Research Report

# Trees outside woods

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WOODLAND TRUST

# TREES OUTSIDE WOODS

## A report to the Woodland Trust

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# Introduction

Woodland cover (woods ≥ 2.0 ha) in England has been measured by the Forestry Commission (2001) as being 1.02 million ha. However, many trees do not grow in woodland, but are found singly, in hedgerows and in parks and gardens, or as small groves in an agricultural landscape. These trees form a forgotten resource that has never been accurately counted and is often undervalued. In this report we review published information on trees outside woods (TOWs), their quantification and their role in the landscape.

We present a summary of findings from an analysis of the TOWs resource carried out in three areas of lowland England.

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WTPL/Fiona Grainger

## The Definition of TOWs

A major problem that has bedevilled the inventory of TOWs has been differences between studies in the definition of what they are. Results vary considerably depending on the minimum size of trees that are included and the minimum size of woods that are excluded. For example, estimates of the number of non-woodland trees in England made by the Forestry Commission in 1951 counted trees in woods of less than one acre (0.4 ha) (1953). The 1980 census, however, used a minimum of 0.25 ha (Forestry Commission, 1983) making direct comparison of the results difficult.

If we want to compare the TOWS resource in different areas or to monitor changes over time then we need consistency in the ways that they are measured. Logically, trees outside woods are those growing in places not defined as woods and forest, but even the definition of woodland is not straight forward. Three criteria appear to be important: woodland area, tree cover and tree height.

The Food and Agriculture Organisation of the United Nations defines forest, in its global forest resource assessment, as "land spanning more than 0.5 hectares with trees higher than 5 metres and a canopy cover of more than 10 percent, or trees able to reach these thresholds *in situ*" (FAO, 2004). This definition sets a seemingly useful threshold area – 0.5 ha – above which a group of trees is defined as woodland. However, it fails to recognize that many TOWs grow along roads, rivers and railways as continuous lines of canopy, often no more than one or two trees wide. These sinuous tree belts are clearly not woodland since they lack core woodland habitat with its characteristic microclimate, but may often be greater than 0.5 ha in total extent. In order to overcome this problem Paletto *et al.* (2006), in an inventory of TOWs in Italy, classified non-woodland trees into two groups – copses of between 0.05 and 0.5 ha that had to be at least 20 m wide, and linear woodland with a width between 3 and 20 m and at least 20 m long. The Forestry Commission surveys of non-woodland trees (1983, 2001) used a maximum group size of 0.25 ha and defined linear woods as being less than 20 m wide and at least 25 m long.

Tree group area can be strongly influenced by subjective decisions about when a break in the canopy is sufficiently wide to divide a group in two. When canopy cover is delineated from aerial photographs it is often very difficult to draw a precise boundary between forest and non-forest (Kleinn, 2000). The definition of woodland area is particularly difficult in areas with a sparse canopy such as wood-pasture or where trees are colonising heathland. Bunnell (1999) argued that there is little evidence that many forest dwelling organisms see the division between woodland and open habitat as distinctly as it is often portrayed in a GIS. With this in mind, our analysis of TOWs should perhaps compare core woodland habitat with transitional or edge habitat rather than place undue emphasis on woodland patch area.

In this analysis, therefore, we have delineated only those parts of the landscape that are actually covered by tree canopies, down to the level of single trees. We have aimed to be comprehensive – including woods of all sizes as well as TOWs. We have only grouped trees as a single object when their canopies are contiguous. Kleinn (2000) proposes that different methods of assessing the TOWs resource may be necessary for different purposes. Our intention has been to create a multi-purpose GIS database of tree cover which can be interrogated in a variety of ways depending on the research needs.

## Effective sampling methods

The growing awareness of the importance of landscape heterogeneity in maintaining biodiversity has focussed attention on mapping and describing fine-scale patterning. This presents a number of methodological challenges – firstly in the detection of fine-scale patterns and secondly in its sampling. These are key problems in mapping TOWs – which are, by definition, tiny but often numerous elements in the landscape, that are distributed very unevenly.

The automatic classification of large, remotely sensed scenes allows the problem of sampling to be overcome. Once an adequate classification algorithm has been developed it can be applied to very large samples. However, until comparatively recently the coarse resolution of satellite data meant that they could not be used for the precise mapping of fine detail. Where satellite-borne sensors record data at a pixel size greater than the minimum patch size used in a study, their interpretation is largely based on those spectral properties. Vegetation abundance can be extracted from measures of radiance in one of two ways:

- ❑ Vegetation indices such as the Normalised Difference Vegetation Index (NDVI) and the MODIS (MODerate Resolution Imaging Spectroradiometer) Vegetation Continuous Fields (VCF) (Hansen et al., 2003) use the ratio of radiance in specific wavebands to predict average tree cover values.
- ❑ Spectral Mixture Analysis (SMA) is a technique in which the known reflectance values of likely land covers surfaces are used to find the 'best explanation' for the combined reflectance recorded in a pixel that encompasses several land cover types.

Perry *et al.* (2009) compared estimates of forestland area in the Midwest made during the annual USDA Forest Service Forest Inventory and Analysis to estimates of total tree cover derived from MODIS Vegetative Continuous Fields (VCF). VCF data have a spatial resolution of only 500 m. This study found that estimates of percent tree cover had "a significant and unknown level of uncertainty", particularly where tree cover was low. They proved to be inadequate for TOWs inventory.

SMA has been used with some success to monitor tree cover in urban areas using Landsat ETM data (Small and Lu, 2006) and in semi-arid landscapes (Elmore et al., 2000). SMA produces an estimate of the proportion of a pixel covered by green vegetation and whilst it is a useful method where trees dominate vegetation, is less likely to be useful where scattered TOWs are mixed with other vegetation types.

High-resolution aerial and satellite images are ideal for identifying individual objects, but require either manual photo-interpretation or the development of complex automated object and spectral recognition techniques.

Aerial photographs generally are of sufficient resolution to enable individual tree crowns to be identified and mapped. The Forestry Commission used stereoscopic pairs of images to identify trees for the 1979-1982 survey, and this is likely to have made the job of identifying trees on black and white photographs very much easier. For this report we used black and white photographs taken in 1948 and found great difficultly in identifying trees on those images taken during the leafless times of year.

High resolution, orthorectified and geolocated colour images of the whole of the UK, no more than five years old, can be readily obtained on-line via Google Earth (http://earth.google.co.uk/). Tree crowns can be readily identified by eye on these images, but manual digitisation over large areas is very laborious and may also require time-consuming ground-truthing if species data are required (Nemitz et al., 2007). Our experience during this study was that the time taken to digitise all tree canopies in a 40 $km^2$ sample plot varied from approximately an hour in a sparsely wooded landscape, to over four hours in those areas with large numbers of scattered trees. Akbari *et al.* (2003) compared several methods of characterising land use, including trees, in urban areas in Sacramento, California and concluded that colour digital orthorectified aerial photos were the best medium.

