

Short Rotation Forestry: Review of growth and environmental impacts

Helen McKay (Ed.)

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Introduction

Helen McKay

1. Background

The UK has a target for 15% of its energy consumption in 2020 to be from renewable sources. Under Article 4 of the European Renewable Energy Directive (2009/28/EC) each Member State had to submit a plan (National Renewable Action Plan) to reach this target. The UK's lead scenario is that around 30% of electricity, 12% of heat and 10% of transport energy come from renewable sources by 2020. While this does not represent fixed targets for particular sectors it is anticipated that biomass will play a key part in delivering the overall target.

Delivering the European renewable energy target is expected to increase the demand for biomass feedstocks in the heat and power sectors. The analysis presented in the National Renewable Action Plan suggests that there could be sufficient biomass resource potential in the UK to meet this demand for heat and power in 2020, if the supply potential is fully developed. This assessment assumes that we could rapidly increase the production of energy crops in the UK, increase wood fuel supply from sustainable forestry, make better use of agricultural residues and fully exploit waste biomass currently going to landfill.

It has been suggested that short rotation forestry, particularly using Eucalyptus, has the potential to deliver greater volumes of biomass from the same land area than alternative biomass crops. Short rotation forestry (SRF) is the practice of cultivating fast-growing trees that reach their economically optimum size between eight and 20 years old; each plant produces a single stem that is harvested at around 15 cm diameter. The crops tend to be grown on lower-grade agricultural land, previously forested land or reclaimed land and so do not directly compete with food crops for the most productive agricultural land. Experience of SRF in Britain is limited creating a need to establish whether it is a viable renewable energy source.

In 2006, the influential Hardcastle review of the environmental impacts of growing fast-growing broadleaved species as SRF in Britain (Hardcastle 2006) recommended that a series of field trials was set up to further investigate impacts where relevant data were particularly sparse; corroborate conclusions where some data were available; and

provide the foundation for assessments of growth and yield. Funding was made available in 2008 to establish the viability, feasibility, sustainability and appropriateness of SRF as a renewable energy source in England; this review is one of seven contract deliverables.

2. Scope and rationale

This review updates the Hardcastle review and where possible strengthens it using primary unpublished data. It draws on the expert judgement of many Forest Research scientists who were given a common brief to review:

- the growth, yield and biomass distribution of the most promising species managed under short rotations
- potential impacts – both positive and negative – of SRF on hydrology; above-ground biodiversity; the historic environment; soil and nutrient dynamics
- the potential impact of pathogens and pests – both insects and mammals – on growth

The species covered are those being considered for the complementary field trials: ash, hybrid aspen, red alder, Italian alder, sycamore, silver birch, sweet chestnut, shining gum (*Eucalyptus nitens*), Tingiringy gum (*E. glaucescens*), cider gum (*E. gunnii*), and rauli (*Nothofagus alpina*, syn. *N. procera*). Authors were directed to concentrate on the species creating most concern among environmental NGOs and the general public, i.e. the eucalypts, with less intensive effort devoted to the native/naturalised species (ash, sycamore, sweet chestnut, and birch).

Authors were guided to focus on the most relevant references and data on the basis that it is more useful to have fewer relevant and thoroughly evaluated datasets than many references that are only peripherally related to the candidate SRF species or British conditions. Information from other countries is included only when it is obviously relevant with preference given to publications and data from Britain. Since crop rotation is likely to be <10 years for the eucalypts, <20 for the native/naturalised species and <15 for the others, the review does not consider information from older stands.

3. Comparators

Within each chapter impacts are described in both absolute and relative terms. The project board, which comprised a senior representative from each of the funders (DEFRA and later DECC, Natural England, the Environment Agency and Forestry Commission England) gave clear guidance that the field trials were to be located as far as possible on

improved grazing of Class 3 and 4. The rationale for this selection was that better land was unlikely to be available because of land prices and the imperative for food production while on the other hand availability of unimproved grassland would be restricted because of its higher biodiversity value. Consequently the review emphasised comparisons with crops found on Class 3 and 4 land and only occasionally made comparisons with more intensive arable crops, even though they might well have been even more in SRF's favour. Likewise, comparisons were not made with Ancient Semi-natural Woodland since SRF would not be used to replace this current land use.

4. References

Hardcastle, P.D., Calder, I., Dingwall, I., Garrett, W., McChesney, I., Mathews, J., Savill, P. 2006. *A review of the impacts of short rotation forestry*. Final Report on SRF by LTS International, February 2006

5. Acknowledgements

Funding was provided by Defra, subsequently DECC. We received valuable guidance from the Project Board which comprised Andrew Perrins and Jayne Redrup (DECC), Keith Kirby (Natural England), Laura Jones (Forestry Commission England) and representatives of the Environment Agency. The project also benefited greatly from its steering group of Jonathan Hayward and Katharine Donne (DECC), Mark Broadmeadow and Chris Rider (Forestry Commission England) in addition to Alan Harrison and Helen McKay (Forest Research). Particular thanks are due to Andy Mason who was instrumental in securing funding for and initiating the project.



Mature wet birch, alder riparian woodland with mat

Short Rotation Forestry and Water

Tom Nisbet, Huw Thomas and Nadeem Shah

Summary

The planting of short rotation forestry (SRF) poses a number of potential threats and benefits to water quality and quantity. Potential threats to water quality mainly arise from harvesting operations, which can be controlled by good practice. The overall quality impact of SRF is expected to be largely beneficial compared to the alternative land uses currently practised in areas generally considered appropriate for conversion to SRF. In particular, SRF is expected to significantly improve water quality compared to arable cropping and while conversion of only limited areas of more intensively farmed land may be appropriate, there are believed to be major opportunities for targeted planting of SRF to mitigate potential pollutant sources and interrupt delivery pathways to watercourses. Thus judicious planting of SRF could help tackle the major diffuse nutrient and sediment pollution pressures associated with agriculture and urban activities.

The principal water concern relates to the potential high water use of SRF crops, which could have a major impact on local water resources. The greatest threat is presented by conifer and some broadleaved SRF crops such as Eucalyptus, while the use of other broadleaved species could possibly benefit water resources. Much will depend on location and scale of planting, with inevitable trade-offs between improved biomass yield, carbon sequestration, and water quality on one hand and reductions in the volume of water reaching rivers and groundwater on the other. Specific opportunities exist for utilising the high water use of particular SRF crops for environmental gain, such as for wastewater treatment and flood risk management.

This assessment of the likely impact of SRF planting on water is largely based on our understanding of the hydrological effects of conventional forestry, supplemented by a number of studies on Short Rotation Coppice (SRC). SRF is a relatively new approach and we await the results of ongoing research to underpin the findings of this review.

Correct advice is essential to ensure that SRF planting schemes deliver the multiple environmental benefits they offer. In future catchment management plans may provide an effective way of integrating SRF and alternative land uses, especially for land and water managers and landscape planners.

1. Introduction

This section considers the potential impacts of short rotation forestry (SRF) on water. The review emphasises comparisons with crops found on Class 3 and 4 agricultural land rather than with the more intensive arable cropping on Class 1 and 2 land. Likewise, comparisons were not made with Ancient Semi-natural Woodland since SRF would not be used to replace this current land use.

The planting of SRF poses a number of potential threats and benefits to water, with the magnitude of effect dependent on scale, location, and forest design and management (Armstrong, 1999; Hardcastle *et al.*, 2006). Scale is probably the most important issue determining the significance of impact at the level of a river, lake or groundwater body. To a large extent, as the proportion of the catchment area occupied by SRF declines, its 'signature' will be progressively diluted by that of other land covers. This is particularly relevant to SRF, since planting is unlikely to be extensive, but involving small blocks rather than large plantations. Studies generally show that it is very difficult to identify any water impacts against natural background variation when less than 20% of a catchment is subject to woodland creation or removal (Cornish, 1993).

The principal water concern relates to the typically high water use of SRF crops, which could have an adverse impact on local water resources. In contrast, an expansion of SRF is thought likely to benefit water quality due to lower chemical inputs and reduced soil disturbance compared to the previous land use. The main water quality and quantity issues concerned with an expansion of SRF are reviewed below.

To date, few studies have investigated the impacts of SRF on water, with no field studies in the UK. Much of this review draws on related work on SRC or on wider knowledge of conventional, longer-rotation forestry. In doing so, it is necessary to take account of a number of important differences in site sensitivity and management practice between SRF and conventional forestry that are expected to influence the impacts on water. Key modifiers include:

- Location: with a preponderance for more lowland, ex-agricultural sites, where the impact of SRF is likely to be less than that of the pre-existing land use in terms of water quality but greater for water quantity
- Scale: with planting likely to involve small blocks, reducing the impact on water yield as well as providing significant scope for targeting pollutant sources and interrupting pathways to reduce diffuse pollution

- Species choice: with the potential to plant higher yielding species (e.g. Eucalyptus species) with characteristically high nutrient uptake and water use
- Management: with more frequent harvesting and constraints on machine choice presenting a greater risk of site disturbance and damage, leading to increased sediment delivery and nutrient leaching.

2. Water quality

SRF has the potential to both threaten and benefit water quality. The threats are considered first and mainly relate to the management practices and interventions associated with SRF, rather than the presence of the crop *per se*. The main exception is the increased canopy capture of atmospheric pollutants.

a) Threats to water quality

Impact of woodland operations

The main threat to water quality is likely to be the delivery of sediment to watercourses as a result of ground damage caused by the eventual harvesting of SRF crops (Armstrong, 1999). Heavy, poorly drained agricultural soils would be most at risk, especially if machine harvesting was constrained to winter periods to maximise woodfuel quality and avoid impacts on breeding birds or disturbance to other wildlife. The disruption of existing field drains by rooting or machine trafficking could accentuate this problem. Impacts, however, could be less than those of the previous agricultural land use due to the reduced frequency of soil cultivation and cropping cycles.

No studies appear to have assessed the risk of diffuse sediment pollution but it is expected that 'standard' good practice measures would be effective at protecting soils from compaction and rutting. Restricting operations to drier periods on vulnerable soils and the retention of undisturbed riparian buffer areas would be crucial, as would be the use of part of the biomass crop to protect machine access routes from trafficking. Selecting appropriate machinery for the ground conditions would also be important, although machine choice is likely to be restricted for smaller-scale, farm working. In such cases, trailer loads may need to be limited to protect soils. Particular care is needed when operating in riparian zones. Implementation of the Forestry Commission's Water Guidelines (Forestry Commission, 2003) should help to control the risk of sediment pollution from harvesting and other operations, such as site cultivation, machine planting or track construction (Hardcastle *et al.*, 2006). Specific good practice guidance for SRF is being drafted.

Other possible water quality issues could arise from fertiliser and pesticide use, although needs for both are likely to be less where SRF replaces arable or intensive grassland. Fertiliser is unlikely to be required for the first and possibly second rotation, with the potential for crop nutrient requirements to be partly sustained by atmospheric deposition and nutrient runoff from adjacent land, depending on location. The magnitude of nutrient losses and thus potential drain on the site will depend on factors such as species choice, harvesting frequency and the degree of utilisation, e.g. whether trees are harvested in leaf or tree stumps removed. Pesticide requirements will be governed by the need for weed control in the first few years after planting. The diffuse pollution risk from both fertiliser and pesticide use in SRF is judged to be significantly less than for the previous land use (see below) and effectively controlled by good practice measures vis-à-vis the Forests & Water Guidelines (Shepard, 2006; Newell Price *et al.*, 2008).

Scavenging of atmospheric pollutants

In common with forests in general, SRF crops can increase pollutant deposition due to the high aerodynamic roughness of the canopy (Forestry Commission, 2003). The faster growth and therefore more rapid canopy development of SRF could exacerbate the scavenging effect, as could the greater edge of smaller SRF blocks. Scavenging would be greatest for evergreen species downwind of local pollutant sources. The main risk would result from the increased capture of sulphur and nitrogen (ammonia) pollutants, which could contribute to surface water acidification and nutrient enrichment of local waters.

Water acidification would only be an issue in acid sensitive areas, which are largely confined in the UK to parts of the uplands of southwest and central Scotland, Cumbria, the Pennines, Wales and the Mourne Mountains in Northern Ireland (Forestry Commission, 2003). The ongoing recovery of waters due to emission control (ECRC, 2010), the limited suitability of these areas for SRF and the restricted scale of any planting would present a marginal risk of acidification. In contrast, nutrient enrichment is more of a lowland issue and therefore of greater relevance to SRF expansion. However, the strong nutrient demand of fast growing SRF crops is likely to maximise nitrogen retention within the crop and minimise losses to water (see below).

A related issue is the risk of increased acidification posed by the greater drain on soil base cations by repeated rotations of high yielding SRF crops. If bases are not replaced through fertilisation or the application of wood-ash, this could degrade soils in the longer-term and lead to increased surface water acidification within acid sensitive areas.

b) Benefits to water quality

Providing good practice controls the threats outlined above, SRF planting has the potential to improve water quality. The main benefit would be using the strong nutrient demand of SRF crops to help reduce losses from ex-agricultural land, or to intercept and remove diffuse pollutants in waters draining from adjacent ground. Alternatively, SRF

could be used to help treat applications of wastewater or recycled organic materials to land. To be most effective, planting would need to target nutrient sources or run-off pathways.

Controlling diffuse pollution

Land conversion to SRF could accentuate the acknowledged benefit of woodland to significantly reduce N losses relative to conventional arable or grassland (Addy *et al.*, 1999). For example, annual mean N leaching losses for woodland in the Marlborough catchment in Southeast England were estimated to be less than a sixtieth of that for arable fields (26.4, 15.5 and 0.4 kg N ha⁻¹ yr⁻¹ for arable fields, grassland and farm woodland, respectively) based on a modelling study (SHETRAN) by Koo and O'Connell (2006). Silgram (2005) compared nitrate leaching losses between fields in the Nitrate Sensitive Areas (NSA) scheme over the period 1989-2003, with reference areas in adjacent coniferous and deciduous woodland of differing ages and species composition. Based on 60 site-years of data, mean annual leaching losses for agricultural fields, subject to strict NSA rules limiting N fertiliser applications, were 40 kg ha⁻¹ for winter cereals, 48 kg N ha⁻¹ for oilseed rape, 66 kg N ha⁻¹ for potatoes and 17 kg N ha⁻¹ for woodland. The latter value is consistent with the results reported in the wider literature for woodland, which typically lie in the range 0-24 kg N ha⁻¹ yr⁻¹.

SRF could be a particularly effective option for mitigating nitrate leaching in Nitrate Vulnerable Zones by maximising nitrogen removal while providing a high yielding crop for farmers. Although repeated cycles of fast growing SRF would eventually require regular N additions to maintain productivity (which could result in potential high losses of nitrate in the establishment and removal years), losses averaged over individual rotations are likely to be markedly lower than under agricultural crops (Heilman and Norby, 1998).

No data are available from SRF studies in the UK but there are relevant data from studies of both SRC and SRF elsewhere in Europe. Considering firstly European SRC data: Goodlass *et al.* (2007) showed nitrate leaching losses from an SRC plot of willow to be overall very low in relation to more intensively-managed arable land. These results are supported by the findings of a Swedish study by Aronsson *et al.* (2000) who found that despite an SRC crop receiving N fertiliser applications of up to 153 kg N ha⁻¹ during the main growth phase, average nitrate concentrations leaving the root zone remained low (<0.05 mg NO₃-N L⁻¹). Thus, despite repeated fertiliser additions and site disturbance during harvesting phases, the risk of nitrate leaching from SRC appears relatively small (Lord *et al.*, 2007). The longer crop rotation associated with SRF compared to SRC could be expected to further limit overall nitrate losses (Goodlass *et al.*, 2007). Considering secondly European SRF data: Jug *et al.* (1999) evaluated the effects on soil ecology of three SRF plantations on former arable land in Germany. Following afforestation with three species, including hybrid aspen (*P. tremula* x *P.*

tremuloides), there was a ~50% reduction in leaching of nitrate compared to agricultural control plots. On the other hand intensive soil cultivation and SRF establishment on grassland accelerated organic matter mineralization causing high losses of soil carbon (15%) and organic nitrogen (12%). These disturbance effects are likely to be reversed in the medium term with tree rooting, crop growth and inputs of leaf litter. However, the authors recommended cautious strip tillage instead of whole-area ploughing to keep microbiological activity as low as possible and to harmonize N mineralization in soil with the uptake by trees, thereby avoiding nitrate leaching and contamination of groundwater. Studies by Heilman and Norby (1998) and Nyakatawa *et al.* (2006) showed that nitrogen and phosphorus losses remained low in SRF systems due to a combination of strong nutrient uptake and soil denitrification.

The amount of fertiliser applied and the application regime affect the quantity of nutrients leached. A study of a short rotation sycamore plantation found that a single fertiliser application had similar total leaching losses as multiple applications, although nitrate concentrations in the leachate were higher following the single application (van Miegroet *et al.*, 1994). Ground cover may also reduce nitrate leaching losses to water, for example both seeded and natural ground vegetation in a hybrid poplar plantation played an important role in preventing leaching of fertiliser and native soil nitrogen (McLaughlin *et al.*, 1985).

The nutrient removal benefit of SRF could be used to particularly good effect where planting is targeted to downslope field edges or riparian zones to form 'biogeochemical barriers'. A number of long-term investigations in Poland have shown woodland strips to be effective at reducing nitrogen run-off from adjacent fields (Ryszkowski *et al.*, 1999) and exerting a cleansing effect on the chemistry of groundwaters (Cooper *et al.*, 1987; Ryszkowski and Bartoszewicz, 1989; Bartoszewicz, 1990; Bartoszewicz and Ryszkowski, 1996; Ryszkowski *et al.*, 1997, 2002; and Ryszkowski, 2000). Concentrations of nitrate in groundwater within woodland strips, or pine and birch woodland patches, adjacent to cultivated fields were reduced by 76-98% of the input. The efficiency of N removal by shelterbelts was found to be influenced by woodland age, width of strip, season and depth to the water-table (Ryszkowski and Kędziora, 2007).

Other studies have shown that conventional riparian woodland buffers form effective 'nutrient soaks' for both nitrate and phosphate (Kovacic *et al.*, 1991; Osborne and Kovacic, 1993; Schulze *et al.*, 1995; Hubbard and Lowrance, 1997), although there is conflicting evidence on their relative efficacy compared to that of wet grassland (Haycock and Burt, 1993; Lowrance *et al.*, 1995; Schnable *et al.*, 1996; Correll, 1997). The case appears more clearcut with respect to SRF buffers, with experimental work by Schultz *et al.* (1995) showing that fast-growing species were best at stream-side due to their quick establishment, providing nutrient filtering as well as soil stabilising effects

within the first three years of planting. The use of willow was particularly effective due to its ability to root sprout and form a dense root structure.

One case where SRF may not yield a nitrogen benefit is planting in dry regions of the country. A study by Calder *et al.* (2002) found that the high water use of conifer forests led to a disproportionately large concentrating effect with the result that nitrate concentrations in groundwater were similar to those draining arable crops. The Forests & Water Guidelines address this risk by recommending that large-scale conifer planting should be avoided within Nitrate Vulnerable Zones receiving <650 mm annual rainfall. The potential high water use of SRF crops could pose a similar issue, necessitating equivalent planting restrictions.

While the water quality benefits of SRF are likely to be strongest for nitrate control, planting could also help to reduce sediment delivery to water. It is widely acknowledged that soils under woodland are usually well protected and improved (Forestry Commission, 1998), with measurements generally displaying consistently lower sediment losses for watercourses draining well managed woodland compared to other land uses (Nisbet *et al.*, 2010). A review of soil erosion associated with biomass crops concluded that SRF plantations are likely to have a beneficial effect in protecting the soil, reducing surface runoff and physically anchoring the soil (Kort *et al.*, 1998). The potential rapid establishment of SRF could be used to quickly provide effective buffers for intercepting surface runoff from adjacent land and enhancing the retention of sediment and related pollutants, such as adsorbed phosphate. The high water use and vigorous rooting of SRF would also promote the infiltration and storage of runoff, although the bare ground surface under stands would have limited sediment trapping capacity if surface runoff was sustained. However, this limitation could be addressed by the inclusion of grass edge strips in the planting design. Phillips (1989), Haycock and Pinay (1993) and Hubbard and Lowrance (1994) found that a combination of woodland and grassland buffers (as an understorey or adjacent strip) enhanced sediment removal.

Riparian planting would have the added advantage of protecting stream and river banks from erosion by the strengthening action of tree roots (Nisbet *et al.*, 2010). The intensive planting of willow stakes is a recognised bioengineering solution for speeding up the protection of actively eroding river banks but SRF cultivation and harvesting operations would have to be carefully managed to ensure that they did not cause significant damage to riparian soils, which are particularly vulnerable to disturbance and erosion. The riparian zone would remain vulnerable to erosion after harvesting, until the replanted crop became established.

No studies could be found that have quantified pesticide load reductions by SRF buffers, either in the UK or elsewhere, but in common with all types of woodland, SRF is likely to have the potential to reduce diffuse pesticide pollution. Studies have shown woodland

shelterbelts at field margins to be highly effective, achieving reductions in spray drift of between 60 to 90% (Ucar and Hall, 2001; Lazzaro *et al.*, 2008). Similarly, riparian woodland buffer areas can provide effective protection for streams and groundwaters from pesticide applications on adjacent land by both intercepting aerial drift of pesticides and trapping pesticides bound to sediment in run-off (Lowrance *et al.*, 1984).

Water treatment

Energy crops are particularly suited to waste applications and can form an effective waste management strategy. Most research to date has focused on the use of SRC rather than SRF crops but the findings are likely to be transferable. The known purification functions (phytoremediation) of SRC and SRF have raised the possible use of these crops for final water treatment on farms (Jorgensen *et al.*, 2005). Results from the Water Renew study found SRC species of willow, poplar and eucalyptus to be efficient in removing nitrate, phosphorus and potassium, providing scope for forming an effective wastewater polishing system (Sugiura *et al.*, 2008). Applications of sewage sludge and nutrient-rich waste water have been shown to enhance biomass yields of SRC willow, while the nutrient demand for growth helped to reduce leaching and improve effluent quality (Perttu, 1993; Sims and Riddell-Black, 1998; Britt *et al.*, 2002; Hall 2003; Berndes *et al.*, 2004; Berndes, 2006; Borjesson and Berndes, 2006; Christersson and Verma, 2006). SRC has also been shown to be suitable for the application of landfill leachate, with survival rates of 97% or greater for willow, 78% for hazel and 63% for alder (ADAS, 2006). Less investigated is the ability of SRC or SRF buffers to remove pathogens such as faecal indicator organisms (FIO).

Treatment of contaminated land

When planted and managed as part of a controlled programme, trees and woodland can play an important role in the rehabilitation of derelict land, including landfill sites (Jones Jr., *et al.*, 2001; Pulford and Watson, 2003; French *et al.*, 2006; Strycharz and Newman, 2009). From a water perspective, a key benefit can be reducing the mobilisation and leakage of contaminants that have the potential to pollute both surface and groundwaters.

Trees can assist remediation in a number of ways: by helping to enrich the soil with organic matter, which is important for immobilising many contaminants (Hutchings, 2002); by providing a semi-permanent landcover, reducing the risk of soil disturbance and erosion (*ibid.*); by reducing surface runoff/groundwater recharge and thus the potential for leaching of contaminants to water (*ibid.*); and by the active uptake of contaminants and fixation in woody biomass (Dickinson *et al.*, 2009). There is also a role for planting woodland adjacent to contaminated land, which can help to reduce the offsite migration of contaminants by intercepting polluted runoff and by reducing wind erosion and trapping airborne contaminated soil.

SRF offers a number of advantages over conventional woodland for reducing the diffuse pollution risk from contaminated land. These include: the rapid establishment of a protective land cover; the high crop water use, which can greatly reduce leachate volumes; and the strong uptake of contaminants and removal/retention in biomass.

3. Water quantity

a) Threats to water resources

As noted in the introduction there are no measured hydrological data for land use change scenarios of SRF replacing alternative agricultural crops in the UK. Our understanding of impacts on water quantity, in particular the volume reaching rivers and groundwater, relies heavily on the known effect of SRC and to a lesser extent on models and experimental data from more conventional forestry.

Trees and forests are well known to generally use more water than shorter types of vegetation (Nisbet, 2005). This is mainly due to the interception of rainwater by their aerodynamically rougher canopies, but also to higher transpiration rates sustained by deeper rooting on drier sites. The resulting impact on water resources is an increasingly important issue for a wide range of stakeholders, particularly water resource managers and planners, as demands for water continue to rise and climate change threatens warmer, drier summers.

