receiving little management and human access is discouraged (Ishii et al. 2010). In contrast, Buddhist temples are intensely managed and are often used for places of religious training (Ishii et al. 2010). Consequently, temples have a smaller area of forest cover than do shrines (Ishii et al. 2010).

There was also no clear relationship between species composition and the measured soil attributes. This is contrary to what was found in cemeteries of the US. Betz and Lamp (1992) reported that cemeteries that lacked evidence of ever having been ploughed or had soils with a shallow A-horizon had different species composition and relative abundance than other cemeteries (Betz and Lamp 1992). The differences in species composition of the groves of the Pondicherry region could also be attributed to soil characteristics (Ramanujam and Cyril 2003). The absence of any detectable influence of soil characteristics on site-scale plant communities resulted in the inability to distinguish what separated the four groups of sites that formed at the 20% level of similarity.

Conclusion

This study showed that urban sacred sites had a wide variety of woody plant species composition and abundance, with more than half of the species being exotic. This variation is likely to be largely due to the influence of site maintenance actions and species selection. Factors such as site age, area and denomination had no significant influence on vegetation characteristics, indicating that urban sacred sites show characteristics more similar to urban domestic gardens than that of rural sacred groves or sacred groves of eastern countries or religions. Environmental characteristics had limited influence on the plant assemblages throughout the sample sites, with sites showing very little similarity. The environmental relationships were overshadowed by human influence through maintenance actions and species selection by congregants. Nonetheless, this study has shown that urban sacred sites do harbour a variety of woody vegetation in terms of both species and sizes. Nodal sites such as these therefore contribute to providing ecosystem services in urban settings, especially cultural ecosystem services which have been poorly examined to date, but the significance of the provisioning and regulating services they provide also require consideration.

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