There are a number of sources of high-resolution satellites images, including IKONOS, QuickBird, SPOT 5, OrbView and EROS (Fritz, 1999 ), however, they are presently only commercially available to order. They can be prohibitively expensive for low-budget survey purposes and coverage, in space and time, is not comprehensive.

Spectral recognition of tree cover from high-resolution images has often proved unsatisfactory (e.g. Goetz et al., 2003, Mathieu et al., 2007). This is largely because of problems of misclassification are accentuated by the fine detail contained within these images. Tree shadows, gaps between trees and a variety of surfaces beneath trees, particularly in urban areas, cause statistical confusion. One method for circumventing such problems is to use course resolution images (such as Landsat ETM) to map large areas of continuous tree cover where detail is not required and fine resolution images to add individual trees and small copses. Levin *et al.* (2009) used a combination of Landsat and SPOT5 images to map paddock trees in New South Wales, Australia. They compared their results with an object recognition method utilising the shadows cast by trees on bare ground. Spectral recognition proved to be more accurate than object recognition – correctly identifying over 90% of all paddock trees.

The remote sensing of tree cover is a research priority for both conservation and carbon monitoring purposes and it is likely that there will be rapid developments in this area over the next few years. However, at present, although it is very time-consuming, manual digitisation of aerial photographs is the most reliable method of mapping TOWs.

The distribution of TOWs is highly spatially variable and therefore a critical issue in designing effective means of estimating the size of the resource is the sampling framework. Both bigger samples and greater replication will give more precise estimates of mean values but in view of how laborious it is to digitise individual tree canopies there is, inevitably, a trade-off between sample size and replication. The Forestry Commission used rectangular plots measuring 250 m by 3,000 m (total 75 ha) to sample non-woodland trees in their 1979-82 survey. It was found easier to obtain the required aerial photographs for long thin strips. Twenty plots were measured per county. Aerial photo interpretation was augmented by field visits both to ground-truth results and to ascertain the species and condition of trees.

# Ecological role of TOWs

## Habitat value

The habitat value of TOWs depends critically on their species, landscape context and group size. They are likely to be a valuable resource for many species in those areas with low woodland cover, such as regions of intensive agriculture and urbanisation. Here they can provide habitat refuges that support species populations in an otherwise hostile environment (Fischer and Lindenmayer, 2002a). In well-wooded landscapes they act as corridors and stepping stones that increase the permeability of the landscape and contribute to the total area of edge or transitional woodland habitat.

There is ample evidence in the research literature that isolated trees and copses enhance biodiversity above those levels found in the surrounding matrix. This may be because they are relics of a former richer and more widespread ecosystem that has now disappeared from the majority of the landscape or because they are habitat 'honey pots' that attract species because they provide resources not available elsewhere in the landscape.

### TOWs as relics of old-growth woodland

Isolated ancient trees are often relics of former old-growth woodland – typically left as boundary markers or as pasture woodland. Ancient trees have often survived outside woodland, maintained by pollarding and shredding, whilst their woodland counterparts have disappeared. These relics of

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former ancient woodland may constitute a distinct and important genetic resource which no longer survives elsewhere. In addition to their own intrinsic importance they may harbour a particularly rich and distinctive community of wood-decay invertebrates, fungi lichens and epiphytes (Butler et al., 2001). A high proportion of rare and threatened dead wood species are now associated with such ancient trees in the UK. The nature conservation importance of pasture-woodland systems has been reviewed by Kirby *et al.* (1995). They emphasize the value of this habitat type for lichen, invertebrate, bird, and bat populations and probably fungi.

Conservation biology theory tells us that the larger a nature reserve, the more species it is likely to contain. This implies that isolated trees and small copses should not be a high priority for the conservation of woodland species. However, Götmark and Thorell (2003) found that veteran trees and their associated dead wood habitat were more abundant in tiny patches of woodland in southern Sweden than in large forest areas. They attributed this to the fact that many tiny woodland reserves were often protected because they were biological rich remnants of former ancient woodland whilst large forest reserves were typically more degraded and were protected for other reasons, including recreation. This suggests that in old, highly fragmented landscapes such as those found in Britain and southern Sweden, TOWs may merit being made a conservation priority because of the quality of the resource that they protect.

## TOWs as honey pots for biodiversity

Although there are comparatively few studies of the biodiversity associated with TOWs in a UK context, there are many studies globally that highlight their importance. TOWs have been found to be hotspots for diversity of oribatid mites in Scotland (Brooker et al., 2008), ant and beetle abundance and richness in Ghana (Dunn, 2000), flying arthropods in Switzerland (Grüebler et al., 2008), beetle diversity in Japan (Ohsawa, 2007), frugivorous birds in Kenya (Eshiamwata et al., 2006), birds (Fischer and Lindenmayer, 2002b) and bats in southeastern Australia (Lumsden and Bennett,

2005) and understory vegetation in Mexico (Otero-Arnaiz et al., 1999). Whilst the communities associated with TOWs may be qualitatively different from those found in continuous woodland (Ozanne et al., 2000) they often contain a significant component of woodland specialist species.

TOWs enhance diversity by providing resources not available in the wider landscape. These include structural diversity, shelter, a more equable microclimate, cover, and specific niche requirements, including food and breeding sites. By attracting keystone species, TOWs enhance diversity in other guilds. For example isolated trees in agricultural landscapes are known to attract birds – as cover, stepping stones between habitat fragements and for the resources that they can provide. This can lead to increased seed rain, brought by frugivorous birds and deposited when they defecate. This, in turn, leads to increased rates of seedling establishment beneath their canopy and makes TOWs foci for the natural recolonisation of deforested landscapes. This effect is well-documented from tropical landscapes (Toh et al., 1999, Eshiamwata et al., 2006, Fischer and Lindenmayer, 2002a, Guevara et al., 1986, Otero-Arnaiz et al., 1999).

Both the size and species of tree can be important in determining the quality of the habitat provided. A study of ants and other arthropods found on the bark of isolated trees in a Japanese city concluded that tree species and size were more important than tree patch size or tree cover in determining both abundance and diversity (Yasuda and Koike, 2009). Dominant native species of tree have long been known to have more insect species associated with them than those that were recently introduced (Southwood, 1961). The habitat value of new TOWs planting could be enhanced by selecting those species known to be attractive to large numbers of birds and insects.

Surveys of TOW species by the Forestry Commission (1983, , 2001) indicate that the most abundant TOWs species in England are native broadleaves (ash 10.3%, oak 7.7% and willow 7.4%). Most ash and willow trees are less than 20cm dbh whilst most oaks are greater than 50cm dbh.

In contrast, urban trees across the globe are predominantly exotic species. In Rio de Janeiro two thirds of the street trees are exotic species (dos Santos et al., 2009). In Christchurch, New Zealand exotic tree species dominated in parks and streets whilst native trees were more common in residential gardens (Stewart et al., 2009). In New York over 40% of the street tree population consists of two exotic species, Norway maple (*Acer platanoides*) and London plane tree (*Platanus acerifolia*) (Small and Lu, 2006).