Research to date on the water use of energy crops has focused on the effects of SRC; although the water use of SRC will differ from SRF, the results give an indication of the likely impacts of the higher stocking densities used compared to conventional forestry. A major study of the hydrological effects of SRC poplar on farmland at two sites in southern England by Hall *et al.* (1996) confirmed the potentially high water use of these crops. While interception losses at 14% of annual rainfall were typical of conventional broadleaved woodland, transpiration rates were much higher when crops were well supplied with water, with values of around 500 mm per year for SRC poplar, compared to 350-390 mm for ash and beech woodland. Groundwater recharge volumes were estimated to be reduced on average by at least 50% compared to grass in the final year before cutting. Losses were less when averaged over a three or four year cutting cycle, with the water use of one-year-old shoots estimated to be only 55% of that of three-year-old shoots. They were also lower for crops in dry years due to the sensitivity of poplar to soil water stress, approaching values for grass.

Hall *et al.* (1996) concluded that the planting of SRC crops on any sizeable scale was probably best avoided in locations where water demand is expected to exceed available

supplies. Hall (2003) provides guidelines on suitable locations in England and Wales for planting SRC where the benefits for biomass production are expected to outweigh the hydrological impacts. Southeast England is most at risk, with the guidelines recommending that where the annual precipitation is <600 mm only a small proportion of a catchment should be planted, due to SRC using all of the effective precipitation. The only caveat is for sites with shallow soils, where the lack of water will severely restrict SRC water use, such that recharge volumes are likely to be similar to those for grass or arable crops; in such situations however SRC productivity is likely to be low and provide a poor economic return.

The above results are in line with those from studies of SRC poplar and willow in Scandinavia and elsewhere (Stephens *et al.*, 2001). Several water use models have been developed from field studies in Sweden, which suggest that annual evaporation from SRC exceeds Penman open water evaporation by between 5% and 40% (Grip *et al.* 1989; Persson and Jansson, 1989). In a simulation study, Grip *et al.* (1989) estimated the water use of a SRC willow stand with a production of 12 t ha⁻¹ yr⁻¹ to be 526 mm, which was 22% higher than the Penman open water evaporation rate of 430 mm for that location.

In their modelling study of SRC water use, Stephens *et al.* (2001) estimated that replacing wheat or grass with willow could reduce hydrologically effective rainfall by as much as 75-90%. They predicted that water losses would be greatest for isolated patches or thin strips of SRC, especially where well supplied with water, such as within the riparian or floodplain zone. The water use of such crops could be up to twice of those without access to groundwater, and exceed annual rainfall.

No studies have yet measured the water use of SRF in the field in the UK but evaporation losses are expected to be similar or greater than SRC (depending on species choice), and exceed those of conventional forests, due to faster growth rates and the greater proportion of time spent at maximum water uptake. Studies have shown tree water use to be directly related to growth rate and to decline in older aged stands (Vertessy *et al.*, 1995; Watson *et al.*, 1999; Xavier *et al.*, 2002; Almeida *et al.*, 2007).

Evaporation losses and therefore the impact on water resources will depend on many factors, the most important of which is likely to be species choice. The water use of SRF conifer crops is likely to be slightly greater than that of conventional stands and have the greatest impact on surface water runoff and/or groundwater recharge. Research in the UK and elsewhere shows that the effect of the high interception loss of conifer forest canopies, which in general equates to 25-45% of the annual rainfall, can range from a 15-20% reduction in water resources in the wetter uplands (Calder and Newson, 1979) to 70-100% in the drier lowlands for a completely forested catchment (Calder *et al.*, 2002).

Likewise, the water use of many SRF broadleaved crops is unlikely to differ greatly from conventional broadleaved woodland, although closer tree spacing and faster height growth could be expected to promote canopy closure and thus overall water use. Studies in the UK show the water use of broadleaves to be greatly influenced by species and soil/geology (Wullschleger, 1998; Roberts and Rosier 2005). For example, work by Green *et al.* (2006) on the water use of oak woodland on sandstone in Nottinghamshire found that groundwater recharge was 14-32% less compared to a grass ley. In contrast, Roberts *et al.* (2001) and Harding *et al.* (1992) showed that recharge was 13-25% greater under beech and ash woodland compared to grass on chalk at a site in Hampshire and 14% greater for ash than grassland on clay in Northamptonshire.

The potential impact on water resources is expected to be greatest for the fastest growing and most productive SRF species; these are likely to be non-natives. Most of the high yielding species suitable for SRF, such as *Eucalyptus nitens*, are known to have a high potential water use, partly due to deep rooting sustaining high transpiration rates (Lima, 1984; Honeysett *et al.*, 1992; Calder *et al.*, 1997). In a 6-year study of the water balance and growth of a *Eucalyptus grandis* hybrid plantation in Brazil, Almeida *et al.* (2007) showed annual crop water use to average 1092 mm compared to 1147 mm precipitation, leaving only 3% as runoff. Calder (1992) found that young Eucalyptus planted on 8 m deep soils appeared to be “mining” water as the roots penetrated greater depths, so that annual evapotranspiration exceeded rainfall by a significant margin (3,400 mm and 2,100 mm, respectively).

Rooting depths are likely to be much shallower (<2 m) in the UK but high water use could be sustained where there is access to groundwater. Greenwood *et al.* (1985) found that annual evapotranspiration from Eucalyptus at a site in Australia exceeded annual rainfall by a factor of four due to groundwater extraction. Eucalyptus has been used to “drain” marshland and to deliberately lower the water table where saline water has risen to the surface (Calder, 1992).

b) Potential benefits for flood risk management

As already noted, the potential high water use of SRF could be used to good effect in certain locations, such as on landfill sites to reduce the volume of leachate and thus the delivery of diffuse pollutants to watercourses or groundwater. Another, potentially greater, opportunity is to use SRF to help reduce downstream flood risk.

Nisbet and Thomas (2006) highlight three mechanisms whereby trees could help alleviate flooding; their greater water use, the higher infiltration rates of woodland soils, and the greater hydraulic roughness of floodplain and riparian woodland (although crops would have to be planted on a large scale to exert a significant effect at the catchment level for large floods). SRF has the potential to enhance all three compared to

conventional woodland, with the ability to increase the hydraulic roughness of the floodplain expected to offer the greatest benefit.

Work by Thomas and Nisbet (2006) showed that the increased hydraulic roughness associated with planting native floodplain woodland on floodplain grassland along a 2.2 km length section of the River Cary in Somerset could reduce water velocity by 50% and raise the flood level within the woodland by up to 270 mm for a 1 in 100 year flood. Temporary flood water storage increased by 71 % and the downstream progression of the flood peak was delayed by 140 minutes. These results were considered significant for reducing downstream flood risk by potentially desynchronising flood flows, lowering the flood peak and providing more time for issuing flood warnings. A second modelling study in the River Laver catchment at Ripon in North Yorkshire predicted that planting floodplain woodland at four sites totalling 40 ha in area (<1% of catchment) could delay the progression of a 1 in 100 year flood by around one hour. This had the potential to reduce the flood peak at Ripon by 1-2% by desynchronising the flood contribution from the adjacent tributary, the River Skell. A much greater reduction was expected with a larger planting area (Nisbet and Thomas, 2008). The authors highlight the need for care when selecting sites for planting to avoid the enhanced risk of flooding to upstream properties due to the backing-up of flood waters. In the case of the River Laver study, the backwater effect was predicted to extend a distance of between 130 and 330 m upstream of the woodland. Another issue is the potential threat posed by the blockage of downstream bridges and culverts by the wash-out of woody debris.

The greater planting density and faster growth of SRF could be expected to promote the development of hydraulic roughness and accentuate the delaying effect on flood flows. Another important factor is the high potential productivity of such crops on fertile floodplains, which would provide a more attractive option to landowners compared to conventional woodland planting. The perceived loss of land value and reduced income has proved to be a major constraint for landowners interested in planting floodplain woodland to date. However, care would be required to avoid sites where there was an issue over water supplies or maintaining low flows for freshwater ecology. Careful attention would also need to be given to SRF design in order to balance effectiveness in providing flood mitigation against visual impact, while harvesting operations would need to be correctly phased to maintain maximum site roughness and so provide continuous flood risk mitigation. The lack of dead wood associated with SRF crops would reduce the wash-out of woody debris and thus the risk of downstream blockage of river structures.

Well designed and managed SRF could have a particularly valuable role to play in helping to reduce surface runoff and pluvial flooding within urban areas. Planting as part of sustainable urban drainage systems could help to greatly reduce and delay local flood runoff, as well as provide other environmental and related benefits. The warmer and well

ventilated nature of the urban climate would promote tree interception and transpiration losses, accentuating the reduction in water runoff.

4. Anticipated impacts of climate change

Climate change projections of winters being 20% wetter across the UK and summers 20% drier in the south are expected to have wide ranging impacts on water, especially on the timing and volume of river flows and extent of groundwater recharge (Nisbet, 2002). The key water impacts are thought to be an increased frequency and severity of seasonal drought (particularly in the south) and floods, an increased risk of diffuse pollution, and greater thermal stress to freshwater life. From the preceding sections, it is clear that an expansion of SRF has the potential to both exacerbate and ameliorate these impacts. The largest threat is thought to be posed by the potentially high water use of fast growing SRF crops.

Calder *et al.* (2009) undertook a model evaluation of the impact of SRF on water resources at eight sites across the UK under present and future climate scenarios. They predicted that planting evergreen broadleaved species such as *E. nitens* and *E. gunni* or southern beech (*Nothofagus spp*) would have serious implications for water resources in areas receiving <800 mm rainfall, reducing the mean annual water yield by 2080 for a given area of SRF from a mean of 86 mm under grass to only 9 mm under these non-native species, based on a low emissions scenario. This was in sharp contrast to planting native ash as SRF, which was predicted to increase water yield by 2080 by a margin of 15 to 202% compared to grass. For all sites and for all climate scenarios the predicted ordering of the species in terms of increasing evaporation and reducing water yield remained the same: *Fraxinus excelsior*, grass, *E. gunnii*, *Nothofagus*, *E. nitens*. These results are based on limited data and partly drawn from analogue species but serve to highlight the possible major impact on water resources if fast growing, especially evergreen species were planted on any sizeable scale. However, they also highlight the potential scope to control this impact and perhaps even benefit water availability through appropriate species choice and site selection. Further measurements and model testing are necessary to cover a wider range of species and site types, and to demonstrate model suitability. Work is ongoing to check the water use of SRF crops of ash and sycamore at a site in east Scotland, while a study of Eucalyptus in south east England is planned to start in 2010.

It is important to note that critical gaps remain in our knowledge of the impact of climate change on tree water use, especially for trees grown under SRF systems, where very little experimental data is available and none for the UK. While climate change predictions of warmer, drier summers could potentially increase SRF water use and put further pressure on existing water resources, other related factors could limit or even

reduce the effect. A major uncertainty concerns the effect of rising CO₂ levels on tree water use efficiency. Eastham *et al.* (1990) found that the water use efficiency (ratio of biomass produced per unit volume of water evaporated) increased in densely planted biomass plantations and rising carbon dioxide concentrations may further enhance this ratio in the future. Stomatal conductance and water use are generally reduced at higher CO₂ (Broadmeadow and Randle, 2002), with reductions of 19-40% having been recorded in experimental studies for a range of species (Lodge *et al.*, 2001; Hungate *et al.*, 2002). Another issue is the impact – especially on deeper-rooted vegetation such as trees – of increasing soil water stress due to summer drying on evaporation rates, which is predicted to lead to a convergence in water yields from SRF and grass through time (Calder *et al.*, 2009).

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Impacts of Short Rotation Forestry on soil sustainability

Elena Vanguelova and Rona Pitman

Summary

This review evaluated the likely impacts of SRF on soil quality, including organic matter, nutrient capital, compaction, erosion, and soil biodiversity. It focuses on the tree species that are best suited to UK energy forestry.

Land available for SRF in the UK comes mainly from former agriculture, and is rich in base cations, nitrogen and phosphorus. Growing SRF crops for biomass may potentially lead to significant soil nutrient depletion and acidification over time. However, these effects may be species specific, as under *Salix* and *Populus* base cations have been seen substantially reduced, whilst under *Fraxinus*, *Tilia*, *Alnus*, *Betula* and *Nothofagus* soils have become less acidic. Litterfall quality and quantity, in addition to original soil status, play an important role in altering soil organic matter content and C sequestration under SRF. Soil C sequestration rate is likely to increase along a gradient from ash (& broadleaves) to spruce (& conifers). Conversion of agricultural land to SRF has potentially beneficial effects on soil carbon dynamics, with reported gains in soil C of up to 20%. Soil C sequestration by SRF is highest on arable soils previously having very low soil C content. Leaf litter inputs and tree rooting will enrich the low soil carbon levels, improving soil quality and biodiversity. The impact of SRF on the higher carbon stocks of grassland soils is less certain, although any reductions are likely to be outweighed by the carbon gain in woody biomass.

Management of SRF plays a significant role in the impacts on soil sustainability. Although the need for N fertilisation under SRF is much reduced compared to arable crops, there is the possibility of leaching from excessive applications to young tree stands. Changes to the carbon cycle from fertiliser use or excessive ground preparation can affect respiration from the soil, and result in extra production of CO₂.

SRF aids long term soil stability and physical soil erosion should be much reduced compared to annual cultivation, with improved infiltration from greater litter and deeper rooting depth. However, during site preparation and harvesting there may be compaction of soils through frequent traffic movements.

The effects of climate change on soil quality under SRF plantations are still uncertain. There may have to be very different regional responses to projected changes. SRF in the South East may have reduced productivity under increased moisture stress, resulting in lower carbon sequestration in the soil, whilst in the North and West soils under wetter and windy conditions may be more affected by increased erosion and nutrient leaching.

Important gaps in our knowledge are the long term impacts of continued SRF rotations on soil properties and functions. Research on the impacts of exotic species (*Eucalyptus* and *Nothofagus*) on soil quality is essential. There is an urgent need to validate and improve models by quantifying actual soil carbon sequestration rates under bioenergy crops.

1. Introduction

Intensive SRF plantations have raised concerns about environment, biodiversity, hydrological and landscape issues (LTS International, 2006). The resolution of the Kyoto Protocol to include effects of land use and land-use change in global carbon budgets, has put focus on C sequestration following afforestation of former arable and pasture land. Carbon is sequestered in the aggrading biomass of the new forests, but the question remains, to what extent the former arable soils will contribute as sinks for CO₂. The role of short-rotation woody crops in sustainable land use, such as providing wildlife habitat, reducing nitrate pollution in water, and as a potential carbon sink, was reviewed using data from several European countries (Börjesson, 1999). More recently, the potential effects of SRC on soil issues were described in detail by Baum *et al.* (2009).

Although the above benefits of biomass crops have been predicted, data concerning the potential effects of these crops on soil quality are not as widely available, especially changes by Short Rotation Forestry species with longer rotation lengths (Trinkaas, 1998, LTS Report, 2006). This issue is crucial to long-term sustainability and the economic success of biomass production, especially where biomass crops are alternatives on farmland (Lal, 1998). Although some studies and preliminary modelling provide estimates of C sequestration rates for SRF and SRC (Defra's project NF0418, 2004), the impact on soil carbon and nutrient status remain largely unknown, especially for SRF. Although the land for SRF is likely to be ex-agricultural and thus fairly rich in base cations, nitrogen and phosphorus, growing short rotation forestry for biomass may over time lead to significant soil nutrient depletion and soil acidification. There is a need to compare systems and the effects of different tree species and rotation lengths on C sequestration efficiency, soil nutrient capital and soil biodiversity as well as to assess the wider environmental issues associated with them.

In this Chapter, we evaluate the likely impacts of species of SRF and SRC on soil quality, including organic matter, nutrient capital, compaction, erosion, and soil biodiversity. This review focuses on the tree species that are best suited to UK energy forestry, including native species such as alder (*Alnus* sp), ash (*Fraxinus* sp), birch (*Betula* sp), and poplar hybrids (*Populus* sp), and non-native species such as eucalyptus, sycamore and southern beech (*Nothofagus* sp).

Although findings from regions throughout the world are included, particularly from areas of similar climatic regimes, the focus is primarily on recent research results from biomass crop production in the UK.

2. Impacts in absolute terms

Woody biomass crops could affect soil quality by causing changes in: (i) organic matter; (ii) soil acidity; (iii) the relative flux of nutrients and nutrient capacity; (iv) soil biodiversity; (v) erosion, especially during establishment; and (vi) soil compaction resulting from equipment movement during planting, maintenance, and harvest (Doran *et al.*, 1994; Paul *et al.*, 1997; Reeves, 1997; Lal *et al.*, 1998). These changes are affected by the biological activity of microfauna and macrofauna, thus SRF could also affect the soil biodiversity.

Short-term studies have demonstrated increases in surface soil organic-matter content, reduction in erosion and nutrient losses in surface runoff (Mann and Tolbert, 2000). Research to date shows promising short-term changes in soil quality due to short rotation forestry, but responses vary. The absolute impacts of SRF on soils are summarised into two sections:

- Direct soil effects of the crop: a. Organic matter; b. Acidity; c. Nutrient dynamics; and d. Biodiversity
- Effects of silvicultural and management operations, including the impacts on soil erosion and compaction.

Direct effects

a) Changes in organic matter and soil carbon sequestration potential

Organic carbon content is perhaps the most widely measured indicator of soil quality or potential soil productivity (Doran *et al.*, 1994; Paul *et al.*, 1997; Reeves, 1997; Emmett *et al.*, 2006) even though its effects on other soil characteristics is not always predictable. Crop systems that increase in soil organic carbon generally yield gradual, positive changes in structure, water-holding capacity, and the storage and availability of

nutrients - this in turn leads to increased abundance and diversity of soil biota, as well as increased resistance to compaction (Mann and Tolbert, 2000). Inputs of crop residues and their decomposition are the main factors determining the organic matter content of soils (Paul *et al.*, 1997). Minimum tillage, in particular, can increase the organic carbon content in soils of crop systems, especially in surface soil layers (Johnson *et al.*, 1995; Paul *et al.*, 1997; Baum *et al.*, 2009).

Leaf litter from broadleaved trees provides organic material that is generally quickly decomposed and incorporated into the upper soil horizon (Drift, 1961). Litter N and lignin content, C/N ratio, leaf area and Ca content are some of the important litter qualities strongly related to litter decomposition rate (Cornelissen, 1996; Wedderburn and Carter, 1999; Peterken, 2001; Reich *et al.*, 2005; Hobbie *et al.*, 2006; Vesterdal *et al.*, 2008). In general, broadleaved and deciduous exotics, such as *Nothofagus* spp., have a similar effect on soils to native broadleaves (Peterken, 2001). The litter of non-deciduous broadleaves such as *Eucalyptus* species takes longer to decompose (Cornelissen, 1996), but will nevertheless decompose more quickly than that of native conifers (Wedderburn and Carter, 1999). Additional important factors affecting the rate of leaf decomposition are soil pH and soil moisture, with moist, base-rich soils providing conditions for the quickest rate of decomposition (Witkamp and Drift, 1961; Pereira *et al.*, 1998; Hunter *et al.*, 2003; Reich *et al.*, 2005). Due to the high capacity of *Eucalyptus* species for water uptake and interception it is likely that the soils beneath will be relatively dry, which may slow the decomposition rate. In general, the litter of deciduous broadleaved trees has a beneficial effect on soil chemistry and structure. There is very little research from UK and mainland Europe on *Nothofagus* or *Eucalyptus* litter and soil chemistry effects (LTS International, 2006). Litterfall quality parameters - such as N content, Specific Leaf Area (a surrogate for decomposition rate), base cation content and carbon content - and their relation with litter decomposition rate of candidate SRF species are provided in Table 1.

Table 1. Estimated/measured litter parameters of different tree species used in SRF in the UK.

Species	Litter N (%)	Litter SLA (g/cm ³)	Litter BC content ¹	Litter C %	Rate of decomposition	References
Ash	1.24 -2.20	180-300	3.83	31.1	Rapid	Cornelissen, 1996
Alder		210			Rapid	Cornelissen, 1996
Sycamore	0.94	(213 fresh) ~250 litter	3.03	46.2	Rapid	Hobbie et al, 2006; Cornelissen, 1996; Pugh and Buckley 1971
Hazel	1.34	275-310	2.6	n/a	Rapid	FR
Hornbeam	1.1	210	1.46	46.9	Intermediate-rapid	Hobbie et al., 2006
Birch	1.0-1.4	170-320	1.31-1.41	47.8-52.8	Intermediate	Cornelissen, 1996; Hobbie et al, 2006
Chestnut	0.98-1.30	~150-200	1.19-1.28	49.7-50.7	Intermediate	FR
S.beech ~ N obliqua	0.6		~0.97		Intermediate	Wigston,1990; Adams & Attiwill, 1991
Willow (S. alba, S. fragilis)		149/160			Intermediate	FR; Cornelissen, 1996; Withington et al,2006
Poplar (P. trichocarpa), P. tremula/P.nigra		(70-170 fresh) 149-160			Intermediate	FR; Cornelissen, 1996; Withington et al,2006
Oak	1.0-1.38	165-190	1.83-1.95	36.6-51.1	Intermediate -slow	FR; Hobbie et al, 2006
Eucalyptus (E nitens)		61-66			Slow	FR; Wedderburn and Carter 1999; Lopez et al., 2001
Eucalyptus (E gunnii)		57-67			Slow	FR; AFOCEL, France; Adams and Attiwill 1991

¹ BC- base cation Ca+K+Mg% as defined by Cornelissen and Thompson, 1997

The quantity of litter being added influences soil organic matter and can vary over time and with forest species, age and planting density. Litterfall biomass (including leaves, branches, cones, frass) from the UK Intensive Level II sites is between 2.7 and 7.1 t ha⁻¹ y⁻¹ for oak, 3.0 - 5.3 t ha⁻¹ y⁻¹ for beech, 2.9 - 6.3 t ha⁻¹ y⁻¹ for Scots pine and between 2.7 and 5.8 t ha⁻¹ y⁻¹ for Sitka spruce. Difference in climate and deposition could influence the amount of litterfall. For example, total annual litterfall of pine is 3.8 t ha⁻¹ y⁻¹ and of beech is 2.9 t ha⁻¹ y⁻¹ in low N deposition areas compared to 8 t ha⁻¹ y⁻¹ and 3.9 t ha⁻¹ y⁻¹, respectively in a high N deposition area (Vanguelova and Pitman, 2009).

Davis and Trettin, (2006) have reported litterfall in a short-rotation plane (*P. occidentalis* L.) plantation increasing from 0.28 t ha⁻¹ in the first year to 6.11 t ha⁻¹ in the fifth year. This compared with a sycamore plantation in which litterfall started at 0.4 in the first year and reached 7.77 t ha⁻¹ in the fifth and sweetgum litterfall was only 0.06 to 1.85 t ha⁻¹ over the same 5 years.

A simple model using a carbon mass balance approach to predict soil carbon sequestration has been developed by Grogan and Matthews (2002) (DEFRA, 2004). The model is site-specific and calibrated to soil carbon data from a natural woodland regeneration site in the UK. The conclusions from the model outputs were that there is potential for significant soil carbon sequestration in SRC plantations in the U.K. In this very preliminary analysis, which necessitated several major assumptions, soil carbon sequestration rate under willow was estimated to 50 cm depth as 0.5 t C ha⁻¹ y⁻¹ in comparison to 0.41 t ha⁻¹ y⁻¹ under natural regenerated woodland. The model identifies the following factors as being major controls on rates and amounts of soil carbon sequestration under coppiced willow:

- carbon inputs (net primary production)
- decomposition rates of the major soil carbon pools
- initial soil carbon content which is inverse related to soil carbon sequestration
- crop/plantation management practice
- depth of soil being influenced by the bioenergy crop.

Carbon sequestration is most likely on soils whose carbon content has been depleted to relatively low levels due to previous management practices. Carbon sequestration in soil is likely to occur until the climatically-controlled equilibrium point between soil carbon inputs and outputs is reached (DEFRA, 2004).

Soil carbon sequestration rate of oak (*Quercus robur* L.) and Norway spruce (*Picea abies* (L.) Karst.) stands following afforestation of former arable land in Denmark were 0.07 t C ha⁻¹ y⁻¹ and 0.31 t C ha⁻¹ y⁻¹ respectively in the forest floors over 29 years (Vesterdal *et al.*, 2002). These values compared to soil C sequestration rate of 0.41 t C ha⁻¹ y⁻¹ in an adjacent 200-year-old mixed deciduous plantation. In addition to the forest floor, sampling included the three layers of the mineral soil to a depth of 25 cm. Over the 29

years following afforestation, there were no differences between the two tree species in C concentration and storage of the three mineral soil layers. Carbon concentration and storage increased in the upper 5 cm of the mineral soil but decreased in the 5–15 and 15–25 cm mineral soil layers with increasing stand age. Thus the soil C store appeared to be undergoing redistribution following afforestation, and mineral soil C stores in 0–25 cm tending to decrease. Together with the C sequestration of forest floors however, this led to fairly similar total soil C stores of around 65 t ha^{-1} across the 29-year chronosequence. Within this short time span, C sequestration mainly occurred in the biomass of trees while soil C accumulation took much longer (e.g. stores were clearly higher in the 200-year-old plantation 81 Mg C ha^{-1}). The ongoing redistribution of mineral soil C in the young stands and the higher soil C contents in the 200-year-old afforested stand suggest that nutrient-rich afforestation soils may become greater sinks for C in long term (Vesterdal *et al.*, 2002).