There are significant conservation benefits to the promotion of native tree species in urban environments (McKinney, 2006) although stress factors such as limited root and crown space, soil compaction, soil and air pollution, high salinity, and vandalism limit the species that are capable of surviving there. Nevertheless, there are interesting examples in Sweden of the protection of pockets of native vegetation within new urban development (Florgård, 2000) and restoration of native forest in urban areas in New Zealand (Stewart et al., 2004).

## Landscape value

TOWs have ecologic impacts far beyond the proportion of land they occupy, by increasing the permeability and habitat value of the whole landscape. Tree cover, including both woodland and non-woodland trees, has frequently been found to be a key determinant of species distributions and community properties at a landscape level. TOWs not only provide habitat but also enhance the habitat value of non-woodland matrix for many species. A survey of the diversity of vertebrate species in Oxford city concluded that species richness could be preserved by maintaining a system of small (≥0.65 ha) patches of undisturbed woodland throughout the city area (Dickman, 1987). The proportion of urban tree cover has been found to be strongly correlated with bat activity(Gehrt and Chelsvig, 2003). Lumsden and Bennett (2005) found that scattered trees in agricultural land in southeast Australia increased the foraging potential and hence the habitat value to eleven species of bat. In the same area, the richness of bird species tolerant of open conditions was found to be related to the density of scattered trees (Haslem and Bennett, 2008).

TOWs form an important part of the network of woodland habitats and play a key role in maintaining the diversity of larger woodland patches. For example, the pattern of ancient woodland indicator species richness in over 200 woodland patches in the British lowlands was partially explained by the length of hedgerows and lines of trees in a 1 km square around each woodland plot and the total tree cover within a radius of 500 m (Petit et al., 2004).

## Carbon storage

TOWs play an important role in both the sequestration and storage of carbon. Not only do they store substantial amounts of carbon in live biomass but also cause a local increase in soil organic carbon (SOC). A recent analysis of this effect in France (Follain et al., 2007) demonstrated that under typical agricultural field conditions SOC stocks may be increased in the vicinity of trees and hedges by more than 3 kg C $m^{-2}$.

The role of TOWs in carbon sequestration is rarely acknowledged and little consideration has been given to the ways that their important ecosystem function might be enhanced.

The UK is not alone in ignoring their potential. Carbon sequestration projects in the US have largely focused on adaptive management of existing forests and conservation tillage of croplands (Perry et al., 2009). The recently published Read Report (Read et al., 2009) on the role of UK forests in combating climate change, calls for an increase in the rate of woodland creation to 23,200 ha per year over the next 40 years. This would result in afforestation of a total of 928,000 ha of land, equivalent to 5.6% of the 16.5 million ha of non-wooded agricultural land in the UK (DEFRA, 2009). A major argument against the planting of new forests to sequester atmospheric carbon is that it would take land out of use for the production of food and other crops. The carbon cost of importing food rather than producing it locally may well offset some of the benefits of this afforestation. However, by planting TOWs in field margins, urban parks and gardens and on road-sides it would be possible to achieve a significant increase in carbon storage while the bulk of agricultural land remains in its original working land use. The Countryside Survey (2009) found there to be 363,000 km of fenced boundaries in England in 2007. Replacing these with hedges and TOWs would provide over 70,000 ha of new planting – assuming an average width for these boundaries of only 2 m – and would also have many other environmental and conservation benefits.

Falloon *et al.* (2004) have modelled the potential for achieving significant carbon sequestration through expansion and management of field margin strips. Their analysis shows that margins planted with grass, shrubs and trees of 2, 6 or 20 m would capture 0.7 – 20% of the UK Quantified Emission Limitation Reduction Commitment.

Urban trees can also play an important role in storing carbon (Rowntree and Nowak, 1991, Nowak and Crane, 2002) and there is huge potential for increasing their role, both as carbon stores and in ameliorating environmental stress in the face of climate change. Nowak and Crane (2002) found that urban trees were, on average, larger than those in woodland (therefore storing more carbon) and faster growing because they occur at lower densities (and therefore sequester carbon from the atmosphere more rapidly).

## Microclimatic influence

Trees modify microclimate through their effects on wind flow, incoming radiation and the flux of water vapour. Shelter belts and shade trees have been used by humans, for thousands of years, to increase the comfort of their living space and it is only relatively recently that societies have used technology to achieve the same results.

The low albedo, reduced evapotranspiration and high heat output from engines and industry in urban areas leads to the creation of 'heat islands'. Concrete and tarmac have a significantly higher heat capacity and thermal conductivity than vegetated surfaces and once heated by sunlight can act as thermal radiators keeping cities warmer at night. The London Weather Centre report differences in temperature as great as 10˚C between the city and the surrounding countryside. In the US it has been estimated that 5–10% of urban peak electric demand is being used for the additional air-conditioning needed to counteract the urban air temperature increase (Akbari et al., 2001).

Street trees can reduce incoming solar radiation, cooling air temperatures at street level. In Vancouver, Tooke *et al.* (2009) report that street trees reduce solar radiation in summer by 24% and in winter by 13%. They found differences in solar heating between wealthy residential areas and commercial areas due to contrasts in the abundance of street trees. in Hong Kong, Giridharan *et al.* (2008) predict that an increase in tree cover from 25% to 40% in pocket parks in high-density residential areas could reduce the urban heat island intensity by 0.5˚C. Street trees may therefore contribute to reduced carbon emissions both through increased sequestration and by reducing the need for air conditioning. Shade trees gave seasonal cooling energy savings of 30%, corresponding to an average daily savings of 3.6 and 4.8 kWh $d^{-1}$ in two monitored houses in Sacramento, California (Akbari et al., 1997). However, a study by Donovan and Butry (2009) in the same city, has shown that the position of a shade tree relative to a building is critical to its role in reducing heat load. They found that whilst trees planted on the west and south sides of a house reduce summertime electricity use, trees on the north side of a house increase summertime electricity use.

Trees will be of particular value in adapting cities to climate change. Most climate change models do not make predictions for likely urban microclimates but it is evident that the urban heat island effect is likely to be exacerbated by the hotter summers predicted for the UK. (Gill *et al.* (2007) have shown, using a modelling approach, that adding 10% green cover to areas of Greater Manchester with little vegetation, such as the town centre and high-density residential zones would keep maximum surface temperatures at or below the 1961–1990 baseline temperatures up to, but not including, the 2080s high emissions scenario. Trees are predicted to have a superior cooling function to areas of grass or shrubs during periods of prolonged drought as they continue to transpire for longer.

Trees are also important in reducing heat loss through wind chill. The UK is one of the windiest countries in the world and trees have been used for centuries to provide shelter. Many moorland farms in the Pennines are protected from winter gales by a small group of sycamore trees. Shelter belt design is crucial to their effectiveness – some patterns can increase wind speeds locally (Gardiner et al., 2006). Shelter belts have been shown to cut the average energy use of a typical northern US and Canadian farm by 10% to 30% (DeWalle and Heisler, 1988). Trees are also important in giving shelter to stock. Cold, wet and windy conditions stress new-born lambs and freshly shorn sheep leading to a decline in animal health.

In temperate regions, daytime air temperatures within eight times the height of a medium-dense barrier tend to be several degrees warmer than temperatures in the open due to the reduction in turbulent mixing (Brandle et al., 2004). Soil temperatures are also slightly higher in protected areas with advantages for growing crops. However, shelter belts will also shade the soil from solar radiation. Within the shadow soil temperatures may be several degrees lower. The net effect of shelter on crop yields in temperate regions is generally positive although there is huge variation between studies depending on those factors that most constrain plant growth locally and the design of shelter belts (Brandle et al., 2004).

In urban areas tree canopies can act as an efficient filter for gaseous and particulate pollution (Simonich and Hites, 1994, Freer-Smith et al., 1997). However, street trees can also inhibit the dispersion of pollutants emitted by traffic at street level by sheltering the street canyon from turbulent mixing (Gromke and Ruck, 2009). Nor do all species of tree improve air quality equally well. Donovan *et al.* (2005), in a study of tree impacts on urban air quality in Birmingham found that pine, larch, and silver birch had the greatest potential to improve air quality, while oaks, willows, and poplars worsened downwind air quality if planted in very large numbers.

## Hydrological cycles

Trees have an impact on the quantity and intensity of rain water reaching the ground and on the subsequent drainage pathways. Rain falling on a tree canopy first wets the leaves and branches before dripping through to the ground. In brief, low intensity rain showers a very high fraction of rainfall may never reach the ground but will simply evaporate from the canopy. Interception losses can average up to 50% of rainfall in the UK. The leafy canopy also helps to protect the ground surface from the erosive impact of rain drops. Water which falls through the canopy may then infiltrate at the soil surface. Infiltration rates beneath trees are often higher than those in the open. Trees increase soil organic matter making the soil more permeable and giving a higher storage capacity, and their roots and associated mesofauna typically create a surface network of macropores through which soil water can percolate rapidly. Soil moisture is then further depleted by root uptake.

The degree to which TOWs are able to fulfil all of these functions will depend on their landscape context. Rainfall interception and evapotranspiration losses may be higher for more exposed shelter belt and isolated trees. Infiltration rates and saturated hydraulic conductivity have been measured to be higher adjacent to TOWs in pasture and parkland than those in the surrounding grass (Eldridge and Freudenberger, 2005, Chandler and Chappell, 2008).

There is no doubt that in areas where moisture constrains plant growth, competition between TOWs and crops for soil water results in reduced crop yields (Brandle et al., 2004). Historically many TOWS were kept from growing to full size by pollarding and shredding thus reducing shading and competition with crops for moisture.

In urban areas trees may play an important role in regulating problematic storm water run-off. A high level of impermeable surface results in reduced infiltration and rapid run-off resulting in an increased risk of flooding and impacts on water quality. Reduced infiltration affects plant available moisture and the base flow to streams. Not only are green areas a very small proportion of the surface area in most UK cities, many urban trees have an impervious surface beneath the canopy. In most non-residential areas Sacramento, California, Akbari *et al.* (2003) found paved surfaces covered 50–70% of the under- canopy area. Novel systems for storm water management under development in the US, combining structural soils and trees (Day et al., 2008) have potential for use in the UK as intense rainfall and urban flood problems increase (Gill et al., 2007). Such systems, which allow tree roots to develop freely in a highly permeable medium, may help to increase infiltration, promote tree health and circumvent many of the problems of tree root damage to underground services.

# Economic and social value of TOWs

The economic and social benefits of trees in urban areas are well-established. Street trees are now acknowledged to enhance the beauty of cityscapes (Whitford et al., 2001) have health benefits (McDonald et al., 2007, O'Brien, 2005), increase property values (Tyrväinen, 1997), attract business, and improve the liveability of towns (Gill et al., 2007). Trees in towns and cities can also improve living and working conditions by screening undesirable sights, sounds, smells, and dust (Leonard and Parr, 1970).

Trees are not distributed evenly in the urban landscape. Gill *et al.* (2008) found low-density residential areas to be among the most tree covered areas in the Greater Manchester area (25.75%) whilst high density residential, industrial and manufacturing areas were some of the least tree-covered (5-7.5%). The proportion of the population in social groups AB was significantly correlated with the proportion of tree cover in gardens and green space in 5 cities across the UK (Tratalos et al., 2007). Similar relationships between wealth and urban tree cover have been found in the US. Landry and Chakraborty

(2009) found a significantly lower proportion of tree cover on streets in neighbourhoods of Tampa, Florida containing a higher proportion of African-Americans, low-income residents, and renters. Such studies imply that wealthy city residents benefit disproportionately from the benefits of urban trees.

Although trees in urban areas unquestionably provide important ecosystem services to the residents they are often perceived to be a cause of costly problems. Structural damage to buildings and underground services by trees is common. 30% of street trees in Manchester are reported to have caused pavement damage (Wong et al., 1988). Concerns are often expressed about the risks to public safety posed by poorly maintained urban trees in areas of high population density. Eden (2007) reports incidents caused by falling branches, and calls for more rigorous risk inspections.

Scattered trees are also a fundamental part of the cultural landscape of many regions of Britain (Kirby et al., 1995). Hedgerows, copses, orchards and wood-pastures are a product of long-established traditional management systems and bestow a particular local distinctiveness on our countryside. Similarly, many of our most valued townscapes are characterised by majestic street trees.

TOWs in hedges, on roadsides and in meadows have been historically important sources of timber and fodder (Peterken and Allison, 1989) and they continue to provide opportunities to integrate productivity and profitability with environmental stewardship. Wood fuel has the potential to become an important source of clean energy as governments strive to increase renewable energy production, reduce carbon emissions, and expand the use of domestic fuel sources (Lattimore et al., 2009). Although farm woods are seen as critical components of sustainable agricultural systems in much of the developing world, providing a valuable source of fuel as well as other ecosystem services, their potential in this country has been largely ignored.

## Statutory Protection of TOWs

Most TOWs, particularly those growing in rural areas away from public view, have little or no statutory protection, above and beyond the requirement for a Forestry Commission felling license that applies to the felling of any trees with a total timber volume of >5 $m^3$.

Tree Preservation Orders (TPOs) can be made by a local planning authority, under the Town and Country Planning Act 1990 and the Town and Country Planning (Trees) Regulations 1999, if it appears to them to be "expedient in the interests of amenity to make provision for the preservation of trees or woodland in their area". Such orders are, in principle, a good way of protecting trees of high amenity value against imminent threat of damage or destruction, provided that their loss would reduce public enjoyment of the local environment. Other factors, such as importance as a wildlife habitat, may be taken into account, but alone would not be sufficient to warrant a TPO. The tree, or at least part of it, should therefore normally be visible from a public place, such as a road or footpath. The focus on the amenity value of a tree as the criterion for judging whether or not it is worthy of a TPO limits the use of this tool almost entirely to urban areas. In practice most local planning authorities do not have the capacity to monitor the condition of trees covered by TPOs or to enforce those that are violated.