Soil carbon content of the forest floor increased in order ash=lime=maple<oak=beech<spruce in a common garden experiment with 6 sites (Vesterdal *et al.*, 2008). Two sites were planted on agricultural land 20 years ago, and four planted on old beech forest land - all on free draining glacial sandy loams, with one exception on Aeolian soils. The reported differences were attributed not to soil type, but to large differences in turnover rates. Soil C and N sequestration is therefore different under each species. Species such as ash, lime and oak sequestered carbon at depth in the profile, whereas beech and spruce sequestered more in the topsoil and forest floor.

A pronounced increase of soil organic matter content was also evident under a poplar plantation on former arable soil studied by Moscatelli *et al.*, (2008). Other studies also provide some support for carbon accumulation under SRF (Grigal and Berguson, 1998; Guo and Gifford, 2002) which are explained in detail in Section 3 below and shown in Table 2.

Table 2. Changes of soil carbon under SRC and SRF plantations.

SRC/SRF/ species planted	Previous land use	Soil depth (cm)	Time span (years)	Soil C (kg C ha ⁻¹ y ⁻¹ or %)	Reference
SRC Poplar (USA)	Agriculture	100	12-18	1630 (kg C ha ⁻¹)	Hansen, 1993
SRC Poplar (USA)	Agriculture	0-25; 0-100	6-15	No change; no change	Grigal and Berguson, 1998
SRC Willow (USA)	Grass/scrub land	0-60	4	No change; no change	Ulzen-Appiah, Briggs et al, 2000
Mixed coppice of (Germany)	Agriculture	0-10	7-9	100-555 (kg C ha ⁻¹) gain	Jug, Makeschin et al, 1999
Poplar, Aspen and Willow	Agriculture	10-30	7-9	0-555 (kg C ha ⁻¹) loss	Jug, Makeschin et al, 1999
Oak plantation (Denmark)	pasture	0-25 mineral only		~ 420 (kg C ha ⁻¹) loss	Vesterdal et al, 2002
Oak plantation (Denmark)	pasture	' min+ forest floor		~ 333 (kg C ha ⁻¹) loss	Vesterdal et al, 2002
Plantation	Pasture	meta analysis		10% loss	Guo and Gifford, 2002
Plantation	Native forest	meta analysis		13% loss	Guo and Gifford, 2002
Plantation	Arable	meta analysis		18% gain	Guo and Gifford, 2002
Secondary forest	Arable	meta analysis		53% gain	Guo and Gifford, 2002
Eucalyptus nitens (Tasmania, Australia)	Pasture	30	0-10	~ 200 (kg C ha ⁻¹) loss(-1.99%pa)	Paul et al 2003 ¹
Eucalyptus nitens (Tasmania, Australia)	Pasture	30	10-40	~30 (kg C ha ⁻¹) inc. (+0.82%pa)	Paul et al 2003 ¹
Eucalyptus globulus (Victoria, Australia)	Pasture	30	0-10	~ 220 (kg C ha ⁻¹) loss(-2.08%pa)	Paul et al 2003
Eucalyptus globulus (Victoria, Australia)	Pasture	30	10-40	~100 (kg C ha ⁻¹) inc (+0.39%pa)	Paul et al 2003
Eucalyptus globulus (SW Australia)	Pasture	30	0-10	~20 (kg C ha ⁻¹) loss(-0.96%pa)	Paul et al 2003
Eucalyptus globulus (SW Australia)	Pasture	30	10-40	~200 (kg C ha ⁻¹) inc (+1.80%pa)	Paul et al 2003

¹ case studies with partly modelled start values

Resumé: The role of foliar litter fall in altering organic matter content and carbon sequestration in soils under SRF depends on both its quality and the quantity added year on year. However, the initial status of the soil to which these additions are made is also very important, and long term effects seem to be species specific. In relation to C sequestration, the rate is low under SRF willows compared to natural woodland, and is likely to increase along a gradient from ash (& broadleaves) to spruce (& conifers). The time through which the woodland cover has been present is also an important factor to the status of the soils beneath.

b) Changes in soil acidity and base cation content

Most planting of SRF in the UK is likely to be on ex-agricultural land which is rich in base cations, nitrogen and phosphorus. However, over time, growing SRF may lead to significant soil nutrient depletion and soil acidification. Short rotation afforestation with poplars and willows on former arable soils in Germany decreased soil pH in topsoil by up to 0.5 units and Cation Exchange Capacity (CEC) by as much as 15 % at one site after 10 years (Jug *et al.*, 1999). Soil Ca saturation decreased but Mg increased at all sites over seven years of planting combined with fertilisation treatments (Jug *et al.* 1999). Aliksson and Olsson (1995) also report, from Swedish short rotation forestry plantations experiments on former agricultural sites, a decrease of soil pH and lower base saturation with time. Binkley and Valentine (1991) even found a decrease of base saturation from 20 up to 80% in A- and B-horizons after afforestation of arable sites. In contrast, Grandall (1994) reports for former agricultural sites a slight increase of soil pH after tree establishment of willows but a drastic decrease under alder plantations. Declines of 0.5 – 2.0 pH units have been reported by Miles (1985) from studies by Ugolini (1968) and Franklin *et al.* (1968). This is probably due to internal acidification via N fixation, nitrification and nitrate leaching and the coincident losses of base cations (van Miegroet and Cole, 1984; 1985). Other species have been shown to raise the soil pH, for example in a study by Muys *et al.* (1992), pH rose under *Alnus*, *Prunus*, *Fraxinus* and *Tilia* after 20 years by 0.2-1.1 units compared to the original meadow soil at 5.15.

Anderson (1987) resampled the soils in plots established in nursery trials 22 years earlier by Ovington, (1956) and found slight increase in pH under *Nothofagus obliqua*. In studies of birch effects on moorland soils. Miles (1985) showed pH to increase from 3.8 to 4.9 over 20+ years and Skeffington (1983) found soil pH on recolonised lowland heath to have values 0.1-0.2 units higher than under Scots pine. Miles (1985) suggested that this is the result of increases in exchangeable calcium from litter deposition and the rate of organic matter decomposition. Pitman (pers. comm.) has found increases of ~0.5 units after 5 years of birch colonisation of a lowland clay soil site previously under western hemlock in South East England.

Litter Ca concentration appears to be a key driver in long term changes in soils properties, especially those related to acidity and cation biogeochemistry, but also those

related to C and N contents and ratios in forest floor, humus and mineral soils. For example, *Tilia cordata*, *Betula pendula*, *Acer pseudoplatanus* and *Fagus sylvatica* litter increased soil pH and base saturation the most in the organic and top mineral soils after 30 years of growth in a common garden experiment on ex-pine forest land compared with coniferous species (Reich *et al.*, 2005).

Resumé During the growth of SRF, pH is likely to be reduced, particularly if the land use change is from arable to woodland. However, this effect may be species specific - under *Salix* and *Populus*, substantial base cation reductions have been seen, but under *Fraxinus*, *Tilia*, *Alnus*, *Betula* and *Nothofagus* soils have become less acidic through time.

c) Changes in soil nutrient dynamics and capacity

The long-term sustainability of soils in short-rotation biomass production is influenced by the relative balance between the removal of nutrients by harvesting and the nutrients added by fertilisation. Nutrients can leach from the soil or move with eroded soil particles. Harvesting also can remove large amounts of the total nutrients (Mann *et al.*, 1988; Ranney and Mann, 1994; Heilman and Norby, 1998). Macronutrients have been studied fairly extensively in biomass crops (Manna and Tolbert, 2000). Research in short-rotation systems of aspen, hybrid poplar, sycamore, willow, and alder have demonstrated that the amount of nutrients removed in harvested wood is approximately proportional to the accumulation of biomass without leaves over the time of each rotation (Perala, 1979; Korsmo, 1982; Börjesson, 1999). The amount of nutrients removed through harvesting is somewhat less in biomass crops than in agricultural crops (Korsmo, 1982; Ranney and Mann, 1994). Unlike many conventional agricultural crops, large quantities of organic matter and nutrients can be recycled from the annual litter fall of woody biomass crops, especially after the first 2 or 3 years following planting. In addition, harvesting woody biomass crops after leaf fall minimizes the loss of nutrients (Heilman and Norby, 1998). Similarly, harvesting in the fall after nutrients have translocated to the roots at the end of the growing season may conserve a large proportion of nutrients on site. For example, It has been suggested that Sweet chestnut harvesting for biomass should leave behind the leaf and top shoots to help maintain soil fertility into the following cycle (Ranger and Belgrand, 1996).

In a common garden experiment in Norway with 6 sites, 2 of which were planted on agricultural land 20 years ago, and 4 on old beech forest land (Vesterdal *et al.*, 2008) soil nitrogen content of forest floor increased from ash=lime=maple<oak=beech<spruce. In Estonia The concentrations of total N in the humus layer was higher under silver birch plantation compared with aspen, both planted on ex-agricultural land (Soo *et al.*, 2009). The forest floors under birch and spruce were more active than that under pine, having higher respiration and net N mineralization rates, and higher microbial carbon and microbial nitrogen values than pine forest floor (Kanerva and Smolander, 2007). Differences between tree species were smaller in the

Humus (H) layer than in the Litter (L) and Fermenting (F) soil layers. In the reassessment of soils under Ovington plantations in West Tofts (E. Anglia), Anderson (1987) found a rise in soil N under birch over a 22 year period, accompanied by a rise in soil K and Ca, but decline in P. Under *Nothofagus obliqua* at Bedgebury, in SE England, there was a decline in soil N and Ca but rises in soil K and P over the same period (Howard and Howard, 1984).

Higher P and K levels within the humus layer under birch compared to hybrid aspen were found in 7 years-old plantation on ex-arable unfertilised land in Estonia (Soo *et al.*, 2009). The density of the planted trees was twice as high in the birch (nearly 2000 tree ha⁻¹) compared to hybrid aspen (1000 tree ha⁻¹), which produced more root and leaf litter than the studied hybrid aspen plantation, which in turn affected the nutrient concentrations in the humus layer. According to several studies, the soil microbial activity and the humus rate of net nitrogen mineralisation have been found to be high in birch stands compared to coniferous or abandoned grasslands (Smolander *et al.*, 2005; Kanerva and Smolander, 2007; Uri *et al.*, 2008). However, the concentration of total N in the humus layer of the studied former field soils could also have been affected by different fertilisation practices during the previous agricultural land use. Higher N in humus layer corresponded into higher Ellenberg values of the understorey vegetation under the birch compared with the hybrid aspen (Soo *et al.*, 2009). Denser birch plantations start to effect nutrient concentrations in the humus soil layer, and consequently the nutrient status of the ground vegetation sooner than sparser hybrid aspen plantations. Such a difference is likely to become less pronounced over the course of time, as faster growing hybrid aspen catches up with birch in terms of litter quality (Soo *et al.*, 2009).

The different demand for N uptake and the different N levels in tree litter will impact the litter quality and N dynamics in the soil. For example, Eucalyptus (*E. globulus*) leaves caused strong soil inorganic N immobilisation (-7 mg N g⁻¹ residue - C) compared to legume species. Legume leaves immobilised the N first and then remineralised the N later on in incubation experiments (Adams and Attiwill, 1990; Corbeels *et al.*, 2003). These consistent differences in dynamics of N immobilisation/mineralisation between eucalypt leaves and legume residues related to two different soils (Phodic Ferralsol or red earth and Haplic Podzol or grey sand) most likely resulting from the "lower" quality of eucalypt leaves (higher C:N ratio and higher lignin and soluble polyphenol concentration) compared to legume residues (Corbeels *et al.*, 2003) (Table 1). However, in a study of young *E. nitens* in Tasmania, Moroni *et al.* (2002) found that mineralisation rates under plantings on similar soils were highest on ex-pasture sites, by a factor of two compared to ex-forestry uncultivated sites (13-188 kg N ha⁻¹ yr⁻¹). Most N was nitrified but then leached from the soils so that accumulation was limited and N remaining in the surface soil at the end of winter was only between 2-17 kg N⁻¹ ha⁻¹. *E. nitens* has been shown to favour N uptake through its roots in the form of NH₄ rather than NO₃-N from

soils (Adams and Attiwill, 1986; Garnet and Smethurst, 1999), which was suggested to result from *E. nitens* adaptation to growth in cooler, acidic soils compared to other Eucalypt species. Studies by Moroni and Smethurst (2003) over a two year period under *E. nitens* of 3 and 10 years of age, showed that the N flux input from the litterfall represented between 91-98% of all N flux in the plantation system – this represented a slightly higher proportion than the return of litter in native forests (66-90%). Adams and Attiwill (1990) reported that *E. nitens* litterfall was the dominant transfer mechanism of N, P and Ca.

A comparison of the flux of energy and nutrients from a mineral forest soil in which paper birch (*Betula papyrifera*) seedlings were grown over a 22-week growing period with the same soils under five other tree species have been made by Bradley and Fyles (1995). Soil basal respiration rate, metabolic quotient, soil available C, and the affinity of soil micro-organisms for substrate-C left in the soil after harvest all increased significantly, in soils treated with birch root systems. Amounts of rhizosphere activity, were one order of magnitude higher under the birch. Plant uptake of soil-N during the growing period was high while the soil mineral-N pool was low in birch relative to those of other species, suggesting that birch competed well against soil micro-organisms for available mineral-N. Anaerobic N mineralization rates were significantly higher while the degree of nutritional limitation of the microbial biomass was significantly lower in birch-treated soils. These results suggest that high amounts of root labile C compounds in conjunction with rapid mineral-N uptake by birch roots can stimulate microbial communities to acquire nutrients from the native soil.

In a chronosequences study of 2 -19 year old stands of established chestnut coppice in South West France, good forest production needed high soil mineral status (e.g. annual uptake of N between 46 and 84 kg N ha⁻¹), of which 50% returned to the soil through litterfall biomass. The other 50% was provided by translocation in the system, particularly N, P and K. Large losses of K occurred through canopy leaching, and overall decline in soil Mg and Ca as rotation went through to 20 years was evident (Ranger and Belgrand, 1996). There was also a good range of woody and herbaceous species beneath the younger stands (as in oak stands) declining with age as light declined, so that the decline in soil base cations was mainly due to chestnut demand and uptake. The maximum canopy development of the chestnut was at 6-7 years, followed by a static period but the maximum standing biomass was achieved at 15 years of age, followed by die-back (Ranger and Belgrand, 1996). The recommendations were to leave behind finer parts of trunks at harvesting to maintain some initial fertility of the soil for regrowth, and create as little soil disturbance as possible (should replanting be necessary) to minimise nitrification losses in the soil beneath.

Another consequence of afforestation with plantations is a significant export of nutrients from the site such as N, P and K by the stand harvesting following short rotation lengths

of 10-15 years or less. For example, the amount of nitrogen accumulated in the above-ground leafless biomass in 7-yr-old commercial hybrid aspen plantations established on abandoned agricultural land in Estonia varied between 14.4 and 48.5 kg ha⁻¹. The amount of phosphorus accumulated was between 1.7 and 5.9 kg ha⁻¹, and the amount of potassium, between 6.5 and 21.9 kg ha⁻¹. The removal of major mineral nutrients from this site with the removal of woody biomass in 7-yr-old plantations would be relatively small, constituting 0.5–3.4% of the nutrient pool in the humus layer of the previously fertilised field agricultural soils (Tullus *et al.*, 2009). In comparison, total N export via harvested stem biomass of poplar and willow could be up to 250 kg ha⁻¹ after 10 years of growth on ex arable land, of which 140 kg ha⁻¹ only would be stored in root biomass (Jug *et al.*, 1999). Cobb *et al.*, (2008) found that in sycamore, the fastest growing stands contained more than 200 kg N ha⁻¹ (excluding belowground demand and previous foliar production sequestered in the forest floor), indicating a high potential N demand and uptake in this species. Similar results were found by Stoneman *et al.* (1996) for thinned and fertilised *Eucalyptus marginata*. O'Connell and Glove (1999) estimated a loss of more than 500 kg N ha⁻¹ through exports in harvested logs and burning of harvest residues of a highly productive 8 year-old *E. globulus* stand. These results raise concern that, in the long term, large nutrient exports at harvest, combined with reduced external inputs from fertilisers and biological N fixation under plantation, will progressively decrease site fertility and productivity. Field studies have shown that productivity of eucalypt plantations on ex-pasture land in south-western Australia is likely to decline rapidly because of decreasing soil N availability (O'Connell *et al.*, 2003; Corbeels *et al.*, 2005).

Resumé. In comparison with agricultural crops, SRF aids long term soil sustainability. In particular, this comes from less frequent cultivation, litterfall accumulation, and less application of fertilisers. However, some species such as birch are good at mobilising soil nutrients, and particularly for N, accumulation occurs in the above ground biomass. Losses of other nutrients (P or Mg) may be small, but N loss at harvesting can lead to the eventual progressive decline of the site fertility and productivity. The form of the N uptake may be preferentially as NO₃-N (most broadleaves) or NH₄-N (eucalyptus and some conifers) at the root level, and the litterfall may be helping to maintain a fast turnover of N in the system.

d) Impacts of SRF and SRC on soil biodiversity

Soil biodiversity is directly linked with the soil quality including soil texture, nutrient regime, organic matter, moisture, acidity and Green House Gases (GHG). In addition, as tree species differ in the amount and quality of litter produced (see Table 1), these differences may directly or indirectly affect the associated soil invertebrates.

Investigations have revealed that above-ground vegetation can have a great influence on the below-ground earthworm population and diversity (Muys *et al.*, 1992; Zou, 1993;

Sarlo, 2006). Hendriksen (1990) reported that the number of litter feeding earthworms was negatively correlated with the C:N ratio and final polyphenol concentration of the litter. Tian *et al.* (1993) also revealed that earthworm populations were negatively correlated with the ratio of lignin to nitrogen of the plant residue. Further, Muys *et al.* (1992) have showed that earthworm communities varied considerably between grassland sites that were afforested with different tree species depending on the quality and quantity of the litter produced. This research also revealed that earthworm biomass reduced under oak (*Quercus palustris*) due to poor quality of the litter and acidification of the soil. In the study by Muys *et al.* (1992) all young plantations of *Alnus*, *Prunus*, *Tilia* and *Fraxinus* have significantly higher earthworm biomass compared to old forest stands which were still comparable with the original meadow. Most of these young stands developed mull humus types and supported deep-burrowing earthworms, while the *Quercus* older forest stand (planted on ex-forest land) with moder humus type was very poor in earthworms and all the species found were typically acidotolerant species. Earthworm biomass was strongly positively related with litter Ca concentration under 14 broadleaves and coniferous species from a common garden experiment (Reich *et al.*, 2005). Under the Eucalyptus stands investigated so far in the UK, earthworm abundance and activity was very evident under both *E. nitens* and *E. gunnii* at the northern N rich site (Alcan, near Newcastle), but only under *E. gunnii* in the Daneshill site at Retford (on a restored soil). Earthworms have not been detected at the Rogate site in South of England, under *E. nitens* on acid sandy soils.

Whereas earthworm populations are likely to increase with increasing litter, numbers of Carabid beetles are likely to decrease. However, balancing positive impacts due to the non-tillage management might increase both abundance of earthworms and an increased diversity of Carabids (Baum *et al.*, 2009).

In a study under willow (*Salix* sp.) on ex-agricultural land (Minor *et al.*, 2004) measures of abundance, species diversity and community structure of two groups of mites (Oribatida and Gamasida) suggested that soil cultivation had negative effects on their abundance and diversity during the first year of establishment. However, following the initial disturbances, the abundance and diversity of soil mites increased significantly over time.

In contrast to other arable crops SRF species can be colonized by ectomycorrhizal fungi, and consequently positive changes in soil microbial diversity and activity can be achieved in the soils beneath (Baum *et al.* 2009). Ectomycorrhizal fungi are likely to increase under SRF compared to cultivated soils (Rooney *et al.*, 2009), and have been seen in the mineral soils under both willows (*S. viminalis* and *S. dasyclados*) and poplars. Nine varieties of arbuscular mycorrhizal fungi were recorded in Estonia, being highest in soils with higher N and organic matter, and lowest in soils with low pH, P and K concentrations. (Puttsepp *et al.*, 2004). In South Germany, ectomycorrhizal formations

were found on both the poplar clones *P. trichocarpa* and *P. tremula x tremuloides*, but highest on the latter, being present on between 40-80% root tips. These populations were significantly adversely affected by fertilisation of both N and P, particularly in the hybrid poplar, so minimal use of fertilisation was recommended. Ectomycorrhizal communities have also been shown to be important in Eucalyptus stands. Pampolina *et al.* (2002) calculated that the fungal community held up to 10% of the combined nutrients of the tree (*E. globulus*), and that fertiliser addition, particularly P, decreased the production of the basiocarps. In the UK, site inspection revealed a very strong association of the ectomycorrhizal fungus *Laccaria laccata* with both 5-year-old stands of *E. gunnii* and *E. nitens* at the Daneshill site, near Repton, moderate occurrence with *E. gunnii* at the Alcan site near Newcastle, and sporadic occurrence with *E. nitens* at Rogate (Sussex).

Resumé The effects of SRF on earthworm populations appears most closely related to soil C/N ratio, the phenol content of the litterfall and the soil status in terms of mull or moder humus type development. Earthworms role in organic mixing is vital, but where worms might increase under heavier litterfall, Carabid beetle populations might well fall. The beneficial presence of ectomycorrhizal fungi has been demonstrated under poplar, willow, birch and eucalyptus species, present in much higher numbers than in similar arable soils.

Effects of silvicultural and management operations on soils

The complex issue of soil carbon emissions and dynamics is often omitted from calculations of the potential carbon savings offered by biomass systems. The following can be detrimental to the potential for biomass systems to reduce carbon emissions from soils in a 'whole life cycle' perspective:

- 1) Cultivating soils with a high carbon content for the production of energy crops and
- 2) Using intensive crop management regimes including inorganic fertilisers, irrigation and pesticides (Tubby, 2007).

a) Fertilisation

SRF response to fertilisation is generally positive as far as above-ground biomass is concerned, whereas the response of belowground pools and processes, with altered root growth and turnover is less certain. Fertilisation will influence microbial activity, soil chemical properties such as pH, cation exchange capacity (CEC) and soil C dynamics (Lee and Jose, 2003).

As in many agricultural production systems, where nutrients are made available at a steady high level with sufficient water, growth is maximized. However, such treatments

have potential for contamination of water by nitrate leaching or by surface runoff of nitrogen fertilizer (Börjesson, 1999). This is highest in soils with high nitrification rates,; though the timing of fertiliser applications can be managed to maximise productivity and minimize nutrient leaching (Heilman and Norby, 1998; Perttu, 1998; Börjesson, 1999). The amount of nutrient use changes seasonally within each year but also over time as perennial crops become established, unlike annual agricultural crops that attain similar rates of nutrient use every year. For woody crops, this balance can be approached by annually increasing the amounts of fertilizer that are applied or by delaying fertilization until the third year of growth (Heilman and Norby, 1998). Nutrient losses from biomass crops could be further reduced by matching nutrient additions with crop growth. In studies of *Eucalyptus nitens* plantations in Tasmania, Smethurst *et al.* (2004) found growth response to fertilisation of saplings on ex-forest soils for 1-2 years after application. However, higher/more frequent additions did not increase productivity, as losses of N in leaching increased from the sites (Smethurst *et al.*, 2003).

Studies at three locations in the southeastern US have shown that offsite nutrient transport from biomass crop plantings is similar to that from conventional crops during the year of establishment, but generally decreases in subsequent years (Thornton *et al.*, 1998; Tolbert and Wright, 1998). After the first year, concentrations of nitrate-nitrogen and ammonium-nitrogen in runoff and groundwater were lower from sites with woody biomass crops than from sites with either corn or cotton (*Gossypium hirsutum*). Elevated nitrate-nitrogen concentrations occurred in runoff from rain events following fertilizer applications to all crops. However, measured values were otherwise below 10 mg L⁻¹ for woody biomass crops and were always lower than measured values for row crops. Mineralization of nitrogen in leaf litter may also be contributing to nitrate export. Phosphorus concentrations in runoff were generally lower from sites with woody biomass crops than from sites with corn or switchgrass. After the initial year of establishment, losses of nutrients other than phosphorus from the switchgrass plantings were lower than those from conventional crops or from trees grown without a cover crop. In these experiments, fertilizer was applied every year.