Some woods are statutorily protected sites but a very large proportion of TOWs are found outside these areas. The Government's Planning Policy Statement 9 (ODPM, 2005) advises that:

> *"Aged or veteran trees found outside ancient woodland are also particularly valuable for biodiversity. Planning authorities should encourage the conservation of such trees as part of development proposals."*

but this guidance gives no formal protection.

Perversely, some of the best protection for trees is provided by Conservation Area regulations. Conservation Areas are "areas of special architectural or historic interest" and there is a requirement to give a minimum of six weeks' notice to the local planning authority of proposals to cut down or carry out work on a tree.

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WTPL/Jill Butler

## Changes in the number of TOWS.

The number of trees outside woodland responds to changes in their perceived value and of the value of the land which they occupy. Until the mid eighteenth century TOWs were an important source fodder, fuel and of structural timbers for building and other construction work. Over the next 100 years, rapid agricultural expansion and a switch to the use of coal as the dominant fuel lead to a general decline in small woods and hedgerows. Agriculture suffered a prolonged depression after 1870 and many farmland trees and hedges were left untended. Peterken and Allison (1989) hypothesize that at the time of the first Forestry Commission census of TOWs in 1951 there were probably more TOWs than there had been at any time in the previous 200 years. However, timber resources become worryingly depleted during the First World War with very little new planting following in the inter-war years.

It was the potential timber value of TOWs that prompted the 1951 survey of hedgerow, park trees and woods under five acres by the Forestry Commission (1953). This survey – based on comparatively small samples, concluded that there were approximately 73 million hedgerow and park trees in Great Britain (greater than 10 cm dbh) plus a further 75,676 ha of small woods of less than 2 ha.

Unfortunately changes to the census methods make comparison of these figures with those of subsequent surveys very difficult. In the 1979-1982 survey all trees ≥7cm dbh in groups of up to 0.25 ha were included. However, It is possible to compare data collected in 1997 (Forestry Commission, 2001) with that from the 1979-1982 survey.

| | Isolated trees km$^{-2}$ | | Clumps km$^{-2}$ | | Length of linear features km$^{-2}$ | |
|---|---|---|---|---|---|---|
| **Year** | **1980** | **1997** | **1980** | **1997** | **1980** | **1997** |
| England | 110 | 39.9 | 32 | 17.8 | 0.54 | 0.67 |
| Hampshire | 82 | 24.5 | 65 | 19 | 0.75 | 1.0 |
| Cambridgeshire | 24.4 | 24.5 | 8 | 7.3 | 0.2 | 0.65 |
| Oxfordshire | 84 | 15.7 | 29 | 12.2 | 0.7 | 0.5 |

Table 1: Source Forestry Commission (1983, , 2001)

The national figures show that between 1980 and 1997 England suffered a 64% decline in individual trees, a 44% decline in woodland clumps and a 24% increase in linear woodland features. The pattern of change was highly variable from region to region. In the table above we compare the three counties used in this study. Both Oxfordshire and Hampshire would appear to have suffered a dramatic decline in individual trees and clumps of trees, whilst these features would appear to have been conserved in Cambridgeshire.

It is likely that part of this dramatic decline in individual trees in England is attributable to Dutch elm disease. Approximately 25 million elm trees are believed to have died. However, the 1951 census (Forestry Commission, 1953) reports only 4% (by area) of small woods and 19.5% of hedgerow and parkland trees to have been this species. It is also likely that changes to hedgerow management and substantial increases in wild deer populations lead to a failure of tree regeneration.

This pattern of decline may well have continued over the last decade. The Countryside Survey (2009) reports that the total length of woody linear features in England has declined by 8,000 km since 1998 to 547,000km. The Countryside Survey found that the length of managed hedgerows had decreased by 6.1% (26,000km) in England between 1998 and 2007 with a large proportion of these managed hedges turning into lines of trees and relict hedges (which increased by 13.2% across both categories), reflecting a reduction in management intensity. There was a significant increase in the percentage of hedgerows >2 m between 1998 and 2007 whilst the percent of hedgerows between 1 and 2 m decreased. There are no reliable statistics on the extent of wood-pasture in Britain, or on rates of loss or degradation. Rackham (1990)estimated there to be about 3,200 medieval parks in England which equates to approximately 250,000 ha, but very few of these are likely to be actively managed ancient wood-pastures. The UK BAP quotes a figure of 10-20,000 ha currently in a working condition as the best current estimate (Maddock, 2008).

Climate change is further expected to raise losses among hedgerow and urban tree due to increased drought (Redfern and Hendry, 2002)

## Urban Trees

There are very few detailed historical inventories of tree cover in British cities. Growing interest in urban ecology has resulted in a number of recent surveys. Davies *et al.* (2009) using data gathered in 12 cities across the country, estimated that the national stock of trees in domestic gardens was 28,730,986 trees (95% CI = 22,196,978–35,264,995). The Forestry Commission (2001) estimate that there are just over 123 million live trees occurring outside woodland in Great Britain (England, 89,217,000; Scotland, 18,576,900; Wales, 15,334,000). Domestic gardens therefore comprise just under a quarter of the total.

However, it would appear that trees in urban areas are under pressure too. Allison et al 1985 claimed that non-woodland trees were increasing at the greatest rate during their survey but more recent detailed studies suggest that this trend is no longer widespread. Loram *et al.* (2008) suggest that in addition to decreasing numbers of farmland and hedgerow trees, UK garden trees are likely to decline in numbers in the future. They show that the number of trees in gardens is strongly correlated with garden size and warn that government recommendations to increase housing density from 20-25 to 30-50 houses per hectare (DETR 1999) will severely reduce garden size, and therefore numbers of garden trees.

(Pauleit et al., 2005) monitored changes in land use and land cover of 11 residential areas in Merseyside, using aerial photographs taken in 1975 and 2000. They found a loss of green space in all 11 of their study sites. Overall, the more affluent, low density areas lost more green space, especially of tree cover due to infill building on gardens.

It is likely that in addition to losses from domestic gardens there have been significant losses of street trees too. Street trees can suffer major root damage through building construction and utility installation (Williams 1977, Percival 1996, Thomson et al 1997).

The economic and social benefits of urban trees are well-established. Trees enhance the beauty of cityscapes, can have health benefits, increase property values, attract business, and improve the liveability of towns.

WTPL/Mike Townsend

# A summary of the survey of TOWs in three English counties

The objective of this study was to obtain reliable data on the abundance and distribution of TOWs in three areas that could be directly compared with survey data collected in 1997 by the Forestry Commission. If the rapid decline in individual trees and tree clumps, recorded between 1980 and 1997 had continued unabated then this would represent a catastrophic loss of trees and their associated ecosystem services that had occurred more or less unnoticed. At the same time we recorded the abundance and distribution of TOWs for the identical sample areas in 1948 in order to corroborate trends identified by other researchers. We expected to find a continuous decline from a high point in 1948.

## Study areas

Three English counties – Oxfordshire, Hampshire and Cambridgeshire were selected for study. Within each of these counties ten circular sampling plots were finally selected giving a total sample area of 4,000 ha per county. Recent Google Earth images of the study areas were available at 30 cm resolution. The outline of all trees and woods that fell, at least partially, inside a circular sample plot were digitised from these images.

Sheets from the 1948 RAF aerial photographic survey of England were scanned at high resolution to create images that coincided with our circular sample plots. Once again, the outline of all trees and woods that fell, at least partially, inside a circular sample plot were digitised.

It is comparatively straightforward to digitise the boundaries of tree canopies on high resolution colour photographs. However, it is very time-consuming to do this for the large areas necessary to obtain reliable statistics.

## Area of TOWs

In the three counties included in this study, TOWs cover a remarkably constant 7% of the land area, over and above that already covered with woods of 2 ha or more in size. In well-wooded Hampshire they therefore comprise just under half of the total tree cover, but in Cambridgeshire with only 1.6% woodland cover, the vast majority of trees are TOWs (79% on average).

The results suggest that total tree cover is composed of woodland cover plus an extra 7% of TOWs. Trees outside woods cover 7% of the country in all three areas examined.

It would be dangerous, at this stage, to extrapolate this pattern across the whole of Great Britain since the study did not include upland or major urban areas and was restricted to rural areas of lowland England. However, we feel that the results can be applied with some confidence to the counties of Cambridgeshire, Hampshire and Oxfordshire.

## Changes since 1948

Tree cover has increased dramatically since 1948 – more than doubling in Cambridgeshire and Oxfordshire, and increasing 1.6 fold in Hampshire. Most of this increase is attributable to the transformation of hedges into linear woodland and the expansion into farmland of suburban gardens. These changes should have increased the permeability of the landscape to some woodland species.

It is probable that in 1948, at the time of the RAF aerial survey used in this research, many trees, both in and outside woodland, had been harvested to satisfy the huge war-time demand for home grown timber. We conclude that 1948 may have been a significant low-point in tree cover in Great Britain. In this project we have not examined trends over the last twenty years, but other studies, detailed in our literature review, suggest that it is downwards. It is therefore likely that the last half century began with a period of recovery and regeneration, after the depredations of the two World Wars to be followed by a period of decline.

# References


Akbari, H., Kurn, D. M., Bretz, S. E. & Hanford, J. W. (1997) Peak power and cooling energy savings of shade trees. *Energy and Buildings,* **25,** 139-148.

Akbari, H., Pomerantz, M. & Taha, H. (2001) Cool surfaces and shade trees to reduce energy use and improve air quality in urban areas. *Solar Energy,* **70,** 295-310.

Akbari, H., Shea Rose, L. & Taha, H. (2003) Analyzing the land cover of an urban environment using high-resolution orthophotos. *Landscape and Urban Planning,* **63,** 1-14.

Bland, J. M. & Altman, D. G. (1986) Statistical methods for assessing agreement between two methods of clinical measurement. *Lancet,* **i,** 307-310.

Brandle, J. R., Hodges, L. & Zhou, X. H. (2004) Windbreaks in North American agricultural systems. *Agroforestry Systems,* **61-62,** 65-78.

Brooker, R. W., Osler, G. H. R. & Gollisch, J. (2008) Association of vegetation and soil mite assemblages with isolated Scots pine trees on a Scottish wet heath. *Landscape Ecology,* **23,** 861-871.

Bunnell, F. (1999) What habitat is an island? *Forest wildlife and fragmentation: management implications* (eds J. Rochelle, L. Lehman & J. Wisniewski), pp. 1-31. Brill, Leiden.

Butler, J. E., Rose, F. & Green, T. E. (2001) Ancient trees: icons of our most important wooded landscaped in Europe. *Tools for Preserving Woodland Biodiversity* (eds L. Andersson, R. Marciau, H. Paltto, B. Tardy & H. Read), pp. 20–26. Töreboda Tryckeri AB, Sweden.

Chandler, K. R. & Chappell, N. A. (2008) Influence of individual oak (Quercus robur) trees on saturated hydraulic conductivity. *Forest Ecology and Management,* **256,** 1222-1229.

Countryside Survey (2009) Boundary and Linear Features, Broad Habitat. *England Results from 2007*pp. 70-82. NERC Centre for Ecology & Hydrology, Department for Environment, Food and Rural Affairs, Natural England.

Davies, Z. G., Fuller, R. A., Loram, A., Irvine, K. N., Sims, V. & Gaston, K. J. (2009) A national scale inventory of resource provision for biodiversity within domestic gardens. *Biological Conservation,* **142,** 761-771.

Day, S. D., Dove, J. E., Bartens, J. & Harris, J. R. (2008) Stormwater management that combines paved surfaces and Urban trees. *Geotechnical Special Publication*, pp. 1129-1136.

DEFRA (2009) Agricultural Statistics in your Pocket. Farming Statistics, Accounts and Publications, Department for Environment, Food and Rural Affairs, York.

DeWalle, D. R. & Heisler, G. M. (1988) Use of windbreaks for home energy conservation. : . *Agriculture Ecosystems and Environment,* **22-23,** 243-260.

Dickman, C. R. (1987) Habitat fragmentation and vertebrate species richness in an urban environment. *Journal of Applied Ecology,* **24,** 337-351.

Donovan, G. H. & Butry, D. T. (2009) The value of shade: Estimating the effect of urban trees on summertime electricity use. *Energy and Buildings,* **41,** 662-668.

Donovan, R. G., Stewart, H. E., Owen, S. M., Mackenzie, A. R. & Hewitt, C. N. (2005) Development and application of an urban tree air quality score for photochemical pollution episodes using the Birmingham, United Kingdom, area as a case study. *Environmental Science and Technology,* **39,** 6730-6738.

dos Santos, A. R., da Rocha, C. F. D. & Bergallo, H. G. (2009) Native and exotic species in the urban landscape of the city of Rio de Janeiro, Brazil: density, richness, and arboreal deficit. *Urban Ecosystems***,** 1-14.

Dunn, R. R. (2000) Isolated trees as foci of diversity in active and fallow fields. *Biological Conservation,* **95,** 317-321.

Eden, N. (2007) Towards a national standard for tree risk inspections. *Arboricultural Journal,* **30,** 127-136.

Eldridge, D. J. & Freudenberger, D. (2005) Ecosystem wicks: Woodland trees enhance water infiltration in a fragmented agricultural landscape in eastern Australia. *Austral Ecology,* **30,** 336-347.