The application of nitrogenous fertiliser to SRF crops during the first year of establishment has been found to increase nitrate run-off without benefiting tree growth, and should therefore be avoided (Heilman and Norby, 1998). In addition SRF requires less input of fertilisers compared with annual crops. If nitrogen is applied later, it is important to consider both the N adsorbing potential of the soil and the rooting depth of the trees when judging the quantity and time of application. Since water-use by SRF trees is greater than that of annual crops, nutrient export from fertiliser applications from an established SRF should be reduced. Some sites available for planting SRF crops may be in nitrate vulnerable zones, which now cover a considerable area of central and eastern England. In such zones, particular care must be taken to ensure input of N does

not exceed uptake by the trees. Changes in N input can affect soil nutrient status, root exudation, leaf litter chemistry and plant microbial competition for nutrients.

Soil C fluxes can also be affected by fertilisation practices. Increases of soil respiration after nitrogen addition to forest soils have been demonstrated by Gallardo and Schlesinger (1994), whereas significantly lower rates of CO₂ production have been reported in fertilised forest soils by Smolander *et al.* (2005) and Söderström *et al.* (1983). Fertilisation did not modify soil capacity to accumulate organic matter in the medium term under Poplar plantations in Italy, with contrasting effects dependent on short term soil dynamics and on the type of fertiliser applied (Moscatelli *et al.*, 2008). In addition, contrasting effects due to different plant cover have been shown by Lee and Jose (2003). They found decreases of soil respiration under *Populus deltoides* but no significant change under *Pinus taeda*. Chemical analysis of foliar composition gives a clear indication of whether the tree crop would benefit from fertiliser application, and if so, what proportions of N, P and K. Such applications would need careful economic evaluation as well as consideration of their environmental impact (LTS International, 2006).

Resumé The need for N fertilisation under SRF is much reduced compared to arable or row crops, but the possibility of leaching from excessive applications to young stands is high. Fertilisation should be used only at maximum periods of growth, such as in year 2 or 3 of a willow stand, once establishment has been achieved, and other soil physical preparation treatments are complete. Changes to the carbon cycle from fertiliser use or excessive ground preparation can affect respiration from the soil, and result in extra production of CO₂.

b) Choice of species and provenance

All the tree species recommended for biomass production intercept (and subsequently lose to the system via evaporation) more rainfall than either arable crops or swards of grass, but less than coniferous plantations (Hall, 2003). Trees have the ability to increase the scavenging of atmospheric pollution relative to low growing vegetation. In general, coniferous vegetation will scavenge more atmospheric sea-salt and pollutants than deciduous vegetation. Broadleaf canopy intercepts and enriches the water passing through the canopy more with base cations than anions, and can buffer the acidity in the rainfall - as shown in a study of lowland ash in Northamptonshire (Neil, 2002).

Although there are some differences between tree species and provenances with regard their water use efficiency, in general, quicker biomass accumulation will be positively correlated with greater water use. Interception and evapotranspiration of deciduous trees is negligible during the winter months but given long periods of warmer temperatures during the winter, non-deciduous trees such as Eucalyptus species may

consume significantly more water (LTS International, 2006), drying out the soils and intercepting more pollutants.

c) Preparation of site for planting and harvesting

Since only the rows where seedlings are to be planted require cultivation, the initial site preparation associated with SRF establishment is likely to raise fewer environmental concerns such as loss in soil C than soil cultivation associated with the establishment of annual crops. Compared with arable land use, SRF is likely to have a stabilising effect on the soil, due to the relative infrequency of soil cultivation (Makeschin, 1994). Of greater importance is the reduced frequency of site preparation necessary for SRF, i.e. once every 8 - 20 years rather than annually (LTS International, 2006). Soil compaction and the potential for gully erosion is reduced since there is no need for multiple mechanised applications of agrochemicals and fertiliser (LTS International, 2006). This decreases the potential for losses of organic matter and nutrients through leaching, erosion and transfer to the atmosphere, and also diminishes soil compaction (Makeschin, 1994). In addition, the provision of year-round soil cover and the network of fine roots in the upper soil layer improve water infiltration, and, together with leaf litter, resist the impacts of water droplets and thus reduces sheet erosion (Kort *et al.*, 1998).

Quicker-growing tree species grown on shorter rotations such as SRC will require somewhat more frequent establishment operations compared with SRF and conventional forestry and will therefore have a less positive impact on soil nutrients, organic matter content, and physical structure (Borjesson, 1999). SRC is harvested in winter after the leaves have fallen, on a 4- to 5-year cycle. In Scandinavia, harvesting takes place when the soil is frozen but, in the UK and elsewhere in northern Europe, the harvest will frequently coincide with wet weather and high soil water contents (Wall and Deboys, 1997; Mitchell *et al.*, 1999). Under these conditions, compaction, puddling and rutting are particularly likely given the high axle loads of SRC harvesters and associated machinery (Soane *et al.*, 1981; Kofman and Spinelli, 1997). Apart from damaging the soil, rutting and loss of traction delays the harvesting operation, adds to costs and can result in sideways slippage of machinery, causing mechanical damage to stools. This can cause 20% reduction in stem dry mass on a clay loam and sandy loam soils as a result of fewer stems being produced (Souch *et al.*, 2004). Stems and stools of *Eucalyptus nitens* are particularly susceptible to this kind of damage (Little *et al.*, 2002). Arguably the most important factor affecting soil loss and quality of run-off during site preparation is the planning and execution of operations (Kort *et al.*, 1998). For example, intensive cultivation before plantation of SRF (poplar and willow) into grassland resulted in high mineralisation rates of the organic matter with losses of soil carbon by 15% and soil nitrogen by 12% (Jug *et al.*, 1999). Existing publications provide a protocol for sensitive management in this regard (*e.g.* DEFRA 2002; Hall, 2003). Care must be taken not to access the site when the ground is wet, especially if the soil is clay or clay-loam, common under agricultural sites. Using low-pressure tyres and driving on lop and top

whenever possible will reduce soil compaction. As for thinning operations, minimisation of soil compaction during harvesting should be assured through judicious timing and the exercise of care during operations. Removal of for example half of the canopy may subject the unprotected ground to increased risk of erosion, but there is likely to be some compensatory protection by the woody debris left from the thinning operation. Moreover, the reduced shade will enhance the establishment and growth of understorey plants, the cover and root systems of which will reduce erosion until they die back after the re-closure of the canopy (LTS International, 2006).

Resumé During the life-span of the SRF plantation physical soil erosion should be much reduced compared to annual cultivation, and infiltration should be improved with greater rooting depth. However, there may be increased water uptake from evergreen species such as eucalyptus. During site preparation and harvesting there may be increased compaction of soils through frequent traffic movements..

d) Control of competing vegetation

The weed management regime is likely to affect the soils through the effects of:

- over-applied herbicides on soil quality;
- the susceptibility of soil to erosion and also
- the impacts of the mulching material (whether thermodegradable plastic, debris of organic matter, or otherwise) on soil quality (LTS International, 2006).

Compared to conventional agricultural usage, application of herbicidal chemicals during the course of the entire SRF rotation are small, since after canopy closure there is no further requirement for weed control until the start of the next rotation. Glyphosate is known to degrade quickly into environmentally benign components, and has no known negative effects on water or soils. In general, the best way to minimise application of herbicides is to ensure rapid canopy closure through vigorous tree growth (LTS International, 2006).

It is possible that agriculturally marginal land, which is perhaps the grade of land most likely to become available for SRF in the UK, may be intrinsically particularly susceptible to erosion (Kort *et al.*, 1998). In this scenario, the weeding regime can be lightened in order to reduce cover-free soil, and conservation areas can be established in zones most susceptible to erosion.

The use of plastic mulches is becoming common in agriculture, and there is no reason to assume that any negative hydrological or edaphic impacts associated with their use should be greater in SRF than in agriculture. Although no detailed studies on their impacts were found, thermodegradable polyethylene mulch is expected to decompose into inert chemical constituents (LTS International, 2006). The use of slow degrading

plastic has been noted at the sites under Eucalyptus at Daneshill (Repton) and Alcan (Newcastle). Weed growth of grasses and herbs was very noticeable under *E. gunnii* with fast degrading plastic protection, but much less in stands with plastic lasting 4-5 years. The exception was found at the fertile Newcastle site, where a very definite vigorous N loving weed community was evident up to 20 m into the stand rooted into the unprotected 50 cm between the lines of plastic.

If the subsequent tree crop is to be replanted, the stumps will require treatment with Glyphosate. After harvesting, the period before the next crop is established exposes the soil to a greater threat of erosion than during the rest of the cycle (Kort *et al.*, 1998), especially if the understorey vegetation is sparse or non-existent.

3. Relative impacts

a) Arable to SRF

Studies of invasion of old field by forest, and limited information from short rotation plantations, lead to a hypothesis that SRF will increase soil C by 10-25 t ha⁻¹ over 10 to 15 year rotation (studies summarised in Table 3). Soil C under arable land use has usually been reduced from native, undisturbed levels (Grigal and Berguson, 1998), by as much as 30% or more of their organic carbon content (Paul *et al.*, 1997). Highly productive woody crops will add substantial C to soil, both aboveground and belowground. In addition, within 2-3 years after plantation establishment, mulching by leaf litter, the lack of cultivation and increased rhizodeposits (Baum *et al.*, 2009) will slow decomposition and further help retain C. However, some experimental results with five hybrid poplar plantations, from 6 to 15 years old, found no differences in soil C compared to adjacent row crops or hayland (Grigal and Berguson, 1998). A simple analysis of C balance indicated an initial decline and then an increase in soil C, consistent with other literature reports. By contrast, the study by Vesterdal *et al.* (2008) has shown increases in stored C under certain species, notably conifers. Baum *et al.* (2009) report increased carbon sequestration when SRC is planted on former arable soils, however, the amounts of carbon stored seem to be governed by the initial soil properties, and therefore approaches for the selection of most promising sites for carbon sequestration must be developed.

Results of a meta analysis on the influence of land use changes on soil C stocks from 74 publications (Guo and Gifford, 2002) indicates that soil C stocks increase after land use changes from arable to pasture (+ 19%), arable to plantation (+ 18%), and arable to secondary forest (+ 53%).

Table 3. Rates of soil carbon sequestration after re-establishment of deciduous forest on agricultural land.

Forest Ecosystem	Previous land use	Years since land use change	Soil depth (cm)	Soil C rate (kg C ha ⁻¹ y ⁻¹)	Reference
Old field succession to hard woods (US)	Arable	10	10	151	Zak, Grugal et al, 1990
Old field succession to mixed oak (US)	Arable	>250	15	94	Robertson and Vitousek, 1981
Oldfield succession to hardwoods (US)	Arable	>100	10	116	Robertson and Tiedje, 1984
Abandoned field to mixed forest (US)	Arable	66	43	22	Hamburg 1984
Natural oak forest succession, Broadbalk (UK)	Arable	100	30	561	Jenkinson, 1990
Natural oak forest succession, Geescroft (UK)	Arable	102	30	426	Poulton, 1996
Planted hardwood, West tofts (UK)	Heathland	21	70	~47.6	Ovington, 1956
Planted hardwood, Bedgebury (UK)	Hazel coppice/standard s	20	70	(v.high >700)	Ovington, 1957
Planted hardwood, Abbot wood (UK)	Mixed oak wood	45	70	~666	Ovington, 1958
Planted oak, Alice Holt (UK)	Pasture woodland	80	15	116	Pitman and Benham FR

Paul *et al.* (2002) reviewed global data on changes in soil C following afforestation, available from 43 published or unpublished studies, encompassing 204 sites. Data were highly variable, with soil C either increasing or decreasing, particularly in young forest stands (i.e. <10 years old). Weighted average change of soil C (i.e. sum of C change divided by sum of years since forest establishment) relative to the soil C content under previous agricultural systems, generally decreased in the <10 cm (or <30 cm) layers by 3.46% y^{-1} (or 0.63% y^{-1}) relative to the initial soil C content during the first five years of afforestation. This was followed by a decrease in the rate of decline and eventually recovery to C contents found in agricultural soils at about age 30. In plantations older than 30 years, C content was similar to that under the previous agricultural systems within the surface 10 cm of soil, yet at other sampling depths, soil C had increased by between 0.50 and 0.86% y^{-1} . Amounts of C lost or gained by soil were generally small compared with accumulation of C in tree biomass (Paul *et al.*, 2002). During the first phase after afforestation an enhanced mobilization of easily decomposable above ground and root litter residues takes place. Subsequently the carbon and nitrogen contents of afforested soils increase in the long term mainly due to the litter input and the lack of frequent soil cultivation (Makeschin, 1994).

In a study of the effects of previous land use on forest C and N status in upland France, Koerner *et al.* (1997) showed that soils under forest traceable as past crop land contained less C and N even after 80 years. Reductions of up to 10% in deeper soils were discernible in C%, but less than 5% in N content. In a plantation of SRF willow and poplar on arable soils, with previously homogenous soil organic carbon, increases in soil organic carbon in the top 0-20 cm occurred, with lowered values at greater depth (Jug *et al.*, 1999). Soil carbon increase has been reported also in a few other studies for conversion of agricultural lands to woody biomass crops (Hansen, 1993; Grigal and Berguson, 1998; Tolbert and Wright, 1998; Moscatelli, 2008). For example, four years after land was converted from conventional tillage corn production to biomass crops in the southeastern US, soil carbon storage had increased by approximately the same amount in the surface layer of the plots. As a comparison, the soil carbon in the sweetgum plots without a cover crop decreased over the first four years. At other southeastern sites with woody crops, initial increases in carbon storage occurred, primarily in the upper 2.5cm. In the north-central US, the carbon content of the soil under 2- 10-ha plantings of hybrid poplar increased despite initial losses due to soil erosion and carbon mineralization (Hansen, 1993). After the initial 4 to 6 year establishment period, the measured amount of organic carbon stored in the soil was 191 t ha^{-1} , greater than or equal to the organic carbon in sites with row crops (179 t ha^{-1}) or grass (157 t ha^{-1}). The increased storage was especially noticeable at depths below 30 cm, where most coarse root development occurred (Hansen, 1993; Grigal and Berguson, 1998). In comparisons among various ages of established poplar plantations, the carbon gain was most significant in the 30 to 50 cm layer. Furthermore, in older 12 to 18 year-old plantations, the measured carbon gain exceeded soil carbon under adjacent

agricultural crops by about $1.6 \text{ t ha}^{-1} \text{ yr}^{-1}$, and it approached the levels estimated for natural forest soil (50 to 200 t ha^{-1}). These data from the US parallel the soil carbon increases predicted by European studies (Börjesson, 1999).

According to the present data, the changes from agricultural to forest land use initiate slow development towards a new acidity status on a lower pH level and lower soil buffering capacity as long as no corrections are made via liming or fertiliser applications. Reduction in pH under SRF planted on previously arable soils have been recorded by Jug *et al.* (1999) in Germany, by Alriksson and Olsen (1995) in Sweden, and specifically under alder by Grandall (1994) and Ugolini (1968). On the other hand, converting arable land to short-rotation forestry can result in reduced amounts of nitrate, phosphorus, pesticides, and herbicides in runoff and groundwater (Hohenstein and Wright, 1994; Ranney and Mann, 1994; Lal *et al.*, 1998; McLaughlin and Walsh, 1998; Börjesson, 1999). Simulation models predict that the replacement of conventional crops with willow removed nutrients and metal contaminants from waste water (Perttu, 1998; Börjesson, 1999; Wilkinson, 1999). Willows have extensive fibrous root systems, which contribute to soil stabilization and nutrient capture. Less is known about micronutrient requirements, but studies have shown that micronutrients such as boron can potentially be depleted (Vogel, 1996). Soil solution nitrate can significantly be reduced in soils planted with fast growing trees, as long as nitrogen fertilizers are applied in accord with the nutrient demands of the trees. Among soil organisms, microbial biomass and most faunal groups, especially decomposers, are advanced under tree plantations. The diversity of soil fauna is generally increased compared to arable land (Makeschin, 1994).

Resumé Land mainly available for SRF in the UK is former agricultural land, which is rich in base cations, nitrogen and phosphorus. Growing SRF crops for biomass over time, potentially leads to significant soil nutrient depletion and soil acidification. SRF may considerably affect soil moisture since trees tend to use more water compared to arable crops and pasture. Alternatively, changing land use to SRF may improve water quality by reducing soil erosion as well as level of nitrate, phosphorus and other chemicals and surface runoff and ground water. This land conversion may also improve soil biodiversity and providing improved quality habitat. Conversion of agricultural lands to biomass crops has potentially beneficial effects on soil carbon dynamics, but these effects are less documented, with gains in C for conversion from arable to plantation of up to 20% claimed by some authors. This new carbon may be more labile than carbon previously lost from the sites, and increases in C at depth in soil profiles have been reported after a lag of 3-5 years. .

b) Grassland to SRF

Results of a meta analysis on the influence of land use changes on soil C stocks from 74 publications (Guo and Gifford, 2002) indicate that soil C stocks decline after land use changes from pasture to plantation (−10%), and native forest to plantation (−13). One

outcome of the meta analysis, in the context of carbon sink strategies for greenhouse gas mitigation, is that broadleaf tree plantations placed onto prior native forest or pastures did not affect soil C stocks whereas pine plantations reduced soil C stocks by 12–15% (Guo and Gifford, 2002).

The effect of land use change on the dynamics and annual rate of net nitrogen mineralization in a naturally generated silver birch (*Betula pendula*) stand on abandoned agricultural land in southeastern Estonia, was assessed by Uri *et al.*, 2008). Annual net nitrogen mineralization rate in the birch stand ($156 \text{ kg N ha}^{-1} \text{ year}^{-1}$) in the upper 0–20 cm soil layer was higher than in grassland ($102 \text{ kg N ha}^{-1} \text{ year}^{-1}$); but in both cases net nitrogen mineralization covered a major part of the plants annual N demand. In the deeper topsoil layer (10–20 cm) net nitrification made up a significantly higher proportion of the net nitrogen mineralization in the grassland soil than in the birch being 58 and 35%, respectively. More intensive net nitrogen mineralization, however, did not lead to higher N leaching or emission losses from birch ecosystems compared with the grassland because an essential amount was retained in tree biomass (Uri *et al.*, 2008). It has been shown elsewhere that birch roots are very efficient in nutrient acquisition due to high stimulation of microbial communities due to high amounts of root labile C compounds in conjunction with rapid mineral-N uptake (Bradley and Fyles, 1995).

Soil C content decreased over 5 years of 1 x 1 m spaced plantation growth of *Populus deltoides* and *P. deltoides* x *P. nigra* hybrid clones on former pasture of high native fertility in the Missouri River floodplain in the lower Midwest USA (Dowell *et al.*, 2009). The decrease in soil C was primarily in the surface layers. After pasture, soil C may decrease slightly or remain stable, presumably because of the large root turnover and minimal tillage involved previously in permanent pasture have maintained soil C at high levels. The thick root mat of old pasture grasses at the surface produces maximum carbon concentration – of up to 40% increase over normal forest soils.

Consistent with these results, loss of soil C in tree plantations on former grassland is greatest early in the conversion and at the most shallow soil depths (Paul *et al.*, 2002). These authors concluded that early losses of soil C after afforestation might not persist as plantation age moves beyond 5 years. This result suggests that longer-term dedication of land to short-rotation forest cultivation might also to some extent reverse early losses of soil C. However, Koerner *et al.* (1997) have shown that these changes can be much longer term in some cases, and shows reduced values of up to 20% of C at depth in forest soils from previous grassland 60–80 years previous use.

The most important factors affecting change in soil C were previous land use, climate and the type of forest established. Results suggest that most soil C was lost when softwoods were established on ex-improved pastoral land in temperate regions. Long-term management regimes (e.g. stocking, weed control, thinning, fertiliser application)

may also influence accumulation of soil C. Accumulation is maximised by maintaining longer (20-50 year) forest rotations. Furthermore, inclusion of litter in calculations reversed the observed average decrease in soil C, so that amount of C in soil and litter layer was greater than under preceding pasture (Paul *et al.*, 2002).

In contrast to reduced soil pH under SRF on arable land, Muys *et al.* (1992) recorded a rise in pH under some deciduous species such as ash, lime and cherry planted into old pasture. Moffat (1990) recorded decreases in soil pH under alder species planted into rough grassland, compared to oak and grass control plots over 32 years. However, data from afforestation and short rotation forestry from a number of studies have shown that most of the trees species used in SRF in the UK have great tolerance to soil reaction (pH) as long as sufficient nitrate is available for the trees.

The benefits of SRF planted on ex-grassland land are not as clear as with SRF planted on arable land. In general, the precise consequences for soils of a shift in land use from grassland to SRF depend on:

- management: improved, unimproved pasture extensive or intensive use
- soil type: texture (light or heavy) and initial chemical status
- hydrological regime: rainfall amount, seasonal timing, and drainage
- slope
- tree species: growth rate, litterfall amounts and quality, abundance of understorey plants
- length of rotation and silviculture: type of mulch, timing of planting
- care with which thinning and felling operation are undertaken (LTS International, 2006).

Resumé The biggest effects in the transition from old grassland to SRF would be in the initial loss of carbon stored in the root mat of the pasture. This has been shown to recover on some sites after 5 years, and accumulation is best in even longer rotations of afforestation. However, this depends on the land management regimes adopted, including potential soil disturbance for thinning, weeding or the application of fertilisers. Soil N mineralisation frequently increases under woodland compared to grassland, but N stocks are then held in the above ground portion of the trees. Soil pH has been shown to increase under some broadleaf species following the transition from grassland.

c) SRC compared to SRF

The most important changes to soils in this land use substitution would be related to the length of potential rotation. Harmer and Howe (2003) give estimates of the likely rotation lengths for productive coppice being worked under the 'simple' system: 6-12 years for hazel, 12-16 years for sweet chestnut, and 20+ years for oak. In these systems, when the old stools are deemed exhausted, restocking regimes can be i) vegetative propagation from existing stools ii) natural regeneration from seed or iii)

nursery grown seedlings. In the first two cases there would be minimal interference to the soils over a long number of years, apart from some mounding over the stools to aid layering of shoots. If it is assumed that planted SRF were also to be worked as coppice in the second rotation (as expected from poplars and willows), then soil conditions could remain stable or cumulative in terms of carbon and nitrogen for upward of 20-25 years. However, if there is to be replacement of the entire crop on short rotations of 10 + years, the status of the soils will be very dependant on the techniques used to replant. Bearing in mind the potential N resource left by decaying boles and roots in the soil, it might be good practice to plant in whips/seedlings beside the left over stools, as long as there had not been disease in the crop. Minimal cultivation techniques will keep nutrient losses by mineralisation to a minimum.

d) Coniferous woodland to SRF

There is very little data available on this transition, but conifers are likely to have a lower pH therefore higher carbon stocks during the life time of the rotation. The effects of planting some SRF deciduous species into the same soil are likely to be increased pH (Pitman pers. comm. for birch) and a temporary decline in carbon stored in the litter and A horizons of the soil profile. This would result from initial decay of the accumulated coniferous litter organic matter and higher subsequent decomposition rate of the planted SRF broadleaves. On the basis of comparisons of deciduous and evergreen broadleaved species (see earlier in section 2) the effect of conversion from coniferous woodland to Eucalyptus is likely to be similar in direction but less marked than conversion to deciduous broadleaved species.

e) Deciduous woodland to SRF

This has occurred in replacement of native forests in both Tasmania (*Eucalyptus* sp) and Chile for *Nothofagus* species (no record of changes in Pastur *et al.*, 2009). For the case of *E. nitens* (Moroni *et al.*, 2002) a subsequent lower mineralisation rate was found in soils under plantations on ex-forest land than on ex-grassland. In the UK, only the resampling of the Ovington trials reported by Howard and Howard (1984) and Anderson (1987) record any soil effects of typical broadleaf SRF species planted into mixed oak standards with hazel coppice at Bedgebury. Here soil pH increased significantly below 10cm depth, along with K under *Nothofagus*, but soil total C and N decreased over 20 years. In a comparison of native rainforest and Eucalyptus forest in NE Tasmania by Adams and Attiwill (1990), the soils under native woodland were more organic and contained higher extractable P, inorganic N and other nutrients. Litterfall in the rainforest was richer in P relative to the eucalypts, but poor in Mg. They proposed that the root mat of rain forest was more effective at absorption than stands of Eucalyptus.