Elmore, A. J., Mustard, J. F., Manning, S. J. & Lobell, D. B. (2000) Quantifying vegetation change in semiarid environments: Precision and accuracy of spectral mixture analysis and the normalized difference vegetation index. *Remote Sensing of Environment,* **73,** 87-102.

Eshiamwata, G.W., Berens, D. G., Bleher, B., Dean, W. R. J. & Böhning-Gaese, K. (2006) Bird assemblages in isolated *Ficus* trees in Kenyan farmland. *Journal of Tropical Ecology,* **22,** 723-726.

Falloon, P., Powlson, D. & Smith, P. (2004) Managing field margins for biodiversity and carbon sequestration: A Great Britain case study. *Soil Use and Management,* **20,** 240-247.

FAO (2004) Global Forest Resources Assessment Update 2005 – Terms and Definitions (Final Version). *Forest Resources Assessment Programme Working Paper* pp. 36. Forestry Department, Food and Agriculture Organization of the United Nations, Rome.

Fischer, J. & Lindenmayer, D. B. (2002a) The conservation value of paddock trees for birds in a variegated landscape in southern New South Wales. 2. Paddock trees as stepping stones. *Biodiversity and Conservation,* **11,** 833-849.

Fischer, J. & Lindenmayer, D. B. (2002b) Small patches can be valuable for biodiversity conservation: Two case studies on birds in southeastern Australia. *Biological Conservation,* **106,** 129-136.

Florgård, C. (2000) Long-term changes in indigenous vegetation preserved in urban areas. *Landscape and Urban Planning,* **52,** 101-116.

Follain, S., Walter, C., Legout, A., Lemercier, B. & Dutin, G. (2007) Induced effects of hedgerow networks on soil organic carbon storage within an agricultural landscape. *Geoderma,* **142,** 80-95.

Forestry Commission (1953) Hedgerow and Park Timber and Woods under Five Acres 1951. *Census Report*. HMSO, London

Forestry Commission (1983) Census of Woodland and Trees, 1979 - 1982. England. Forestry Commission, Edinburgh.

Forestry Commission (2001) National Inventory of Woodland and Trees - England. *Inventory Report*. Forestry Commission, Edinburgh.

Freer-Smith, P. H., Holloway, S. & Goodman, A. (1997) The uptake of particulates by an urban woodland: Site description and particulate composition. *Environmental Pollution,* **95,** 27-35.

Fritz, L.W. (1999 ) High resolution commercial remote sensing satellites and spatial information systems. *ISPRS Highlights* **4**.

Gardiner, B., Palmer, H. & Hislop, M. (2006) The Principles of Using Woods for Shelter. *Forestry Information Note*. Forestry Commission, Edinburgh.

Gehrt, S. D. & Chelsvig, J. E. (2003) Bat activity in an urban landscape: Patterns at the landscape and microhabitat scale. *Ecological Applications,* **13,** 939-950.

Gill, S. E., Handley, J. F., Ennos, A. R. & Pauleit, S. (2007) Adapting cities for climate change: The role of the green infrastructure. *Built Environment,* **33,** 115-133.

Gill, S. E., Handley, J. F., Ennos, A. R., Pauleit, S., Theuray, N. & Lindley, S. J. (2008) Characterising the urban environment of UK cities and towns: A template for landscape planning. *Landscape and Urban Planning,* **87,** 210-222.

Giridharan, R., Lau, S. S.Y., Ganesan, S. & Givoni, B. (2008) Lowering the outdoor temperature in high-rise high-density residential developments of coastal Hong Kong: The vegetation influence. *Building and Environment,* **43,** 1583-1595.

Goetz, S. J., Wright, R. K., Smith, A. J., Zinecker, E. & Schaub, E. (2003) IKONOS imagery for resource management: Tree cover, impervious surfaces, and riparian buffer analyses in the mid-Atlantic region. *Remote Sensing of Environment,* **88,** 195-208.

Götmark, F. & Thorell, M. (2003) Size of nature reserves: Densities of large trees and dead wood indicate high value of small conservation forests in southern Sweden. *Biodiversity and Conservation,* **12,** 1271-1285.

Gromke, C. & Ruck, B. (2009) On the impact of trees on dispersion processes of traffic emissions in street canyons. *Boundary-Layer Meteorology,* **131,** 19-34.

Grüebler, M. U., Morand, M. & Naef-Daenzer, B. (2008) A predictive model of the density of airborne insects in agricultural environments. *Agriculture, Ecosystems and Environment,* **123,** 75-80.

Guevara, S., Purata, S. & Van Der Maarel, E. (1986) The role of remnant trees in tropical secondary succession. *Vegetatio,* **66,** 74-84.

Hansen, M. C., DeFries, R. S., Townshend, J. R. G., Carroll, M., Dimiceli, C. & Sohlberg, R. A. (2003) Global Percent Tree Cover at a Spatial Resolution of 500 Meters: First Results of the MODIS Vegetation Continuous Fields Algorithm. *Earth Interactions,* **7, 1-15**.

Haslem, A. & Bennett, A. F. (2008) Birds in agricultural mosaics: The influence of landscape pattern and countryside heterogeneity. *Ecological Applications,* **18,** 185-196.

Kirby, K. J., Thomas, R. C., Key, R. S., McLean, I. F. G. & Hodgetts, N. (1995) Pasture-woodland and its conservation in Britain. *Biological Journal of the Linnean Society,* **56,** 135-153.

Kleinn, C. (2000) On large-area inventory and assessment of trees outside forests. *Unasylva,* **51** 3-10.

Landry, S. M. & Chakraborty, J. (2009) Street trees and equity: Evaluating the spatial distribution of an urban amenity. *Environment and Planning A,* **41,** 2651-2670.

Lattimore, B., Smith, C. T., Titus, B. D., Stupak, I. & Egnell, G. (2009) Environmental factors in woodfuel production: Opportunities, risks, and criteria and indicators for sustainable practices. *Biomass and Bioenergy,* **33,** 1321-1342.

Leonard, R. E. & Parr, S. B. (1970) Trees as a sound barrier. *Journal of Forestry,* **68,** 282–283.

Levin, N., McAlpine, C., Phinn, S., Price, B., Pullar, D., Kavanagh, R. P. & Law, B. S. (2009) Mapping forest patches and scattered trees from SPOT images and testing their ecological importance for woodland birds in a fragmented agricultural landscape. *International Journal of Remote Sensing,* **30,** 3147-3169.

Loram, A., Warren, P. H. & Gaston, K. J. (2008) Urban domestic gardens (XIV): The characteristics of gardens in five cities. *Environmental Management,* **42,** 361-376.

Lumsden, L. F. & Bennett, A. F. (2005) Scattered trees in rural landscapes: Foraging habitat for insectivorous bats in south-eastern Australia. *Biological Conservation,* **122,** 205-222.

Maddock, A. (2008) UK Biodiversity Action Plan; Priority Habitat Descriptions. UK Biodiversity Reporting and Information Group.