4. Anticipated impacts of climate change

As for other land covers, the impacts of climate change on forest soils cannot be reviewed in the absence of the wider land management issues of the forestry industry. Both climatic warming and rising CO₂ levels in the atmosphere will enhance tree growth in the short term and in young trees (Broadmeadow and Jackson, 2000), which could well apply in the growth dynamics of SRF tree species. However, no other UK-based research has corroborated these contentions in the longer term, and current predictions are based on a combination of impact studies on young trees and modelling. Globally, research findings on the effects of rising CO₂ levels have been equivocal, with the limited number of studies indicating rather small increases in growth rate and minimal or no impact on soil carbon stocks (Oren *et al.*, 2001; Heath *et al.*, 2005; Lukac *et al.*, 2009). The hypothesis is that in the longer term, increased tree growth may accelerate the depletion of soil nutrient pools through higher uptake rates. However, this is not likely to be the case with SRF planted on fertile ex- arable land. However, wetter and milder winters could increase acidification, nutrient leaching, reducing nutrient cycling within the soils and the SRF plantations. Changes in growth will also affect the amounts and quality of leaf production and litter inputs to the soil, thus having an effect on the soil nutrient pool. Rising CO₂ levels have also been shown in some studies to alter the C:N ratio of leaf litter which would be expected to reduce decomposition rates (Bradley *et al.*, 2005). If elevated CO₂ increases tree leaf area, this will have implications for water supply to the soil, which will decrease due to higher precipitation interception losses. Changes in leaf area will also affect the litter input to the soil. Forest floor microclimate may also be altered as a result of increased or decreased litter input to the soil and increased/decreased light interception.

Assuming constant inputs of carbon to soils from vegetation, soil-climate models predict that expected changes in temperature, precipitation and evaporation will cause significant increases in organic matter turnover and increased losses of CO₂ in mineral and organic soils across the UK. This will result in a positive feedback between CO₂ emissions from soils and further temperature increase. However more work is required to separate effects of climate change and land use change. Potential losses of soil carbon will also affect other soil functions. The greatest losses, relative to existing soil carbon content, are expected in southeast England, where rates of temperature increase will be greatest. This could lead to poorer soil structure in terms of stability, topsoil water holding capacity, nutrient availability and erosion. However, these effects could be offset by enhanced nutrient release resulting in increased plant productivity and litter inputs. Short Rotation Forestry planted on ex-arable land could be beneficial for soils and partly mitigate these suggested impacts due to the expected soil C accumulation. Changes in soil moisture content have also been predicted – including increased moisture deficit for forest soils in south east England and south east Scotland (Bradley *et al.*, 2005), where some of the current SRF sites are located. Increased droughts will increase the likelihood

of shrink-swell in clay soils, which is of particular relevance to SRF being planted on ex-arable land on heavy clay soils.

Increased winter rainfall, and particularly an increased frequency and intensity of extreme rainfall events could increase problems with land stability and erosion. Generally small effects on erosion are expected in Scotland, but there may be an increase on susceptible sites, particularly where tree cover is absent following windthrow or clearfell, or on exposed sites on steeper slopes. Increased winter waterlogging due to higher precipitation and if the storm events become more frequent, may promote soil disturbances as a result of tree windthrow. If good practice guidelines are not adhered to, then an indirect effect of climate change would be soil compaction and deterioration in soil structure (Bradley *et al.*, 2005). Leaching of nitrogen and phosphorus may also be enhanced by increased winter rainfall which will consequently affect water quality.

The effect of climate change on the greenhouse gas balance of forest soils is an area of considerable importance, but also uncertainty. This is particularly the case for organic soil, but may be also applicable to organic soil layer developed under SRF. Rising temperatures would be expected to increase soil respiration, while changing evapotranspiration and rainfall distribution would be expected to lead to changes in the water table and consequent effects on methane and nitrous oxide fluxes (Bradley *et al.*, 2005).

Mycorrhizae play an important role in nutrient cycling in the soils but also in the trees and have also been shown to confer some protection against soil-borne pathogenic fungi. Pollutant deposition, atmospheric CO₂ levels and soil temperature and water content have all been shown to affect ecto and endo-mycorrhizae. However, effects of climate change on the strength of these associations and their effect on tree performance has not yet been well documented (Bradley *et al.*, 2005).

Resumé The potential effects of climate change on the soil quality and sustainability under SRF plantations is still under discussion. It may be necessary to adopt very different Regional responses to change, as the South East becomes dryer and hotter, and the North and West wetter and more windy. Balances of increased growth, foliar production and litterfall through increased CO₂ will be positive additions to the organic status of soils under SRF particularly in the South East, but temperature increases accompanied by moisture stress will have negative effects, reducing vegetative growth and resulting in reduced carbon sequestration. By contrast, higher levels of rainfall and winds could result in more wind throw, disturbance to soils, potentially more surface erosion and nutrient leaching in the upland districts.

5. Information gaps

Primary soil processes affecting soil quality and availability of data from energy forestry research are summarised in Table 4. The information gaps are listed below.

1. Our analysis indicates that conclusive experimental data on soil carbon sequestration is currently constrained by the short time since most of the plantations have been established. The most informative long term literature comes from the US and Australian experience. Before definitive statements can be made about the C balance of SRF selected plantations must be followed in the UK through one or more rotations using standard protocols to facilitate intercomparisons.
2. To what extent intensive harvesting effects may deteriorate soil structure and cause overcompaction with negative consequences for tree growth and ecological functions of soils, needs further attention. This needs to be related to the frequency of movements associated with the routine management of each of the main species planted.
3. There is a need for future soils research, which should include chemical, physical and biological elements and processes. In particular, any changes associated with replanting methods, should coppicing not be an option for the second rotation, must be investigated, as this will increase the potential for mineralisation within the soil profile, with the potential loss of carbon and/or nitrogen. The long term balance of these elements can only be assessed if the harvesting practice is defined at the start. In the search for sustained high productivity, with shorter rotations adopted along with cultivation at replanting, then effects on the soil of repeated disturbance would have major long term effects on both physical and chemical soil quality.
4. The attempts to model the impact of bioenergy crops on carbon sequestration so far are associated with much uncertainty and there is an urgent need to validate and improve models by quantifying actual soil carbon sequestration rates by bioenergy crops.
5. The potential climatic effects on soil mycorrhizal populations does not seem to have been researched in sufficient depth. We would suggest that the action of these fungal groups is in general a very important area, but particularly in the successful establishment of young SRF trees on sites not previously afforested, and should be given more attention.

Table 4. Primary soil processes affecting "soil quality" and availability of data from energy forestry research.

Process	Soil characteristics impacted	Studies from energy forestry research
Carbon dynamics	Organic matter content, turnover, and pools	Several studies on nutrient dynamics of different tree biomass species
Nutrient dynamics	Nutrient content, Cation Exchange Capacity (CEC), soil pH	Several studies on nutrient dynamics of different tree biomass species
Litterfall dynamics	Nutrient addition, accumulation or mobilisation of organic matter	Good coverage of basic litterfall chemistry, very little on decomposition and incorporation.
Erosion	Water regime, loss of surface soil layers, depth of roots	Erosion data from a few studies
Compaction	Bulk Density, Porosity, structure, aggregate stability, depth of rooting	Very few studies and data
Biological activity	Biodiversity, organic matter and nutrient turnover, compaction	Several intensive studies on N dynamics, some on earthworm communities, soil microbiology and other biota

6. Potential future soils research: There is not much information on chestnut soil nutrients (*Castanea sativa*) in the UK- an important traditional land use of the lighter soils in the SE England. A future investigation of the soils status inside and outside chestnut coppice at Rogate on Greensands may provide us with some indication. Further research into the potential of hornbeam (*Carpinus betulus*) and oak (*Q. robur*) – both historically important coppice trees - should also be included into energy forestry research. There is also very little research from the UK and Europe on *Nothofagus*, *sycamore* or *Eucalyptus* litter and soil chemistry effects. This information is urgently needed under the UK climatic and soil conditions, so the suitability and sustainability of using exotic species for UK Biomass forest can be justified.

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Other sources of information

Suggested sources of information and references:

- IEA Bioenergy Task 38 'The role of soil carbon in the GHG balance of bioenergy systems'

www.ieabioenergy-task38.org/publications/T38_Soil_Carbon.pdf

- IEA Bioenergy Task 38 'Answers to ten frequently asked questions about bioenergy, carbon sinks and their role in global climate change'

www.ieabioenergy-task38.org/publications/faq/



Bury Ditches Hill Fort, Shropshire

SRF and the Historic Environment

Peter Crow

Summary

Many forestry or agricultural practices have the potential to severely damage elements of the historic environment and regrettably, this is known to of previously occurred in many parts of Britain. Thankfully, in recent decades, a greater awareness of both the historic environment and potential risks posed from different types of land management has helped to reduce the likelihood of further significant damage.

This improved awareness results less from specific, targeted research, but more from anecdotal evidence and better collaboration and communication between land managers and heritage sector workers. For example, there is very little research into the direct physical or chemical impacts of root growth on the buried archaeological resource, but there are known examples of damage seen during archaeological excavations. Similarly, where the preservation of organic-based artefacts, such as leather and wood, is known to be so much better from anaerobic, waterlogged environments, any process which alters either of these properties is therefore likely to have a detrimental effect on long-term preservation.

The text below is therefore presented predominantly on a basis of observations and experience of heritage sector workers (and others), combined with a knowledge and understanding of the rural environment and its management. To aid this review of SRF and potential interactions with the historic environment, comparisons are also made with other rural land-uses and energy crops where necessary.

This review considers potential implications of SRF (both positive and negative) and provides suggested areas for further research. It is acknowledged that with so many possible mechanisms by which damage to the historic environment may occur (for example windthrow) and the numerous other factors which can influence these risks (such as soil types, tree height, depth of any buried material), then each site must be judged on its own merits. This is best achieved in close collaboration with local authority historic environment advisors who can help to determine the likelihood of any risks and help mitigate against them where necessary. Further recommendations are also provided at the end of this section.

1. Introduction

The historic environment is all around us, providing a key to understanding the past and a valued connection with the cultural heritage of people and places. It has many forms, shapes and sizes, ranging from individual features such as buildings or standing stones to large-scale urban and rural historic landscapes. Individual features do not have to be ancient to make an important contribution to the historic environment but can range in date from the last century to many thousands of years ago. The term historic environment is used as it encompasses not only archaeological features, but also includes living heritage such as ancient woodland, veteran trees; natural heritage such as geological features and entire landscapes.

There are very few parts of Britain unaffected by past human activity and changes within the landscape are an important part of the human story and experience. Implementing a new change in land-use, or establishing new crops may involve the need to identify and conserve historic environment features in accordance with national and local government policies. Planning legislation and guidance provide helpful advice on how to successfully address historic environment issues in land-use change, regardless of whether or not a particular project requires formal planning permission. Some examples of issues relating to the preservation of the historic environment during land-use or crop changes are discussed below. However, with sufficient information and early consultation with the heritage sector, many of these risks can be removed or reduced and potential benefits possibly identified.

2. Potential impacts

a) Rooting impacts

Arguably, the main concern for potential damage to the historic environment from the establishment of trees is that of root impacts on buried archaeological evidence. Detailed studies into the impacts of roots on archaeological materials are very few, however from discussions amongst archaeologists, the following points are generally accepted:

- Roots can cause physical damage to buried artefacts. This can range from the total destruction of small objects, the breakage of larger remains to minor surface damage only.
- Chemical damage of artefact remains due to root exudates can occur, but this tends to be very localised.
- The active growth of roots through the soil can lead to the movement of small artefacts or the mixing of archaeological deposits. This can result in a loss of archaeological context.

- Root induced physical damage and bioturbation can be very localised and may not prevent the archaeological interpretation of a site. Therefore, woodland may be a preferable long-term land-use when compared to others which undergo regular ploughing.
- Roots can form a physical lattice which can help to stabilise soils and archaeological deposits associated within sloping sites or earthworks, reducing the risk of erosion. But, careful management is needed to prevent windthrow.
- Windthrow has the potential to cause significant damage to archaeological evidence. During windthrow events, a single tree or an entire area of forest may be blown over, lifting root plates from the ground in the process and including any archaeological evidence occurring within the upper soil horizons.
- Where soils are thin or impoverished, roots may exploit archaeological deposits if they provide a preferable rooting medium.

The likelihood of any of these risks occurring will inevitably be very site specific, depending not only on the nature, depth and extent of any archaeological evidence present, but also the size, stocking density, rotation length and the species of tree being grown. For example, there may be little below-ground disturbance for many years once a mature, broadleaf woodland has developed a closed canopy and no forest operations are required until final harvesting. This is especially true for woodlands in lowland areas with little risk of windthrow. Where necessary, silvicultural practices such as pollarding can help to reduce the risks of windthrow in situations where the risks are perceived to be higher.

Regular coppicing is also known to limit the diameter of roots, partly due to the lack of requirement for supporting tall above-ground biomass. Such Short-Rotation Coppicing (SRC) tends to produce smaller roots than standard trees of a similar age, most of which occur within the upper soil horizon (Crow and Houston, 2004). However, the high productivity of SRC and the relatively higher stocking density compared to conventional woodland will lead to a high abundance of fine 'feeder' roots required to supply the necessary nutrients and water for sustained growth. This can lead to a greater rate of bioturbation within shallow soils or deposits than that occurring under conventional woodland. The energy crop miscanthus produces a rhizome that proliferates primarily by horizontal growth, periodically sending up the shoots which will be harvested. Whilst some roots will descend vertically from this rhizome, this predominantly lateral growth will occur in the upper soil horizon. Like SRC, this will result in a high degree of bioturbation in the upper soil. However, where SRC or miscanthus are established on land which has been under regular cultivation in the past (for example potato production), any further mixing of a ploughed soil via bioturbation may not be an issue.

The potential rooting impacts of SRF will differ from those of SRC, as the increased crop height and rotation length will inevitably produce structural, stabilising roots of a larger

diameter, more akin to mature woodland. Where the stocking density for SRF is higher than that of a young broadleaf woodland plantation, the bioturbation in the upper soil horizon may be comparable to SRC, but again former site cultivation may make this irrelevant (depending on depth of former cultivation). Whilst SRC will be cut frequently, the stool can remain in place for many years. Conversely, the frequent harvesting and re-establishment of SRF (perhaps in-between previous stumps) may have the potential to create greater rate of bioturbation through new root growth at depths below that seen for SRC or miscanthus. Nonetheless, like conventional woodland, the rooting depth will be determined primarily by the soil conditions of the site, but also to a lesser degree, the species of tree being established (Crow, 2005). The table below is derived from Crow's review of 2005 and shows the maximum rooting depths given in other published literature for some species which may be proposed for SRF schemes. It should be noted that these depths are based on soils with favourable rooting conditions with no impeding horizons which would reduce these figures. These values are indicative only.

Species	Probable maximum rooting depths, (based on other published values)
<i>Acer pseudoplatanus</i>	1.5m
<i>Alnus glutinosa</i>	2.5m
<i>Betula pubescens</i>	2.0m
<i>Castanea sativa</i>	2.0m
<i>Fraxinus excelsior</i>	2.0m
<i>Populus tremula</i>	2.0m

Further research into the typical rooting characteristics of exotic tree species when grown on the types of soils found on proposed woodfuel sites would help inform guidance or the risks of rooting impacts

b) Hydrological impacts

Given that SRF is proposed to obtain a high biomass productivity, fast growing species are selected and need to be established on sites that will sustain their vigorous growth. In the majority of cases this will equate to a fertile, lowland soils, possibly of former agricultural use and with a sufficient water supply. Alluvial floodplains, reclaimed wetlands and natural basins may also meet such requirements and are therefore possible sites, especially where sites do not flood on a regular basis, but are maintained by a high water table. Frequently flooded sites may be less desirable as regular immersion of the roots can inhibit growth although this will be influenced by the species present. Equally, frequently flooded sites will make site access and vehicular operations more problematic. Wetland environments are also well-known for their ability to yield well-preserved archaeological remains, especially organic materials such as wood or leather, which are lost from typical archaeological deposits on drier sites. Good

preservation requires permanent waterlogging, and when these soils begin to dry out, oxygen is able to reach these artefacts, facilitating chemical and biological degradation. Any change in land-use and associated drainage which potentially reduces the water levels on a site will have a negative impact on the preservation of any organic remains present.

The establishment of trees on, for example, a reclaimed floodplain, may often be considered detrimental due to a higher water demand than short vegetation such as grass. The total water uptake from a site during a 12 month period can be very similar for both broadleaf trees and grass, as the latter will have a longer growing season and is therefore removing water from the soil for a greater part of the year (Hall, 1996). Conversely, deciduous woodland will remove water for a shorter period, but to a greater extent during that time. The extent of water removal will differ slightly from one species of tree to another, the stocking density etc, but the fast growing nature of both SRF and SRC may result in a high water demand.

The soil properties of a site will also influence the ability of any vegetation to extract water from greater depths. For example, an alluvial floodplain may contain a soil horizon of compacted gravels, sands or clay that impedes root penetration. Conversely, peats and other organic deposits may facilitate greater root penetration. Similarly, the hydrologic conductivity of the different soil horizons will also determine to what extent capillary action will draw water from further down the profile as it is removed from above. Any artificial drainage systems installed to 'improve' a site's suitability will have a detrimental impact on any wetland archaeology present. Where energy crops are established on a naturally well-drained soil, hydrological impacts on the historic environment will be less of an issue. In these drier soils, the main change will be potentially to the chemistry of the soil water and thus, in turn the burial environment (Crow, 2008).

c) Soil chemistry

A change in land-use can alter the chemical composition of a soil that can, in turn, alter the preservation of any buried archaeological artefact or deposit. Such changes can be surprisingly rapid (as little as 5 to 10 years depending on the land-use change) and difficult to predict as they will result from a combination of various potential influences. The establishment of any tree species, whether as high forest, SRC or SRF will usually result in an increased quantity of organic matter in the upper soil horizon as a result of litter production. For most broadleaf species, this will lead to an increase in carbon and soil moisture within the upper soil horizons. The rate of litter breakdown and incorporation into the soil will be influenced partly by the climate, but also the soil fauna present and the palatability of the litter. For most native tree species, this palatability is likely to be similar, but may be different for exotic species. Alterations to the soil chemistry below the uppermost horizon will be determined primarily by other factors

such as the initial soil chemistry, the site hydrology, associated soil fauna and the geographic location of the site (as this will influence the quality and quantity of the deposition and precipitation), soil temperature and drainage pattern. All of these can influence either the chemical composition of the burial environment or the rates of reaction or change within it, which then determines whether or not an artefact would be chemically preserved or degraded (Crow, 2008).

Given the complexities and uncertainties associated with predicting changes in soil chemistry due to the vegetation type, it is difficult to provide a generalised outcome of the establishment of particular tree species. The chemistry of a burial environment will be primarily controlled by the soil, the local climate and surrounding air quality. The litter associated with broadleaf woodland can increase the carbon and moisture content of a soil, but neither of these are likely to be detrimental to the burial environment. *More research on soil chemistry under potential exotic species would help provide informed guidance.*

Many energy crops will benefit from applications of herbicide during establishment and periodic additions of fertilizer. Most herbicides are applied in low concentrations and are broken down quickly when in contact with the soil. There is unlikely to be any significant impact to buried archaeological materials from herbicides. Fertilizers vary more widely in chemical composition and may be mineral based (such as rock phosphate), organic waste (such as farm manure) or a process waste (such as wood ash). The potential impacts from these chemicals are therefore more difficult to determine as they vary significantly in their chemical composition, the quantities applied and their longevity on site. Equally, the chemical nature of the soil and the movement of soil water will also have a bearing on the chemistry and rates of change in the burial environment. The range of fertilizers can also vary in pH and some have been linked to an increased chemical degradation of metal artefacts (Kars, 1998). However, there is also the potential that rock phosphate could help in the preservation of artefacts in the soil such as bone (Crow, 2008). *More research into the chemical impacts of the various types of fertilizer which could be applied would be beneficial to the provision of informed guidance.*

d) Cultivation

Where the establishment of tree species is proposed either for SRF or SRC on former agricultural land, some degree of soil cultivation is usually recommended. Ploughing is well-known for causing damage to archaeological sites, especially where the plough continues to cut deeper on each occasion into previously uncultivated soil. However, where it is possible to maintain ploughing to a constant depth, archaeological evidence can survive intact below it. Where a plough pan forms, most, if not all roots, are restricted to within the ploughed horizon. It is for this reason that the recommended cultivation prior to the establishment of any tree species is ploughing to a greater depth

than any that previously occurred on site, as this will break any pan present and allow deeper root development. Any increase in the depth of cultivation will inevitably be to the detriment of any archaeological evidence surviving just below the former plough soil. The deeper the extent of new cultivation, the greater the risk of encountering previously unknown archaeological material and causing damage. Unlike root induced damage, which can be very localised, cultivation techniques have the potential to destroy a feature or possibly an entire site.

Cultivation to make a site suitable for SRC or SRF establishment may also involve the cutting of drains to lower the soil moisture content of a predominantly waterlogged site. Physical soil disturbance will be severe, but localised. Whilst not detracting from this physical soil disturbance, perhaps the greatest archaeological concern will be from the drying out of previously waterlogged deposits and artefacts across the whole site leading to their loss as described above under hydrological impacts.

e) Establishment, routine maintenance and harvesting

Trees planted for energy crops are likely to be either planted as rooted stock or by direct planting of sections of cut stem which will develop their own roots. These may be planted by hand or in an automated process from the back of a vehicle. In either case any archaeological impact is likely to be negligible. Given that energy crops are established with the aim of a shorter rotation time than standard forestry, thinning operations may or may not be required depending on the crop and silvicultural regime. Where they do occur, it is most likely to be in the form of a line-thinning (potential impacts are covered below under harvesting). Routine maintenance is unlikely to require much more than possible applications of fertiliser or herbicide. Where they are necessary, application may not require vehicular access into the crop and if it does, only light weight vehicles should be necessary, therefore providing a low risk of soil disturbance (providing saturated soil conditions are avoided) and thus a low risk of damage to any buried archaeological material.

Harvesting is when a site will be most at risk from vehicular trafficking and soil damage from rutting. This may be less of an issue for SRC as the smaller stem weight facilitates a lighter harvester and removal vehicle. For SRF, harvesting is more likely to involve heavier vehicles, either in the form a harvester to fell the trees, or a forwarder to remove the timber from the site. Any risks to buried archaeological remains would originate from the formation of soil ruts caused by heavy traffic passing over soft soils. Equally, the use of skidders to extract timber can result in significant soil disturbance. Some soils are more susceptible to rutting or compaction than others and measures can be taken to reduce the impacts. Where forest residues (side branches and very small diameter stems), produced by the felling operations, are left on site they can be used to create a mat over which vehicles can drive thereby reducing the direct pressure on the

soil. Equally, avoiding harvesting during or following periods of heavy rainfall can reduce a soils susceptibility to damage.

f) Stump removal

Destumping may be proposed either between SRF rotations or prior to changing land use after energy crop production. This involves mechanical removal of the stump which pulls up many of the structural roots from the ground and much of the soil with it. This results in substantial soil disturbance, potentially to a depth of 2 to 2.5 metres (although this may be influenced by the soil type and tree species present). Any archaeological evidence buried within this soil depth can be physically damaged, and/or removed from its context, thus compromising the site integrity and potential interpretation. Given the number of stumps occurring over a site and subsequent rotations, this has the potential to cause severe damage.

If a change of crop or land-use follows SRC (or miscanthus), deep ploughing is often recommended to remove the crops root stock. Unlike destumping, this impact will predominantly be limited to the depth of ploughing and does not have the potential to extend to several metres, however the impact (which may still be to 1 metre or more) will cover the whole site.

g) Subsequent rotation

Whilst SRC can be harvested many times without the need for replanting, eventually the stools decrease in productivity or may be lost due to mortality. If this land use is to continue, some replanting will be necessary. Similarly, a further rotation may be planned on a site following the harvesting of a SRF stand. In either case, further planting is unlikely to lead to any new risks to archaeological evidence, beyond those mentioned in cultivation above. However, it is acknowledged that with rotations as potentially frequent as 8 years, the regular development and loss of root systems will lead to a high degree of bioturbation in the upper soil horizons. It is also possible that a change of species could have a differing rooting habit, water requirement, soil chemistry than that previously on site.

h) Landscape and aesthetics

Most rural parts of Britain have a characteristic man-made landscape associated with their past management. These landscapes are partly a result of their underlying geology which, throughout history has influenced the building materials and character of villages, towns and entire cities and also the types of land-use and agriculture surrounding them. This has contributed to regionally characteristic landscapes, many of which are considered sensitive to changes that are not in-keeping with their appearance. The establishment of a SRF crop with exotic tree species in a historically open landscape such as long-term agriculture or in areas traditionally managed as orchards, may therefore be

deemed as compromising the historic character of a landscape. Where a wood fuel crop is proposed for a brownfield site, there may also be strong heritage links with the former land-use, especially if it supported the local economy. Regardless of whether a brownfield or rural site, the degree of sensitivity will inevitably vary, depending on the perceived impact in relation to the crop's contribution to climate change mitigation, the scale of the historic landscape and the new proposal within it, the visual impact of the latter and the rarity (value) of the historic landscape. Where historic landscape is perceived to be in conflict with a SRF proposal (a perception which may change over time), it may be possible to accommodate the crop by screening, for example by using a line of native trees around the crop which are more in keeping with the surrounding landscape. Similarly, the size, shape, location and therefore visual impact of wood fuel crops can influence a scheme's acceptability.

i) Ancient woodland and veteran trees

Ancient woodlands are usually classified as areas which have had a woodland cover for at least the last 400 years. Many contain native tree species, whilst other ancient woodland sites now contain non-native plantations. There are now many policy initiatives to remove the latter from ancient woodland sites to replace them with native species. There are currently no proposals to establish SRF on ancient woodland sites, but in the longer term, where non-native plantations are being removed, the use of an ancient woodland site for biomass production with native species may be considered. Archaeological concerns of artefact damage due to root impacts will be less on ancient woodland sites, as continuing the woodland management would not be significantly increasing and risks of damage. Indeed there are perhaps opportunities for reinstating traditional management practices such as coppicing, perhaps helping to maintain the historic character of the area. The establishment of any non-native trees as an energy crop on ancient woodland sites would be contradictory to the desire for native species and is therefore unlikely to be proposed.



An example of an old boundary bank with a veteran tree upon it. Care is needed to ensure that such heritage assets are not needlessly damaged or destroyed.

The establishment of energy crops in close proximity to veteran trees may have a detrimental effect, as they may out-compete the older tree for valuable resources. This may be relevant where individual trees occur on agricultural land where they were once part of a boundary which has since been removed. Such trees may be hundreds of years old and therefore have not only valuable ecological habitats associated with them, but also a historic value, perhaps indicating a former boundary location or meeting place. Where agricultural land is proposed for SRF, there may be every good intention to retain these trees within or adjacent to the new crop, but increased cultivation, drainage or competition for nutrients or water may be detrimental to their longevity.

j) Heritage opportunities

Utilisation of woodlands for timber production and provision of underwood as a fuel or fodder crop has occurred for many centuries. Indeed it is only during this last two centuries that the use of the underwood has largely fallen out of fashion, partly due to increased use of coal, gas and oil. In recent years however, there has been a renewed interest in the use of woodfuel for small industrial and domestic use. Where native species and traditional management practices can meet the local requirements of woodfuel production, it may also be possible to work with traders to provide materials

for crafts such as hurdle or basket manufacture, thereby helping to preserve the local traditions. Perhaps there are further opportunities to re-educate people in the forgotten skills of traditional woodland management? It should also be noted, that heritage value can both change and be created. Newly created wood fuel crops may, in the future, be considered a valued part of historic landscape. *More research is needed into methods of evaluating the social and cultural views, values and opportunities with regards to the establishment of wood fuel crops and local heritage.*

3. Anticipated impacts from climate change

Current climate change predictions suggest that Britain may experience greater extremes of weather, with hotter, drier summers and predominantly milder, wetter winters. Many predictive models also indicate a greater frequency of storm events. All of these predictions have implications for the preservation of the historic environment, especially in combination with energy crop establishment. However, there may also be some benefits:

For example, concerns may increase over the desiccation of any waterlogged deposits due to a combination of water use by trees and increasingly drier summers. Similarly, as the climate warms in a SE-NW direction, the introduction of more drought-tolerant tree species into historic landscapes where they did not previously occur, will inevitably change their appearance to something possibly not in-keeping with the historic landscape character.

Conversely, the presence of tree roots can help to stabilise soils and deposits, reducing their risk to storm-induced erosion although significant damage can occur if the crops are not sufficiently wind-firm. Coppicing reduces the above-ground biomass and can therefore decrease the risks of windthrow. *More research is needed on how wind-firm SRF would be in comparison to traditional broadleaf woodland.*

4. Conclusions and recommendations

There are many potential impacts from the establishment, growth and removal of woody energy crops on the various aspects of the historic environment. Equally, many site-specific variables, which will determine the extent of any potential impacts, will inevitably need to be evaluated on a site-by-site basis. With sufficient information and consultation, it may be possible to reduce any risks. Despite this complexity, some general guidance can be provided:

- Gather as much information as possible about the site in question, as this will inform later discussions and decisions.
- Seek early consultation with the relevant local authority historic environment advisor, as they can provide information of known archaeological sites/features and advice on historic landscape characterisations and subsequent sensitivities.
- Sites with known archaeological features should be avoided, or the sensitive area incorporated into open, uncultivated areas. A minimum uncultivated or planted buffer of 20 metres from the known extent of the feature is recommended.
- Sites immediately adjacent to wetland archaeology should be managed to avoid additional water uptake or loss from the archaeologically sensitive deposits. A minimum distance of 30 metres between the known extent of an archaeological feature and the nearest drain is recommended.
- For some proposed sites, buried archaeological remains may not be known to exist, but perceived to be likely on the basis of other known sites from the surrounding landscape. Following consultation with historic environment advisors, energy crop establishment may proceed, but perhaps with minimal cultivation and no ploughing out of stools or destumping once the crop has reached the end of its productive life. A watching brief may also be necessary to monitor the site for any archaeological evidence that may become evident.
- If archaeological remains are thought to exist but buried to a depth of several metres, there may be less of an issue regarding the type of proposed vegetation as root growth would occur in the soil above. However, deep cultivation, destumping or soil desiccation may still represent a risk.
- If, at any time, unknown archaeological evidence is found, seek professional advice from a local authority or national heritage agency historic environment advisor.
- Where historic landscape character is not believed to be conducive to SRF establishment, the use of native species, traditional harvesting methods, rows of screening trees or a sensitive approach to landscape design may reduce any impacts.
- If coppicing of native species is to be utilised, consider the potential benefits of also working with local craftsmen to help preserve local traditions.
- Where proposed schemes enclose or will be adjacent to veteran trees, allow sufficient space around the tree with no planting of further cultivation. Seek advice where necessary.

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Further links for planning advice:

www.britarch.ac.uk/conserve/planning/index.html

www.bajr.org/developerweb/default.htm

<http://heritagecouncil.ie/publications/developers/>

www.forestresearch.gov.uk/heritage

Association of Local Government Archaeological Officers (ALGAO) UK:

www.algao.org.uk/Association/ALGAO_UK.htm

National Aerial Photographic collections:

Scotland: www.rcahms.gov.uk/

Wales: www.rcahmw.gov.uk/

England: www.english-heritage.org.uk/nmr

Online national monument record search (UK):
<http://ads.ahds.ac.uk/catalogue/>

Contact details for UK archaeological bodies:
www.britarch.ac.uk/info/uklinks.html



Wasp, Spider, Fly. Sculpture by Steve Blaylock

Potential impacts of Short Rotation Forestry on Biodiversity in Britain

Mike Smith and Georgiana Watson

Summary

Experience of short rotation forestry (SRF) in the UK is limited to less than ten small-scale plantings and there are no systematic assessments of their actual impact on biodiversity. Potential impacts are therefore estimated on the basis of relevant literature. This review, which builds on the comprehensive Hardcastle review (Hardcastle, 2006), summarises what little additional information has been published since. Here, through an understanding of the early successional nature of the structure and composition of SRF stands, an evaluation of species' capability to utilise these habitats has been made. The findings suggest these stands will have a positive biodiversity impact on woodland generalist species when planted on previously agricultural land. Also depending on placement SRF stands can reduce woodland fragmentation. These benefits can be enhanced if complemented with good management and design practices that promote biodiversity.

1. Introduction

The recent review of the potential impacts of Short Rotation Forestry (SRF) commissioned by the Forestry Commission and Defra (Hardcastle, 2006) considered canopy and leaf decomposition, vascular plants, non-vascular plants and fungi, mammals and other vertebrates – including hares, rabbits, bats, deer, wild boar, birds and bird assemblages – and invertebrates. The summary of this comprehensive review is reproduced in Box 1 with supporting evidence reproduced in Appendix 1. The Hardcastle review highlighted the limited information from the UK on the biodiversity impact of new SRF plantations but suggested that:

Although there will usually be an increase in biodiversity compared with cropland, pasture or SRC [short rotation coppice], those taxa that require mature trees and /or dead wood will not benefit from SRF. The potential increase in biodiversity will be strongly affected by the pattern and scale of the

planting, in particular the relative edge length and the linking up of set aside areas, hedges and existing trees to provide contiguity.

To advance our understanding of SRF's potential impacts upon biodiversity, we have reviewed recent literature, *i.e.* since Hardcastle (2006), covering three issues:

- the structural complexity and number of habitats that SRF provides.
- the impacts of SRF on bird, invertebrate and soil biota, including bacteria and fungi.
- the differences in the levels of dead wood present in SRF versus older plantations to determine the positive and negative impacts on dependant fauna.

2. Potential impacts

a) Stand Structure

The spatial arrangement of the various components of a forest ecosystem can give a good indication of woodland development and its influence upon the species that will be present within the stand. Compared to natural woodlands, and even mature plantations, SRF is likely to be more homogeneous in terms of species and age class. The rotation length will limit the structural diversity of the habitat to stand initiation and early stem exclusion phases; SRF stands can be considered as being structurally similar to some traditional coppice with rotations of ~ 20 years (such as sweet chestnut) and young conifer plantations. On the basis of both homogeneity and rotation length the biodiversity value of SRF is likely to be similar to some traditional coppice with long rotations and early conifer plantations.

The high stocking densities expected in SRF will result in low light levels reaching the ground, with subsequent impacts on the ground flora (Hardcastle, 2006). It is therefore appropriate to use information that relates to the biodiversity value of early successional conifer plantations. An understanding of the biodiversity value of plantations in these structural phases indicates that SRF is likely to benefit generalist open habitat and forest species.

b) Species capability to utilise SRF

An increase in invertebrate numbers in SRC compared to arable land and grasslands has been shown to lead to an increase in the diversity of birds (Sage *et al.*, 2006). These trials in both willow and poplar SRC contained higher densities of resident birds while willow also contained higher numbers of migrant bird species compared to surrounding open habitat.

There is also evidence (Baum *et al.* 2009) that soil biota can benefit from SRF, both in terms of soil fauna and micro-organisms. SRF differentiates homogeneous upper soil horizons into several distinct horizons thereby increasing soil biomass and biodiversity. Willow and several other tree species used in SRF, *i.e.* alder, eucalyptus and poplar, are capable of hosting two types of mycorrhizal associations simultaneously (ectomycorrhizas and endomycorrhizas), which has been shown to increase the fungal diversity of an area (Puttsepp, 2005).

Although there is evidence that SRF benefits several groups, others such as bryophytes, lichens and coleopteran have limited dispersal distances and are likely to take time to colonise sites and subsequent recolonisation after each rotation. The risk of some species failing to recolonise - a situation recorded by Oxbrough (2010) in the case of spiders that were present in the first rotation but were lost during harvesting and did not re-colonise the early stages of the next cycle of conifer plantations - could be minimised by retaining patches of SRF into the next cycle.

There is evidence (Sage *et al.*, 2006) to suggest that SRC can have a beneficial impact compared to arable land and grassland on a number of species from a range of taxon that consist of: Diptera, Arachnids, birds and fungi. It has been shown that the canopy of early stages of the forest cycle can support a wide variety of herbivorous and phytophagous invertebrates (Oxbrough, 2010), which in turn support predatory ground invertebrates.

Compared to native species, the choice of exotic species for SRF is likely to show similar impacts on biodiversity as exotic plantation which start developing old growth habitats and associated biodiversity after 80-100 years (Humphrey, 2005). These exotic plantations have been shown to support similar overall species richness and native counterparts (Quine and Humphrey, 2010).

c) Diversity within Dead Wood

Dead wood is vitally important for habitat provision for a number of species on the UKBAP list (Lindermayer *et al.*, 2005). Within plantation forests the proportion of dead wood capable of supporting dead wood species does not accumulate until 80 years after planting (Brin *et al.*, 2008). As SRF rotations are typically 8-20 years in length there will not be enough time for dead wood resources to develop and therefore the ability of this habitat to support Saproxylic species will be low. On the other hand if SRF plantations are located in close proximity to old growth native woods or mature plantations then SRF may have a use as foraging ground (Afes *et al.*, 2008) which could benefit species from several taxonomic groups, including members from the *Syrphidae* (Hoverfly). For example the majority of *Syrphidae* larvae require dead wood to complete their early development stage, but adults feed off flowers within more open habitat so SRF planted within their range may be beneficial (Gittings *et al.*, 2006).

d) Invasiveness

There is always the risk that any introduced species becomes invasive. Formal assessments of the risk of invasiveness of trees species are carried out under Defra's UK Non-Native Organism Risk Assessment Scheme. A preliminary risk assessment has been made on the Eucalyptus species used in the SRF network trials, which include the species in existing small-scale plantings. Subject to an on-going peer review, this indicates that the risk of invasiveness is low.

3. Methodologies

A number of recent trials investigating the impacts of plantations on biodiversity used invertebrate sampling to illustrate the impacts of differing management regimes. It should be remembered that sampling techniques have limitations and may introduce bias. Comparisons of control and treatment sites even if paired are likely to reflect differences in more than land-use. Some presence-absence assessments do not give a measure of absolute abundance but a measure of the relative diversity of the species groups assessed. This further reinforces the Hardcastle conclusions regarding the importance of monitoring of biodiversity in new SRF plantations, including the gathering of baseline data.

4. Anticipated impacts of climate change

The effect of changing climate on biodiversity in relation to SRF is even less well understood however it is likely that those species that use SRF will be under the same pressures as those that use other habitats. Concerns for the future distribution and function of species and habitats has led to attempts to anticipate these changes using climatic space modelling. This approach involves the construction of bioclimatic envelope models that use present climate-range relationships to characterise species' limits of tolerance to climate (such as temperature and water availability), and then apply climate-change scenarios to enable projections of altered species distributions (Berry, 2002; Baselga, 2009)

The potential changes to the suitable climate space of various UK species based on the UKCIP02 projections has been investigated through the MONARCH (Modelling Natural Resource Responses to Climate Change) project (Berry, 2007). The project assessed 32 UK BAP (Biodiversity Action Plan) species at the 50 x 50 km scale under current and future climate scenarios. It suggests species at the limit of their southerly range in UK may face extinction while others at their northern range limit may expand into new

territories. SRF plantations may have an important role in reducing woodland fragmentation. In the context of climate change, short rotation forests, by virtue of their rapid growth, could be particularly important by providing stepping stones or linking patches to improve permeability of the landscape allowing threatened woodland species to move through sparsely wooded landscapes towards more suitable environments.

5. Conclusions

Since the Hardcastle (2006) report there has been only limited further work providing evidence of impacts of SRF in Britain. The few studies that have been undertaken support the earlier findings. They suggest that expansion of SRF onto previously agricultural habitats and restored land will increase the diversity of these areas, but will probably only be beneficial to more widespread generalist species. The use of exotic trees as opposed to native species in SRF plantations will limit the number of species that can utilise this resource, as many species within Britain have strict microhabitat and foraging needs which will not be met by exotics.

The biodiversity impact of SRF has on Britain's biodiversity will be influenced by the location of the plantation, stocking regime, species used, distance from currently wooded environment and the species presently within the area as well as the design and management of the woodland. Incorporation of design features to promote biodiversity such as varying stand structure, creating open space and deadwood habitat could bring further benefits (Humphrey *et al.*, 2006). The benefits of SRF to generalist species could be increased through following forest biodiversity guidelines, which include requirements that plantation forests must contain 5-10 % open habitat, to increase the structural diversity of the habitat and therefore the number of species that can be supported (Luck and Korodaj, 2008).

SRF has the potential to reduce the levels of fragmentation, where new SRF is planted in close proximity to other wooded habitat. In the context of climate change, short rotation forests, by virtue of their rapid growth, could be particularly important by providing stepping stones or linking patches to improve permeability of the landscape allowing threatened woodland species to move through sparsely wooded landscapes towards more suitable environments.

A formal preliminary assessment indicates that the risk of invasiveness of the most promising *Eucalyptus* spp. for SRF in the UK is low.

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Box 1

Reproduced from Section 3, 9-16 of A review of the potential impacts of short rotation forestry. October 2005. Final Report. P D Hardcastle

1.1 Biodiversity**1.1.1 Landscape-scale impact of SRF location**

1. One of the main factors influencing the suitability of an area as habitat to a diverse range of plants and animals is the number of different types of habitats present in the landscape (Morin, 1999; Gaston & Blackburn, 2000). This principle needs to be considered in planning the location of SRF: all other factors being equal, SRF is more likely to enhance biodiversity when introduced to a treeless landscape than when planted in heavily wooded landscapes.

2. The type of woodland (*i.e.* as influenced by management, age, and species composition) already present in the landscape also has a bearing, because a number of differently managed woodlands consisting of differently aged trees of a range of species will benefit a greater range of plant and animal species than a monoculture of young SRF.

3. Maintaining SRF stands of different ages provides alternative habitat for animals displaced by the clear-cut felling of a stand (Sage, 1995). Also, given a heterogeneous landscape with differing aspect, exposure, soils and soil moisture, greater conservation value would be achieved by siting new plantations so as to cover a range of these differing abiotic conditions. Although a buffer zone should be left between SRF and existing woodlands or hedges (Maudsley, 2000) to avoid the loss of edge habitat due to shading by the plantation crop, plantations can nevertheless be located in such a way as to increase habitat contiguity, *e.g.* between isolated stands of existing woodland.

4. It is possible the existing UK guidelines for the siting of stands of SRC willow are insufficiently rigorous with regard to assessing, on a landscape-scale, the impact of new plantations (R. Collinson, *pers. comm.*).

1.1.2 Impact of Choice of Species

5. The tables in Appendix 1 show the basic ecological characteristics of the different species. In general, exotic species have less biodiversity potential than native species, as animals, especially arthropods (an important basis of the food chain), are not adapted to them. The understorey vegetation is also very important and is largely dependent on the canopy density. Species such as Ash and Birch have a much lighter canopy than Sycamore or *Eucalyptus nitens*.

The biodiversity of SRF plantations of different species appears to be strongly correlated with the density of the canopy, which determines the light level reaching the ground and hence the abundance of the herbaceous layer, and with the rate of litter breakdown. In general, the litter of exotic species breaks down more slowly than similar material from native species. It also carries lower numbers of phytophages. 0 summarises this information.

Table 1. Basic Species Characteristics Influencing Biodiversity

Species	Canopy Density	Litter Breakdown Rate	Arthropod diversity – number of taxa
Alder	Moderately Light	Fast	Very good
Ash	Light	Fast	Moderate
Birch	Light	Fast	Very good
Poplar	Moderately Light	Fast	Good
Sycamore	Dense	Relatively slow	Moderate
<i>Eucalyptus gun X dal</i>	Dense	Relatively slow	Low
<i>Eucalyptus gunnii</i>	Light	Fast	Low
<i>Eucalyptus nitens</i>	Dense	Relatively slow	Low
<i>Nothofagus nervosa</i>	Moderately Dense	Moderately fast	Moderate, increasing
<i>Nothofagus obliqua</i>	Moderately Dense	Moderately fast	Moderate, increasing

1.1.3 Previous land use

Unmanaged or extensively managed sites

7. Another key factor influencing whether SRF has a positive or negative impact on the biodiversity of the site is the site's previous management. Sites not recently managed, such as wetlands, shrubby ground, or neglected farmland, often provide ideal habitat for a large number of plant and animal species, and establishing an SRF on such a site would invariably lead to a decrease in conservation value (K. Kirby, pers. comm.). A number of extensively managed habitats, such as lowland calcareous grassland, lowland dry acid grassland, and unimproved neutral grassland, can be of very high conservation values. Such habitats, and others gazetted under the UK Biodiversity Action Plan (BAP), should clearly not be considered for SRF.

8. The rare and threatened plants and animals listed on the BAP website are not always restricted to habitat that is of obvious conservation value. A survey for rare or threatened species should be undertaken as part of an EIA before SRF is established, especially where the biodiversity is not well studied.

Intensively managed sites

9. Some 67 % of the UK's agricultural land has been managed intensively, for many years, to provide improved pasture and arable crops (MAFF, 2000). Most pastures are dominated by two plant species, namely rye grass and white clover. The arable land generally supports a monoculture of an exotic grain crop. During the spring and early summer, the grain crops (and to a lesser degree the improved grassland) provide both cover and forage to a number of bird species adapted to open habitats, such as skylark and lapwing (Vickery *et al.*, 2001; Anderson *et al.*, 2003).

10. Invertebrate groups can also be well represented in crops such as wheat and oilseed rape (R. Sage, pers. comm.; Baines *et al.*, 1998; Thorbek & Bilde, 2004), but plant and vertebrate diversity tends to be low compared to most other habitats (Wilson *et al.*, 1999; Corbet, 1995; Tews *et al.*, 2004). Plant diversity is kept down by the use of herbicides, which kill the plants and/or their seeds directly, and fertilisers, which enable a small suite of adapted plants (*e.g.* rye grass) to out-compete and smother most other plant species. Furthermore, the physical disturbance associated with agricultural operations prohibits the establishment of perennial plants. Mammal diversity is low due to disturbance and the absence of suitable cover and forage.

11. The non-crop areas of agricultural landscapes, such as hedgerows, roadsides, the verges of ditches, and set-aside, support most of the plant and animal species associated with farmland (Fahrig & Jonsen, 1998; Rypstra *et al.*, 1999; Wilson *et al.*, 1999; Di Giulio *et al.*, 2001; Vickery *et al.*, 2001; Tscharncke *et al.*, 2002; Wilby & Thomas, 2002). These relatively undisturbed areas can be rich in plant diversity, and the structural complexity and species richness of these plant assemblages attract species-rich assemblages of mammals, birds and invertebrates. This will need to be built on in planning SRF.

1.1.4 Biodiversity impact summary

12. Annex 2 provides full details of the biodiversity review. The following merely highlights the main findings. Although SRF will usually increase biodiversity compared with cropland, pasture or SRC, those taxa that require mature trees and /or dead wood will not benefit from SRF. The potential increase in biodiversity will be strongly affected by the pattern and scale of the planting, in particular the relative edge length and the linking up of set aside areas, hedges and existing trees to provide contiguity.

Plants

13. The understorey habitat beneath dense stands of SRF trees will provide habitat for a limited abundance and species richness of common and widespread plants, but can also provide suitable habitat for specialist woodland plants, in so far as they are able to colonise the area. The rides and other open areas associated with SRF will provide valuable habitat for species-rich assemblages of plants typically associated with woodland edges and rides, especially if the issue of soil eutrophication can be managed. Compared with both cropland and improved grassland, the SRF stand is likely to contain a greater abundance and diversity of non-crop vascular plants.

14. Apart from a small number of bryophytes, no particularly rare or threatened plants are likely to benefit from the establishment of SRF. Most red-listed plants in the UK are either associated with sunny, open habitats in southern England, and are rare because the UK represents the northern limit of a larger European population; or they are plants adapted to wetlands, most of which have been drained during previous centuries.

Mammals

15. The establishment of SRF in an agricultural landscape can potentially benefit most species of mammal due to the provision of additional cover by the tree crop and by the herbaceous vegetation associated with unplanted zones. Much like set-aside, these zones will also provide forage for both large and small mammals, and cover for smaller species.

16. This general increase in mammal biodiversity will need to be balanced against the possible increase of rabbits and deer, both potential pest species for trees and other crops. This is particularly important if fencing is required, as it represents a major cost item.

Birds

17. In general, SRF and the associated unplanted zones are likely to support a greater abundance and species richness of birds than intensively managed agricultural land, and the addition of SRF to a landscape will probably provide suitable habitat for additional bird species.

18. Some rare bird species adapted to open habitats would be threatened by the addition of SRF to a landscape, and could become locally extinct if significant areas of SRF were planted.

19. Of greater importance to most bird species than the identity of the tree species used in an SRF are the plants and animals associated with the tree, and the physical

structure provided by the tree. Tree species with darker canopies are likely to discourage ground feedings birds that require dense understorey vegetation, but may encourage insectivorous birds feeding in the canopy.

20. There is little evidence to suggest that exotic broadleaved trees provide poor habitats to UK birds. On the contrary, some exotics are preferred by British birds. It is not yet possible to make predictions as to how birds would fare in SRF plantations of *Eucalyptus* spp. (the most “exotic” and least widely present of the ten species).

Arthropods

21. In general, SRF will provide habitats for a more abundant and more species-rich assemblage of invertebrates than intensively managed farmland. Since many invertebrates will feed directly on the SRF trees, the species of tree used can have a large impact on the number and abundance of arthropods associated with the tree canopy. *Eucalyptus* spp. are likely to support less diverse arthropod assemblages than the other trees recommended for SRF. Increase in this population is likely to be quite slow; long established plantations in New Zealand show that local arthropods do adapt in time.

Appendix 1

Reproduced from Section 3, 9-16 of A review of the potential impacts of short rotation forestry. October 2005. Final Report. P D Hardcastle

Biodiversity Impact – Justin Mathews

Canopy and Leaf Decomposition

239. Two major determinants of the biodiversity within SRF are the density of the canopy and the rate of litter breakdown. Litter breakdown is affected by leaf structure but similarly structured material from native, or naturalised species, will breakdown more quickly than that of newly introduced exotic species because of the time lapse between introduction and the adaptation of associated species that are responsible for breakdown. These include arthropoda, fungi and soil microfauna and microflora.