Mathieu, R., Freeman, C. & Aryal, J. (2007) Mapping private gardens in urban areas using object-oriented techniques and very high-resolution satellite imagery. *Landscape and Urban Planning,* **81,** 179-192.

McDonald, A. G., Bealey, W. J., Fowler, D., Dragosits, U., Skiba, U., Smith, R. I., Donovan, R. G., Brett, H. E., Hewitt, C. N. & Nemitz, E. (2007) Quantifying the effect of urban tree planting on concentrations and depositions of PM10 in two UK conurbations. *Atmospheric Environment,* **41,** 8455-8467.

McKinney, M. L. (2006) Urbanization as a major cause of biotic homogenization. *Biological Conservation,* **127,** 247-260.

Nemitz, D., Kunth, S., Kleinn, C., Fischer, H. & Bücking, M. (2007) Comparing two approaches of mapping woody vegetation in open land (Ein vergleich von zwei methodischen konzepten zur kartierung von feldgehölzen in mittelgroßen untersuchungsgebieten. *Allgemeine Forst- und Jagdzeitung,* **178,** 135-141.

Nowak, D. J. & Crane, D. E. (2002) Carbon storage and sequestration by urban trees in the USA. *Environmental Pollution,* **116,** 381-389.

O'Brien, E. (2005) Trees and woodland: nature's health service. pp. 48. Forest Research, Farnham.

ODPM (2005) Planning Policy Statement 9: Biodiversity and Geological Conservation. (ed Office of the Deputy Prime Minster). HMSO.

Ohsawa, M. (2007) The role of isolated old oak trees in maintaining beetle diversity within larch plantations in the central mountainous region of Japan. *Forest Ecology and Management,* **250,** 215-226.

Otero-Arnaiz, A., Castillo, S., Meave, J. & Ibarra-Manriquez, G. (1999) Isolated pasture trees and the vegetation under their canopies in the Chiapas Coastal Plain, Mexico. *Biotropica,* **31,** 243-254.

Ozanne, C. M. P., Speight, M. R., Hambler, C. & Evans, H. F. (2000) Isolated trees and forest patches: Patterns in canopy arthropod abundance and diversity in Pinus sylvestris (Scots Pine). *Forest Ecology and Management,* **137,** 53-63.

Paletto, A., De Natale, F., Gasparini, P., Morelli, S. & Tosi, V. (2006) L'Inventario degli Alberi Fuori Foresta (IAFF) come strumento di analisi del paesaggio e supporto alle scelte di pianificazione territoriale. *Forest@,* **3,** 253-266.

Pauleit, S., Ennos, R. & Golding, Y. (2005) Modeling the environmental impacts of urban land use and land cover change--a study in Merseyside, UK. *Landscape and Urban Planning,* **71,** 295-310.

Perry, C. H., Woodall, C. W., Liknes, G. C. & Schoeneberger, M. M. (2009) Filling the gap: Improving estimates of working tree resources in agricultural landscapes. *Agroforestry Systems,* **75,** 91-101.

Peterken, G. F. & Allison, H. (1989) Woods, Trees and Hedges: a Review of Changes in the British Countryside. *Focus on Nature Conservation*. Nature Conservancy Council, Peterborough, UK.

Petit, S., Griffiths, L., Smart, S. S., Smith, G. M., Stuart, R. C. & Wright, S. M. (2004) Effects of area and isolation of woodland patches on herbaceous plant species richness across Great Britain. *Landscape Ecology,* **19,** 463-471.

Rackham, O. (1990) *Trees and Woodland in the British Landscape*. Dent, London.

Read, D. J., Freer-Smith, P. H., Morison, J. I. L., Hanley, N., West, C. C. & Snowdon, P. (2009) Combating climate change – a role for UK forests. An assessment of the potential of the UK's trees and woodland to mitigate and adapt to climate change. The synthesis report. The Stationery Office, Edinburgh.

Redfern, D. & Hendry, S. (2002) Climate change and damage to trees caused by extremes of temperature. *Climate Change: Impacts on UK Forests* (ed M. Broadmeadow), pp. 29-39. Forestry Commission, Edinburgh.

Rowntree, R. A. & Nowak, D. J. (1991) Quantifying the role of urban forests in removing atmospheric carbon dioxide. *Journal of Arboriculture,* **17,** 269-275.

Simonich, S. L. & Hites, R. A. (1994) Importance of vegetation in removing polycyclic aromatic hydrocarbons from the atmosphere. *Nature,* **370,** 49-51.

Small, C. & Lu, J. W. T. (2006) Estimation and vicarious validation of urban vegetation abundance by spectral mixture analysis. *Remote Sensing of Environment,* **100,** 441-456.

Southwood, T. R. E. (1961) The number of species of insect associated with various trees. *Journal of Animal Ecology,* **30,** 1-8.

Stewart, G. H., Ignatieva, M. E., Meurk, C. D. & Earl, R. D. (2004) The re-emergence of indigenous forest in an urban environment, Christchurch, New Zealand. *Urban Forestry and Urban Greening,* **2,** 149-158.

Stewart, G. H., Meurk, C. D., Ignatieva, M. E., Buckley, H. L., Magueur, A., Case, B. S., Hudson, M. & Parker, M. (2009) Urban biotopes of Aotearoa, New Zealand (URBANZ) II: Floristics, biodiversity and conservation values of urban residential and public woodland, Christchurch. *Urban Forestry and Urban Greening,* **8,** 149-162.

Tilzey, M. (2000) Natural Areas, the whole countryside approach and sustainable agriculture. *Land Use Policy,* **17,** 279-294.

Toh, I., Gillespie, M. & Lamb, D. (1999) The role of isolated trees in facilitating tree seedling recruitment at a degraded sub-tropical rainforest site. *Restoration Ecology,* **7,** 288-297.

Tooke, T. R., Coops, N. C. & Voogt, J. A. (2009) Assessment of urban tree shade using fused LIDAR and high spatial resolution imagery. *2009 Joint Urban Remote Sensing Event.*

Tratalos, J., Fuller, R. A., Warren, P. H., Davies, R. G. & Gaston, K. J. (2007) Urban form, biodiversity potential and ecosystem services. *Landscape and Urban Planning,* **83,** 308-317.

Tyrväinen, L. (1997) The amenity value of the urban forest: An application of the hedonic pricing method. *Landscape and Urban Planning,* **37,** 211-222.

Whitford, V., Ennos, A. R. & Handley, J. F. (2001) "City form and natural process" - Indicators for the ecological performance of urban areas and their application to Merseyside, UK. *Landscape and Urban Planning,* **57,** 91-103.

Wong, T. W., Good, J. E. G. & Denne, M. P. (1988) Tree root damage to pavements and kerbs in the city of Manchester. *Arboricultural Journal,* **12,** 17-34.

Yasuda, M. & Koike, F. (2009) The contribution of the bark of isolated trees as habitat for ants in an urban landscape. *Landscape and Urban Planning,* **92,** 276-281.

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