1.2 Rate of leaf litter decomposition

240. gives indicative comment on the canopy density and the rate of litter breakdown in the ten species being reviewed.

Table 2 Decomposition Rate for Leaf Litter

Species	Canopy Density	Rate of litter decomposition	Reference
Alder	Moderately light	Rapid	Cornelissen, 1996
Ash	Light	Rapid	Cornelissen, 1996
Birch	Light	Intermediate	Cornelissen, 1996
Sycamore	Dense	Rapid	Pugh & Buckley, 1971; Mignolet & Lebrun, 1975; Cornelissen, 1996
Poplar	Moderately light	Intermediate	Cornelissen, 1996
<i>Nothofagus nervosa</i>	Moderately dense	Intermediate	(Wigston, 1990)
<i>Nothofagus obliqua</i>	Moderately dense	Intermediate	(Welch, 1997); pers. obs. (J. Mathews) from Tintern, Monmouthshire
<i>Eucalyptus gunnii</i>	Light	Slow	No published findings; however, author's experience in UK climate suggests decomposition is quicker than for <i>E. nitens</i>
<i>Eucalyptus gundal</i>	Dense	Slow	No published findings found. AFOCEL may have data from their work on this species in France.
<i>Eucalyptus nitens</i>	Dense	Slow	Wedderburn & Carter, 1999; Lopez <i>et al.</i> , 2001

241. Leaf litter from broadleaved trees provides organic material that is generally quickly decomposed and incorporated into the upper soil horizon (Drift, 1961). There are some differences between species in the rate of decomposition of the leaves. In general, broadleaved and deciduous exotics, such as *Nothofagus* spp., have a similar effect on soils to native broadleaves (Peterken, 2001). The litter of non-deciduous broadleaves such as *Eucalyptus* spp. take longer to decompose (Cornelissen, 1996), but will nevertheless decompose quicker than that of native conifers (Wedderburn & Carter, 1999).

242. However, the main factors affecting the rate of leaf decomposition are soil pH and soil moisture, with moist, base-rich soils providing conditions for the quickest rate of decomposition (Witkamp & Drift, 1961; Pereira *et al.*, 1998; Hunter *et al.*, 2003; Reich

et al., 2005). Due to their high capacity of *Eucalyptus* spp. for water uptake, it is likely that litter falling from *Eucalyptus* spp. will be exposed to relatively dry conditions, which may slow the rate at which the leaves are decomposed.

1.3 Chemical Composition of Litter

243. In general, the litter of deciduous broadleaved trees has a beneficial effect on soil chemistry and structure, and there are no reports of such trees causing problems in this regard.

244. Reports from the Congo suggest that in some cases, *Eucalyptus* spp. plantations can have a beneficial impact on soil organic matter and fertility (Bernhard-Reversat, 2001), but no research findings on the impact on soils in temperate Europe of the *Eucalyptus* and *Nothofagus* species under consideration have been uncovered in this review.

245. There is a widely-held opinion that *Eucalyptus* spp. trees can 'sterilise' the soil, i.e. emit chemical compounds which have an allelopathic (= growth inhibitory) effect on other species of plant growing in their vicinity. The scientific literature recognises that vegetation under *Eucalyptus* spp. plantations can be, depending on species and site, very sparse, but in most cases this is due to the ability of the trees to effectively capture available moisture, nutrients and light at the site (Cossalter & Pye-Smith, 2003). Watson (2000) recorded reduced germination on seeds treated with *E. globulus* extracts compared with extracts of Oak but this is an unusual finding.

246. In the British context, the popularity of *Eucalyptus* spp. trees, including *E. gunnii*, as ornamentals for gardens (*e.g.* RHS website) might also be construed as evidence to counter claims of an allelopathic effect. Visits to densely planted stands of *Eucalyptus* spp. in the UK gained by one of the authors (J. Mathews) during field visits to sites in Devon, Monmouthshire, Gloucestershire, Surrey, Kent, East Anglia and Nottinghamshire gave the impression that *Eucalyptus gunnii*, with its relatively light crown, supported a dense understorey of herbs, grasses and shrubs. Stands of *E. nitens* on the other hand, although less barren than most pole-staged Sitka spruce plantations, supported a depauperate understorey compared to native species other than beech. Figure 7 illustrates the canopy density of *Eucalyptus nitens*, Figure 8 shows that of *Eucalyptus gunnii*.

1.4 Effect of Litter on Soil Organisms

247. Under plantations, the high deposition of organic matter in the form of leaves and woody debris provides soil organisms with a more varied and abundant resource than is provided by agricultural crops, and this leads to a more species-rich and abundant assemblage of litter and soil-living organisms (Makeschin, 1994; Bardgett, 2002). There is evidence, however, that the leaf litter shed by monoculture stands supports a less

diverse soil organism assemblage than that shed by more species-rich stands of tree (Hansen, 2000; Zimmer, 2002).

248. Compared with less tall vegetation, forests can significantly increase the transfer of acidifying pollutants from the air to the soil and surface waters (Cannell, 1999), an issue which may be important downwind of cities and industrial areas in the UK.

2. Effect of SRF on vascular plants

2.1 Colonisation of SRF

249. It is expected that SRF will be planted on a site free of all competing vegetation. The use of Glyphosate-based herbicide and the presence of the thermodegradeable plastic mulch are intended to maintain the site relatively free of competing plants until the tree crop has achieved canopy closure. Glyphosate application kills most plants the leaves of which come into contact with the chemical, but does not kill seeds in the seed bank. Once regular application has ceased at a site, plants soon begin to establish (Lindgren & Sullivan, 2001).

250. However, the habitat conditions under the SRF canopy are likely to become such that many plants, especially weeds of arable crop systems, receive insufficient light to grow and reproduce, and can survive only under occasional gaps in the canopy (Persson *et al.*, 1989; Kirby, 1993; Parrotta, 1995; Smith *et al.*, 1999; Rich *et al.*, 2001). Examples of such plants are the annuals *Cirsium arvense*, *Galium aparine*, *Elymus repens*, and *Alopecurus myosuroides* (Sage, 1995), but also wild flowers of high conservation value once commonly associated with less intensive arable farming but now firmly in decline, such as *Adonis annua*, *Centaurea cyanus*, *Euphorbia platyphyllos* and *Scandix pecten-veneris*. The extent to which they are shaded out of SRF stands depends to a large degree on the choice of SRF tree species, but on ex-arable sites the initial understorey cover provided by this group of plants is likely to be sparse.

251. Soils of ex-grassland sites contain seeds and propagules of species better able to survive in shaded habitats, such as *Rumex spp.* For this reason, SRF established on ex-grassland are likely to have more species-rich and abundant understorey than those established on ex-arable land (Tucker *et al.*, 1997).

252. As well as those plants already present in the soil seed bank becoming established in SRF, the propagules of invasive short-lived perennial species characteristic of disturbed habitats are likely to be dispersed into the stand (Dzwonko, 2001; Wulf, 2004). These plants, most of which are widespread throughout much of the UK, are again not well suited to the undisturbed, low-light conditions under the plantation canopy, but are likely to become established in sun-spots and along rides. Examples of such species are: *Rubus fruticosus*, *Glechoma hederaceae*, *Geum urbanum*, *Poa trivialis*, *Urtica dioica*, *Chamaenerion angustifolium*, *Ranunculus repens*, and *Cirsium vulgare*.

(Peterken, 1993) gives a more complete list of such plant species commonly found in secondary woodlands.

253. Plants that are true woodland specialists, *i.e.* those adapted to competing for scarce resources in shaded habitats, tend to be very poor colonisers (Bierzychudek, 1982), and some secondary woodlands established centuries ago on farmland still do not contain the woodland specialists generally associated with ancient woodland (Peterken, 1993).

Examples of such taxa include long-lived perennials such as *Ranunculus acris*, *Ajuga reptans*, *Angelica sylvestris*, and *Veronica serpyllifolia*. Their inability to quickly colonise newly-created woodland habitats is unfortunate for a number of reasons:

- such taxa generally have low nutrient and water requirements, and therefore do not compete strongly with the tree crop;
- they are often attractive plants which, although not nationally rare, are appreciated by the general public;
- they are uniquely capable of providing a herbaceous understorey in low-light conditions;
- the understorey they create provides cover for woodland birds and mammals, and provides forage for most groups of animals, including mammals, birds and insects;
- this understorey also provides the soil with physical protection against erosion.

254. The single most important factor in determining the rate at which such taxa are able to colonise SRF is likely to be their level of abundance in the surrounding landscape (Dzwonko, 2001). SRF in areas devoid of ancient woody cover, such as woodland and hedgerows, are unlikely to be colonised by woodland specialist plants. SRF in habitats with abundant semi-natural ancient woodland are more likely to be colonised by such taxa, but the rate at which they become established can still be extremely slow (Peterken & Game, 1984; Sage, 1995).

255. Other factors affecting the ability of woodland specialist plants to colonise SRF are soil type and moisture status, and the species of tree planted (Persson *et al.*, 1989; Parrotta, 1995). Regarding the former, alkaline soils with available surface moisture are more suitable than dry soils with a low pH. The impact of the latter is caused by:

- the degree of shading caused by the tree canopy;
- the amount of available soil moisture;
- the physical and chemical attributes of the leaf litter.

2.2 Colonisation of rides and headlands

The species richness and abundance of non-crop plants will potentially be much higher in open areas associated with SRF, such as headlands and access routes, than within the stands themselves. If areas of pre-existing unmanaged ground are incorporated into the SRF, these areas probably already support diverse plant assemblages. Management

objectives for such areas are to prevent them 'shrubbing up', for which annual mowing in late summer is likely to be sufficient (J. Birks, pers. comm.).

257. Areas of the SRF to be left unplanted which were previously under intense agricultural management may or may not develop species-rich plant assemblages, depending on site characteristics and on previous management. Frequently, soil nutrient status has been improved through many years of fertiliser application, and this residual fertiliser effect creates conditions in which many of the rarer plant species typical of woodland rides and edges, and desirable for conservation, are unable to compete with the small number of ubiquitous species, often grasses, which are well adapted to rich soils (Tucker *et al.*, 1997).

2.3 Summary of Impacts on Plants

258. The understorey habitat beneath dense stands of SRF trees will provide habitat for a limited abundance and species richness of common and widespread plants, but can also provide suitable habitat for specialist woodland plants, in so far as they are able to colonise the area. The rides and other open areas associated with SRF will provide valuable habitat for species-rich assemblages of plants typically associated with woodland edges and rides, especially if the issue of soil eutrophication can be managed. Compared with both cropland and improved grassland, the SRF stand is likely to contain a greater abundance and diversity of non-crop vascular plants.

259. Apart from a small number of bryophytes, no particularly rare or threatened plants are likely to benefit from the establishment of SRF. Most red-listed plants in the UK are either associated with sunny, open habitats in southern England, and are rare because the UK represents the northern limit of a larger European population; or they are plants adapted to wetlands, most of which have been drained during previous centuries.

3. Non-vascular Plants and Fungi

260. There are many hundreds of species of non-vascular plants and fungi in the UK, many of which remain poorly studied. The main taxonomic groups are the bryophytes (consisting of the mosses, liverworts and hornworts), the fungi and the lichens. In general, little research has been done on the impacts on these groups of establishing farm woodlands, or SRC willow, on previous agricultural areas. Hence, few clues are available as to how they might be affected by SRF.

261. However, many non-vascular plants prefer undisturbed habitats and are typically associated with trees. The trunks of SRF trees could support a range of epiphytic bryophytes, including the red-listed moss *Atrichum angustatum*. Their windborne reproductive spores are dispersed over long distances, and they are capable of quickly colonising suitable new habitats (C. Cheffings, pers. comm.). The liverwort *Lejeunea mandonii*, also red-listed, may benefit from the elevated moisture and shade provided by

SRF. The Lesser smoothcap moss (*Atrichum angustatum*), of similar conservation concern, is likely to be well-adapted to the understorey habitat of SRF. There are no red-listed fungi or lichens known to be particularly associated with woodland .

4. Effect of SRF on Mammals and Other Vertebrates

262. The establishment of SRF in an agricultural landscape can potentially benefit most species of mammal due to the provision of additional cover by the tree crop and by the herbaceous vegetation associated with unplanted zones. Much like set-aside, these zones will also provide forage for both large and small mammals, and cover for smaller species.

4.1 Small mammals and their predators

263. The suitability of the understorey habitat for small mammals depends mainly on the abundance of understorey vegetation (Christian *et al.*, 1998). Stands with sparse understorey vegetation will provide neither cover nor forage for small mammals, but if a degree of understorey cover is achieved, species such as field vole, bank vole, common shrew, and wood mouse will find cover and forage. Irrespective of the quality of the understorey habitat, these mammals will benefit from the cover and relative lack of disturbance (when compared with agricultural land) of the unplanted zones associated with the stand. Red-listed species such as dormouse and red squirrel are unlikely to find suitable habitat in SRF due to the absence of the mature trees required for nesting sites.

264. Mammalian predators such as foxes, badgers, polecats, stoats, and weasels will be attracted by the presence of prey species, by the relative habitat stability, and by the provision of cover, from whence they can also make foraging trips into the surrounding landscape (J. Birks, pers. comm.).

4.2 Hares and rabbits

265. The red-listed brown hare and common rabbits will benefit from the relative lack of disturbance caused to the habitat and from the increase in landscape structural complexity. They will use the understorey at least as cover, if not as a breeding area. This could lead to increased numbers particularly of rabbits, which will require culling if damage they cause to adjacent crops is significant.

4.3 Bats

266. Three red-listed bat species, namely the pipistrelle bat and the greater and lesser horseshoe bats are all likely to benefit from the insects attracted to the SRF and associated unplanted zones (R. Isted, pers. comm.). The absence of mature trees (and hence nesting sites) will make the habitat unsuitable as a breeding area for bats.

4.4 Larger mammals

267. Whether or not large herbivores (deer, and in some areas, wild boar) will benefit from SRF will depend on the degree to which the stand is fenced (J. Birks, pers. comm.). SRF surrounded by deer fencing which is well maintained for the length of the rotation will provide no benefits to large herbivores; on the contrary, if previously unfenced areas of rough ground are incorporated into the fenced-off SRF, total available forage area in the landscape will decrease. If the stand is not deer-fenced the trees will provide harbourage for deer and enable them to forage in the surrounding agricultural landscape. Depending on tree species and on the development of understorey vegetation, the planted area may also provide forage, as will the unplanted zones.

4.4.1 *Impact of deer*

268. The population size of all but one of the UK's deer species is increasing (DEFRA, 2005). Foresters and conservationists are increasingly aware of the negative effects of heavy deer browsing in wooded habitats: these include prevention of natural tree regeneration, browsing of planted trees, damage to the stems of young trees, and reduction of the structural complexity of the understorey (Baines *et al.*, 1994; Rambo & Faeth, 1999; Fuller & Gill, 2001; Stewart, 2001).

269. In addition, deer damage farmers' crops, and are the cause annually of an estimated 20,000 - 40,000 road traffic accidents (<http://www.dcs.gov.uk>). However, exclusion of deer from stands of SRF is likely to further reduce the rate at which shade-tolerant plants are able to colonise the area, as in some cases deer act as seed vectors for such species (Vellend *et al.*, 2003; Mouissie *et al.*, 2005).

4.4.2 *Impact of wild boar*

270. Wild boar are now firmly established in southern England, and lack of cover is one of the key factors slowing their expansion to other areas of the country (J. Birks, pers. comm.). Determined boar are unlikely to be deterred by most fences (see website location <http://www.britishwildboar.org.uk/issues1.htm>), suggesting it is unlikely to be cost effective to try and exclude boar from SRF. Furthermore, wild boar seldom damage trees and their grubbing activity favours the establishment and spread of understorey plants.

4.5 Birds

4.5.1 *Interaction of SRF and Bird Populations*

271. This section evaluates the likely impact of SRF plantations on the abundance, composition, and species richness of bird assemblages in the area. Many UK farmland bird species have suffered a decline in range and abundance during the last 30 years, although in recent years there is some evidence this trend is reversing. It is noteworthy that the RSPB regards climate change as the biggest long-term threat to British birds (Eaton *et al.*, 2005).

272. Some bird species associated with farmland require open, treeless landscapes for nesting and foraging. Such birds include the skylark, corn bunting, stone curlew, and corncrake, all of which are red-listed species. A study of short-rotation willow coppice found that although some such species frequented plantations during the first year after establishment, they abandoned the area after the second year of tree growth (Sage, 1996). Clearly, then, areas of farmland which currently support threatened farmland bird species adapted to open habitat should not be converted wholesale to SRF.

273. Similarly, woodland birds associated with mature trees or decaying wood, such as nuthatch and marsh tit, are equally unlikely to colonise SRF because the trees are harvested before they begin to provide the habitat required by such species (Sage, 1998). It is wrong to assume that SRF will provide a suitable habitat for all or even most of the bird species associated with mature semi-natural woodland.

274. However, a number of rare British birds are almost exclusively associated with plantations. Siskin, common crossbill and goshawk are examples of species the abundance of which has increased in areas of large spruce plantations in upland Britain. Non-native tree plantations in lowland Britain have facilitated growing populations of firecrest and golden oriole (Fuller, 1997).

275. Furthermore, other species of farmland bird, as well as birds adapted to the early successional stages of woodland, traditionally managed coppice, and scrub, are likely to benefit from the structural diversity, habitat stability and foraging opportunities provided by the addition of SRFs to an agricultural landscape. Indeed, bird assemblages in landscapes containing SRF are likely to contain more species and greater numbers of birds than in purely agricultural landscapes. This is the finding of a review on the effects of bio-energy crops on farmland birds made by the RSPB (Anderson *et al.*, 2003), a finding that to a large degree is also applicable to the less transient habitat provided by SRF.

276. In general, key factors affecting the abundances of all bird species are the availability of suitable nesting sites and the abundance and accessibility food in summer and in winter (Anderson *et al.*, 2003). Bird species typically associated with hedgerows, young woodland and coppice are likely to value SRF for the provision of visual cover and its milder micro-climate.

277. Among others, the following species of conservation concern have been recorded in SRC willow during the non-breeding season, and are likely also to frequent SRF: kestrel, grey partridge, common snipe, woodlark, dunnoek, woodcock, song thrush and bullfinch (Coates & Say 1999, in Anderson *et al.*, 2004). Seed-eating birds are not likely to fare well in SRF, especially if understorey plant cover is sparse. Birds that feed on litter and

soil dwelling invertebrates, such as blackbird, thrushes, and woodcock are likely to find more suitable food in SRF than in adjacent agricultural landscapes. A woodcock, an amber-listed species which feeds mainly on worms and other invertebrates of the forest floor, was spotted during a visit to a *Eucalyptus* stand in Thetford forest (J. Mathews, pers. obs.).

278. During the breeding season, birds found in SRC willow and likely also in SRF are mostly common and widespread species of low conservation concern. These include pheasant, wren, robin, blackbird, sedge warbler, and chaffinch. Records of some species of higher conservation concern, such as reed bunting, song thrush and nightjar, have also been made (Anderson *et al.*, 2003 and references therein). The plantation edge, especially where there is a gradual transition from the tree crop to the surrounding agricultural landscape, is likely to provide a habitat suited to species such as yellowhammer, curl bunting and corn bunting, which are of greater conservation concern. Birds of prey, such as owls, will also find suitable habitat for hunting small mammals along the edges and rides of SRF plantations (Sage, 1998).

4.5.2 *Effects of different tree species on bird assemblages*

It is widely acknowledged by ornithologists that bird species richness and abundance is more influenced by local vegetation structure, habitat heterogeneity at the landscape level, and by site productivity than by the species identity of the trees (e.g. Fuller 1997). There are some exceptions to this rule, the most notable among them in the UK context being the dependence of the Scottish crossbill on Scots pine and the strong association of the golden oriole with planted stands of hybrid poplar.

280. Trees supporting greater quantities of insect are likely to be of more value to insectivorous birds than those supporting higher species richness but lower abundances. Sycamore, which is only mediocre in the diversity of insect phytophages associated with it, was nevertheless found to be actively favoured by birds gleaning insects from the canopy because of the high load of aphids it can support (Peck, 1989). In general, it appears the degree to which a tree species is native or exotic is a poor indicator of the abundance of potential insect prey foraging in its canopy. Rather, some native tree species, such as birch, tend to support abundant insect loads while other equally native species, such as yew, support very limited arthropod loads; this pattern is shared by exotic tree species, with some species, such as sycamore, being associated with heavy insect loads, and others, such as *Rhododendron ponticum*, being associated with very low insect abundances (Ward & Spalding, 1993; Welch & Greatorex-Davies, 1993; Yela & Lawton, 1997).

281. Relatively dense foliage can be of advantage to nesting birds, as it provides greater visual and physical cover, thus reducing the risk of predation and offering more protection from the elements. Non-deciduous species, such as *Eucalyptus*, will offer

greater cover during winter. Conversely, dense tree canopies, such as those of sycamore and *Eucalyptus* spp. are usually associated with very sparse vegetation cover in the understorey, which is deleterious for birds that feed on understorey plants or depend on them for cover. Tree species that flush relatively late in the year, such as ash, are more likely to facilitate the development of abundant vernal understorey vegetation, especially, again as in the case of ash, if their canopy, once developed, allows appreciable light to pass through to the forest floor.

282. Flowers and fruits produced by some tree species can provide birds with important food resources. For example, birch and alder seed are major winter foods for siskins and redpolls (Smart & Andrews, 1985). However, given that most of the trees suggested for use with SRF do not reach sexual maturation until age c. 15 years (Savill, 1991), for the greater part of the rotation none (with the possible exception of birch) of the recommended tree species will provide such resources.

4.6 Non-mammalian vertebrates

283. No species of red-listed amphibians and reptiles are likely to be affected, either positively or negatively, from the establishment of SRF. However, other amphibians and reptiles are likely to benefit from the relative habitat stability of SRF compared with agricultural landscapes. For example, a stand of 7-year-old *Eucalyptus* spp. growing at Redmarley in Gloucestershire supports a healthy population of adders. These are probably attracted by the abundance of small mammals in the understorey (J. Mathews, pers. obs.).

5. Invertebrates

5.1 Species richness of invertebrates associated with SRF

284. Invertebrates are generally small animals, and on land are constituted by the arthropods, the molluscs (i.e. slugs and snails), and the worms. The arthropods include the insects, spiders, crustaceans (e.g. woodlice) and the myriapods (i.e. millipedes and centipedes). With over 30, 000 species native to the UK (Kirby, 2001), the invertebrates contribute greater species diversity than the vascular plants, mammals and birds combined. Amongst them are many well-loved groups of species, such as the butterflies and moths, the beetles and the dragonflies. The invertebrates are deemed worthy of protection both for their own sake and because they provide food for species higher in the food chain, such as birds and mammals.

285. The molluscs and worms are taxonomically less diverse than the arthropods, and relatively few terrestrial species are listed as threatened under the UK Biodiversity Action Plan. However, both groups are likely to be more abundant and species-rich in newly created woodland than in farmland:

- Molluscs (in the main, snails) are not well suited to withstanding the disturbances caused by the soil cultivation of cropland, and are poorly adapted to quickly recolonise fields once annual crops have become established. The relative stability of the SRF system, combined with the higher humidity of the understorey environment, will provide habitat better suited to molluscs (Tousignant *et al.*, 1988).
- Worms are also likely to benefit from the reduced exposure of the SRF habitat to disturbances, e.g. ploughing and discing. In addition, worms are well adapted to the cool and moist understorey environment and benefit from the provision of large quantities of organic material in the form of leaf litter (Hubbard *et al.*, 1999; Whalen, 2004).

286. Mature woodland, with large canopy gaps, abundant veteran trees and coarse woody debris, provides habitat for more arthropods species than any other type of habitat (Speight & Wainhouse, 1989; Kirby, 2001). Stands of young trees, such as SRF, support fewer species because they lack:

- large mature trees, the architectural complexity of which provides more microhabitats than immature trees
- coarse woody debris, which provides habitat to the specialist xylophages and their associates
- gaps in the canopy, which permit the sun to warm the forest floor and provide suitable conditions for thermophilous species.

287. Nevertheless, in comparison with farmland, the canopies, understorey vegetation, litter-layer, rides and forest margins of SRFs are likely to provide better conditions for species-rich arthropod assemblages (Huhta & Niemi, 2003). The abundance and species-richness of farmland arthropods are greater in un-managed areas such as hedgerows and waysides than in the crop itself (Frank & Reichhart, 2004; Moreby, 2004). This is because:

- the crop habitat is ephemeral, with the crop being harvested and removed annually, and the soil being subject to the disturbance of cultivation. Every year, all but the few species that can persist in the soil must recolonise the crop habitat from elsewhere;
- most crops are sprayed annually with insecticide;
- most crops are planted as monocultures, and herbicides and fungicides are applied to control non-crop plants; this reduces the availability of non-crop food and structural diversity for arthropods.

288. In contrast, SRFs are managed on a much longer cycle, and the year-round stability and continuity of the habitat they provide will enable a continuous increase in species richness as new species colonise the area. It is generally not economical to apply pesticides to tree crops (although outbreaks of willow beetle on SRC willow have

occasionally been treated), and although the trees are likely to be planted as a monoculture, the understorey habitat is open to colonisation by appropriately adapted plants.

289. Further reasons why SRF plantations are likely to support greater species richness and abundance of arthropods than farmland are their greater overall biomass and greater structural complexity. The greater biomass of SRF can be viewed as greater food availability to phytophagous arthropods, and the year-round structural complexity contributed by the tree stems, understorey vegetation and litter layer provide microhabitats for a greater range of arthropod species (Nicolai, 1986; Maudsley, 2000). Tree trunks, even those of young trees, can be colonised by epiphytes (C. Cheffings, pers. comm.), which themselves provide resources for a suite of arthropod species (Kirby, 2001).

290. Rare arthropods that are associated with woodland, and which might benefit from the establishment of SRF, are the Southern wood ant (*Formica rufa*), which requires access to aphid-bearing trees; the Hairy wood ant (*Formica lugubris*), which nests on woodland rides and needs access to trees; the Waved carpet moth (*Hydrelia sylvata*), which feeds on the canopy of, among others, alder and birch; and the Argent and sable moth (*Rheumaptera hastata*), which also feeds on birch.

291. As for the UK's rarer plants, many of the rarest arthropod species in the UK are thermophilous or denizens of wetlands, and hence will not benefit from the cool and moist conditions provided by forested habitats (Hamblen & Speight, 1995). For example, although concern is often raised concerning the reduced abundance of charismatic species such as butterflies, no red-listed butterflies, or indeed bees and wasps, are particularly associated with forests (BAP website). This said, they may benefit from habitats associated with plantations, such as rides or forest edges (Sage, 1998).

292. There is very little specific information in the literature regarding the species richness or abundance of invertebrates associated with individual crop species. For this reason, it was not possible to compare e.g. the species richness of wheat with that of existing plots of SRC willow, in order to obtain a more quantitative estimate of the likely impact of SRF on agricultural invertebrate assemblages.

5.2 Impact of different tree species on invertebrates

293. Although many invertebrates will feed on understorey plants, the largest potential source of food in SRFs will be the trees themselves, and in particular their leaves. Trees growing in Britain differ markedly in the species richness and abundance of phytophages they support, which in turn has a bearing on the abundance and diversity of predatory and parasitoid invertebrates associated with them.

294. Contrary to popular opinion, the degree to which a tree species is native or exotic has less bearing on the diversity of associated invertebrates than leaf chemical composition and the abundance of the tree and close relatives in the country (Southwood, 1961)(Kennedy & Southwood, 1984; Welch, 1997; Ozanne, 1999). Thus, for example, native trees such as holly and yew support a fraction of the number of species supported by exotics such as *Nothofagus* spp. (Welch & Greatorex-Davies, 1993). Some exotic species, however, such as for example *Rhododendron ponticum*, do have very impoverished suites of invertebrates associated with them. The numbers of insect species associated with the trees recommended for SRF are presented in Table 3 below.

Table 3 Numbers of plant-feeding invertebrates associated with each of the recommended tree species when grown in the UK.

Tree species	Native or exotic	Number of plant-feeding invertebrate species associated with tree	Reference
<i>Acer pseudoplatanus</i> – Sycamore	Exotic	99	Biological Records Centre*
<i>Alnus glutinosa</i> – Alder	Native	190	Biological Records Centre
<i>Betula pendula</i> – Birch	Native	192	Biological Records Centre
<i>Eucalyptus gunnii</i>	Exotic	??	No records available
<i>E. nitens</i>	Exotic	??	No records available
<i>Fraxinus excelsior</i> – Ash	native	101	Biological Records Centre
<i>Nothofagus nervosa</i> syn. (<i>N. procera</i>)	exotic	94	Welch 1997
<i>Nothofagus obliqua</i>	exotic	98	Biol Rec Centre & Welch 1997
<i>Populus nigra</i> – Black Poplar	native	138	Biol Rec Centre

* Records from BRC provided by David Roy.

295. No lists of the number of invertebrate species associated with *Eucalyptus* spp. trees in the UK or northwestern Europe have been unearthed despite intensive literature searches - probably because this research has not yet been undertaken. Despite this lacuna, there are several sources of evidence that give insights into the likelihood of *Eucalyptus* spp. developing species-rich invertebrate assemblages in the UK:

- In their native Australian ranges, *Eucalyptus* support similar numbers of phytophagous insects as do British trees in the UK, and *Eucalyptus* in Australia appear to suffer greater defoliation caused by insects than do British trees (Majer *et al.*, 2000; Strauss, 2001; Cunningham *et al.*, 2005).
- Since the 1860s, when *Eucalyptus* spp. were first introduced to New Zealand, the trees have steadily accumulated insect phytophages; 57 such species have now been recorded. Some of these species (31 species) are polyphagous species native to New Zealand which have adapted to the new host, while the remainder (26 species) are Australian species that been accidentally introduced into the country (Withers, 2001). The rate at which insect species native to Australia will be able to colonise *Eucalyptus* spp. planted in the UK is likely to be markedly slower, given the greater distance to, and reduced trade contact with, their country of origin.
- The larger the area planted with *Eucalyptus*, the greater the number of native phytophagous species likely to adopt them as hosts (Kennedy & Southwood, 1984; Brandle & Brandl, 2001).
- Unlike *Nothofagus*, which has widespread and abundant 'cousins' from the same plant family (Fagaceae) in the UK, including *Fagus sylvatica* and *Quercus* spp., *Eucalyptus* spp., as members of the Myrtaceae family, have no such close relatives in the UK. This is likely to render it more difficult for native phytophages to become adapted to *Eucalyptus*.

296. Therefore, despite the lack of direct evidence, it is probable that with time:

- Some native phytophages will become adapted to feeding on *Eucalyptus* spp.;
- A small number of adapted species from abroad will be accidentally introduced; and
- The diversity of phytophages supported by *Eucalyptus* will remain relatively poor compared to most widespread native trees.

6. Interaction of Hydrology and Biodiversity

297. Water use and quality of run-off likely to be associated with SRF must be compared with the agriculture it replaces. Depending on species and site, SRF crops are likely to use somewhat less water than SRC willow crops but broadly speaking, their impact on the hydrology of a site will be similar. A fuller discussion of this issue is presented in a report recently written for the Department of Trade and Industry (Hall, 2003).

298. After the initial year of establishment, as trees become taller and structurally more complex than agricultural crops, they intercept and subsequently evaporate a greater proportion of incipient rainfall, and thus reduce the net amount of water reaching the soil. In addition, their greater leaf area index (LAI) enables higher potential water uptake from the site. If the trees have no access to the water table and they are therefore dependent on soil water recharge via local precipitation, their water consumption is likely to be similar to that of agricultural crops in drier areas of the UK, but may exceed that of agricultural crops in areas of higher rainfall (Cannell, 1999).

299. At sites where trees are able gain access to perennial ground water not available to the more shallow-rooted agricultural crops, overall water extraction of the tree crop will be greater. Furthermore, quicker-growing tree species will extract more water than slower-growing tree species. This may need to be considered when making species choice (Calder, 1986). Net effects on hydrology of conversion from agricultural use to SRF production of biomass may:

- reduce percolation to aquifers;
- reduce plant-available surface water;
- reduce surface run-off from site.

300. The impact of these factors will be site specific. Reduced percolation to aquifers is likely to be problematic in areas where irrigation water is mined from the aquifer, but also reduces the potential for leaching of soil nutrients, which may represent a substantial benefit in nitrate vulnerable zones (Rijtema & Devries, 1994). In general, water use by SRF is likely to be higher than that of most agricultural crops, slightly higher than that for SRC willow, similar to that of broadleaved forests, and slightly lower than that of coniferous forests (Perry *et al.*, 2001).

301. The lack of available surface water may render understorey conditions too dry for some potential herbaceous colonisers; this effect will benefit the trees somewhat, due to reduced competition for moisture and nutrients, but will also reduce the biodiversity potential of the site. Reduction in surface run-off reduces the potential for soil erosion but also reduces yield to downslope streams. In some areas of the country this may be undesirable while in others it will be beneficial, depending on the downstream benefits of stream flow (e.g. potential for flooding, presence of ecologically valuable areas of wetland etc.). Overall, surface soil erosion is not considered to be a major risk, given the topography of most potential SRF sites (Ranney & Mann, 1994; Lugo, 1997; Kort *et al.*, 1998; Malik *et al.*, 2000).

302. The important overall consideration is whether the water is of greater value in the ground, or as run-off, or consumed in order to produce woody biomass (Cossalter & Pye-Smith, 2003).

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A review of the growth, yield and biomass distribution of species planted in the English network trials of Short Rotation Forestry

Gary Kerr

Summary

There has recently been an increase in the interest in short rotation forestry (SRF) as a way of producing woodfuel. Short rotation forestry aims to produce single-stemmed trees of between 10 and 20 cm diameter at breast height on a rotation of *circa*. 10 years. Any environmental or economic analysis of SRF requires accurate estimates of growth and yield; however, at present information is limited. The objective of this review is to provide the most authoritative information available on the growth, yield and biomass distribution for British conditions of species planted in the network trials of short rotation forestry.

The two main findings of the review are: (1) at present the quantity and quality of growth data are inadequate to underpin the development of SRF in Britain, and (2) the relationship between productivity and risk has not been evaluated for the range of non-native species being considered for SRF. The latter is particularly important for eucalypts and *Nothofagus* due to their sensitivity to unseasonal frost and winter cold. Using a combination of published information and new data located as part of the review process, the following are the best estimates of the range of possible biomass productivity:

Species	Biomass (oven dry tonnes ha ⁻¹ yr ⁻¹)
Tingiringi gum (<i>Eucalyptus glaucescens</i>)	2.5 – 7.6
Cider gum (<i>Eucalyptus gunnii</i>)	1.5 – 8.2
Rauli (<i>Nothofagus alpina</i> syn. <i>N. procera</i>)	3 – 10.5
Sweet chestnut (<i>Castanea sativa</i>)	1.2 – 6.0
Red alder (<i>Alnus rubra</i>)	0.9 – 4.8
Ash (<i>Fraxinus excelsior</i>)	0.5 – 4.7
Silver birch (<i>Betula pendula</i>)	0.5 – 5.7
Sycamore (<i>Acer pseudoplatanus</i>)	0.5 – 5.7

The species fall into two distinct groups, Tingiringi gum, cider gum, and rauli in a band of potentially high productivity and sweet chestnut, red alder, ash, birch and sycamore in a band of lower productivity. Unfortunately, it was not possible to locate any data or information on which to base estimates for Italian alder, hybrid aspen or shining gum (*Eucalyptus nitens*). The review has located some new data on the relationship between spacing and productivity for *Eucalyptus glaucescens* and *Eucalyptus gunnii* and has highlighted the need for further work to define appropriate silvicultural systems for SRF in Britain.

1. Introduction

Present government policy is to increase the amount of energy that is derived from renewable sources. This policy offers considerable potential for the development of woody biomass as a source of energy (McKay, 2006). In the past the development of woody biomass crops has focussed on short rotation coppice (SRC). This involves growing multi-stemmed woody material over short rotations, usually of less than five years. Yet despite SRC having relatively high rates of growth and being productive on short rotations, material used in SRC, such as willow (*Salix* spp.) and poplar (*Populus* spp.), do not meet six other criteria for an ideal fuelwood as suggested by Ramsay (2004) (cited in Cope *et al.*, 2008). The criteria are:

1. Produce high density wood
2. Have suitable chemical characteristics
3. Exhibit low moisture content
4. Be easily harvested
5. Be harvested using conventional machinery
6. Be capable of being harvested all year round.

Short rotation coppice of willow and poplar produces small diameter material with a high moisture content, low wood density and high bark content, which can produce corrosive substances when burned. It is harvested using converted agricultural machinery and this limits harvesting to periods when the soil is relatively dry. Because of these problems with SRC, attention has turned to the use of short rotation forestry (SRF). This differs from SRC in that the material is single-stemmed and the rotation is normally longer, usually being harvested between 8 and 20 years old and yielding material of between 10 and 20 cm diameter at breast height (DBH) (Hardcastle, 2006). An additional advantage of SRF is that it can be harvested using conventional forestry machinery.

Short rotation forestry is a challenge to traditional silviculture In Britain in a number of ways:

1. Rotations of 10-20 years are much shorter than traditional rotations of even-aged stands, which are generally 50-60 years for conifers and longer for broadleaved species.
2. There is a limited amount of information available for some species advocated for SRF (see Table 1 of Hardcastle (2006)).
3. SRF advocates the use of more intensive silviculture during the establishment phase, for example using a combination of plastic mulching and herbicides (Hardcastle, 2006). Any growth data currently available for SRF species is based on conventional establishment practice rather than this more intensive approach.
4. Ex-agricultural SRF sites are likely to differ from traditional forestry sites where the majority of growth data have been collected.

Table 1 Yield information presented in Hardcastle (2006)

Species	Biomass (odt ha ⁻¹ yr ⁻¹)	Rotation (yr)
Alder	5.0	20
Ash	7.4	20
Birch	5.0	20
Poplar	5.6	14
Sycamore	7.0	20
<i>E gunnii</i>	9.0	12
<i>E nitens</i>	15.0	8
<i>Nothofagus</i>	11.8	12
<i>SRF Species</i>		
Willow / Poplar	8	3

These constraints are apparent in three recent examples of the presentation of growth and yield of SRF.

1. A recent review of the potential impacts of SRF by Hardcastle (2006) contained a table of yield information for species that could be grown as SRF (reproduced as Table 1). The basis of the data is 'through discussion with current practitioners and [the figures] are based on very limited field data, on sites probably more favourable than may be achieved on a wide scale and certainly with very close supervision'.

2. An internal report for the Forestry Commission by Kerr (2003) attempted to produce indicative figures for the biomass production of SRF using published information (reproduced as Table 2). The estimates are constrained by existing growth and yield data and a basic method was used for taking account of the fact that SRF may use more intensive silviculture during establishment and use different sites.
3. The recent woodfuel strategy produced by Forestry Commission England (Forestry Commission, 2007) contained a table comparing total carbon savings for various silvicultural options (Table 7 in Annex 1 on p.26). This indicated that SRF using eucalypts is an attractive option compared with SRC willow, SRF ash, native woodland and conifer woodland, in terms of the amount of carbon sequestered and the greenhouse gas emissions avoided. However, it is unclear how these figures were derived.

It is evident from the above that accurate information on growth and yield is essential for any type of economic or environmental analysis of SRF. The shortcomings in the above examples are clear. Firstly, the Hardcastle (2006) report states that the figures are based on limited data, because of this they are questionable and will not help to build confidence in this new type of silviculture. Secondly, the review by Kerr (2003) is arguably the soundest as growth and yield estimates are based on published information, but the attempt to indicate how intensive silviculture could increase productivity was rather crude. Finally, the source of the data is not shown in the FC England Woodfuel Strategy.

The objective of this review is to provide the most authoritative information available on the growth, yield and biomass distribution for British conditions of species planted in the network trials of short rotation forestry. It builds on the earlier review (Kerr, 2003) by including previously unpublished data from Forest Research. Although it is possible that some species coppice sufficiently well to be used for several cycles this review does not address the yield in later coppice cycles because there is even less information available than for the first harvest.

Table 2 Indicative figures for biomass production of short rotation forestry from Kerr (2003)

Species	GYC	Rotation	No. trees ha ⁻¹	Mean dbh (cm)	Volume (m ³ ha ⁻¹)	Basic density	Biomass odt ha ⁻¹ yr ⁻¹	'MS' biomass ¹ odt ha ⁻¹ yr ⁻¹	Source
(a) Estimates based on published yield tables									
Cherry	9	20	1950	12	117	0.50	2.9	3.9	Pryor (1988)
SAB ²	10	20	(664) ³	(14.9)	133	0.50	3.3	4.4	Edwards and Christie (1981)
Oak	8	20	6025	6.9	73	0.56	2.0	2.7	
Corsican pine	16	20	(1747)	(13.9)	184	0.40	3.7	4.9	
Western red cedar	18	20	3592	12.0	134	0.32	2.1	2.9	
Nothofagus	16	18	(912)	(17)	265	0.60	8.8	11.8	Tuley (1980)
Sweet chestnut ⁴	-	20	-	-	160	0.55	4.4	- ⁴	Rollinson and Evans (1987)
(b) Estimates based on readily available plot data									
Poplar	-	16	2250	13	185	0.36	4.2	5.6	Edwards and Christie (1981)
Eucalyptus	-	8	1200	22	298	-	13.5	-	Purse and Richardson (2001)
Eucalyptus	10-14c ⁵	10-12	-	-	-	-	10-15	-	Evans (1986)
Ash	8-10	4	40000	-	56	0.53	7.4	- ⁴	Table 3 – in Kerr (2003)
Ash	-	20	2074	9.7	68	0.53	1.8	2.4	Evans <i>et al.</i> (2002)
Ash	-	20	2600	9.8	58	0.53	1.5	2.0	

¹ Production using modern establishment silviculture ('MS') has been estimated by assuming that the same biomass could be achieved on 75% of rotation

² The combined yield table for sycamore, ash and birch in Edwards and Christie (1981)

³ Where brackets are shown thinnings have occurred before the age shown; figures in brackets show the stand composition after thinning but cumulative volume (including volume removed) is shown.

⁴ This is coppice or direct sowing data; full site utilisation is achieved quickly and this may indicate the upper limit of production – therefore no 'MS' figures are quoted

⁵ GYC range for conifers

2. Methods

The species included in the review were divided into three categories (Table 3) that were aligned with the priorities of the review. Hence most attention was given to eucalypts, then other species without a yield table, and lowest effort in searching for new information was given to the species with a published yield table. The main methods for conduct of the review were:

1. Searching for published papers and any 'grey' literature that was known to exist for any useful information.
2. Searching the experimental database in Forest Research (FR) for experiments or plots that may have valuable data.

Table 3 Short rotation forestry species and rotations considered in this review

Common Name	Scientific Name	Rotation
Species with no published yield table – eucalypts		
Shining gum	<i>Eucalyptus nitens</i>	10
Tingiringi gum	<i>E. glaucescens</i>	10
Species with no published yield table – others		
Italian alder	<i>Alnus cordata</i>	15
Red alder	<i>Alnus rubra</i>	15
Hybrid aspen	<i>Populus tremula x tremuloides</i>	15
Species with a published yield table		
Sycamore	<i>Acer pseudoplatanus</i>	20
Silver birch	<i>Betula pendula</i>	20
Ash	<i>Fraxinus excelsior</i>	20
Rauli	<i>Nothofagus alpina</i> (syn. <i>N. procera</i>)	15
Sweet chestnut	<i>Castanea sativa</i>	20

To calculate the volume of an individual tree from height and diameter data for eucalypts the formula from Purse and Richardson (2001) was used:

$$\text{Volume} = (\text{dbh}/100)^2 \times \text{height} \times 0.35$$

Where: volume is in m³; dbh is in centimetres; height is in metres

The conversion from volume to oven dry tonnes (odt) ha⁻¹ assumed a basic density of 0.45 for eucalypts; values for all other species are specified. **Any mention of biomass in the report assumes oven dry tonnes.**

Previously unpublished growth data were examined using analysis of variance. Analysis of survival data used a generalised linear model with a binomial distribution and a logit link function; in each case the dispersion parameter was estimated. A significant effect was defined as $p \leq 0.05$. In addition, to compensate for lack of replication in the trials, effects were also examined in terms of the percentage of total variation or deviance accounted for by different effects. All statistics were carried out using Genstat version 10 (Anon, 2007).

The relationship between the existing data and top height-age curves was examined for ash. Some care is required here because for ages of <20 the curves were produced by extrapolations from the main body of the data, which were generally for older stands. Some of the available data have been collected after the height-age relationships were defined, especially 'increment plots' (see Evans *et al.* (2002)), and the objective was to see if there was any indication that the current maximum GYC (10) needs to be revised to take account that the species has been planted on more productive sites in the recent past.

The final process of the review has been to compile a likely productivity range for each of the species considered. To do this the following criteria were used:-

1. The review must have produced a reasonable amount of evidence and plot-based data on how the species will grow in Britain.
2. For species with a published yield table the productivity range is defined by the extent of site productivity in the yield tables.
3. For species with no published yield table the productivity range is defined as the median of the lowest and highest productivities from each site that has produced useful data.

Figures from the report that have been used to formulate the range of productivity for a species are shown in **bold** in Tables 4-7 and 11-15.

3. Results

Species with no published yield table - eucalypts

The available information on eucalypts originates from four main sources:

1. The published papers of Evans (1980; 1986).
2. Recent published papers that have reassessed some of the trials established by the Forestry Commission in the 1980s.
3. Unpublished data from the trials in (2) above.
4. Other published and unpublished information.

The published papers of Julian Evans

The work of Julian Evans in the 1980s (Evans, 1980; 1986) is well known to many people interested in SRF. It reports the results of a series of tests of 102 species, provenances and seed origins located throughout Britain. The establishment of the trials coincided with an exceptionally cold winter in 1981/2 in which trial sites experienced temperatures between -19°C and -23°C, which killed trees of all species planted. The two main findings of the study were:

1. *Eucalyptus gunnii* and the snow gums *E. debeuzevillei* and *E. niphophila* possessed enough cold-hardiness to survive and were judged to have the greatest potential in Britain.
2. Eucalypts could be considered to produce short fibre material on moderately exposed upland sites where yield classes of 12 to 16 on 10-year rotations could be expected (i.e. *circa.* 5.4-7.2 odt ha⁻¹ yr⁻¹).

Eucalyptus nitens and *E. glaucescens* were included in the trials. *E. nitens* showed better growth but was found to be less cold hardy than *E. gunnii*; this remains the judgement of the author (J. Evans, *pers. comm.*) and agrees with published information on tolerance to winter cold in Booth and Pryor (1991)¹. *E. glaucescens* was found to be less cold hardy than *E. gunnii*, this also remains the judgement of the author, although his observation is that the stem form of the former is generally superior; although stem form may not be as important in SRF compared with traditional forestry (J. Evans, *pers. comm.*).

Recent published papers that have reassessed some of the trials established by the Forestry Commission in the 1980s

Due to the increased interest in SRF some of the trials of eucalypts established by the Forestry Commission in the 1980s have been reassessed and accounts published in an

¹ This paper quotes >-14°C as the absolute minimum for *E. gunnii* and >-12°C for *E. nitens*.

attempt to provide more objective information. The first of these, Thetford 206 planted in 1981, has been reassessed 21 years later by Bennett and Leslie (2003).

Their results show that four seedlots of *E. gunnii* and one of *E. glaucescens* had good early survival, although it is clear that deaths continued until 2002 (Table 4). Results indicate a range of biomass productivity of between 2.2 and 8.2 odt ha⁻¹ yr⁻¹, although this is likely to have been compromised by only moderate survival of between 31% and 53% over the 21-year period.

Table 4 Data for *E. gunnii* and *E. glaucescens* from Thetford 206/81 trial after 21 years

Species and seedlot	Survival (% to year 2)	Survival (% to year 21)	Volume ¹ (m ³ ha ⁻¹ yr ⁻¹)	Biomass ¹ (odt ha ⁻¹ yr ⁻¹)
<i>E. gunnii</i> (3)	83	42	13.5	6.1
<i>E. gunnii</i> (5)	83	53	18.2	8.2
<i>E. gunnii</i> (8A)	83	50	14.2	6.4
<i>E. gunnii</i> (16)	89	33	4.9	2.2
<i>E. glaucescens</i> (34)	78	31	16.2	7.3

¹ Low stocking may have compromised productivity

The second trial to be reassessed was Glenbranter 19, which was planted in 1981 and results are reported in Cope *et al.* (2008). The results from this trial for *E. gunnii* have been divided into three seedlots that survived the 1981/82 winter and subsequently showed reasonable volume growth, and seven seedlots that did not tolerate the cold winter in 1981/82 but after subsequent beating-up showed good volume production to year 25 (Table 5). The range of biomass production of the first group was 1.0 to 8.1 odt ha⁻¹ yr⁻¹ and for the second group 4.2 to 11.8 odt ha⁻¹ yr⁻¹ (Table 5). The results for *E. glaucescens* (Table 6) indicate the early survival was poor for all seedlots but, after replacement of dead trees, productivity could be in the range 3.6 to 13.3 odt ha⁻¹ yr⁻¹ if the new trees survived. If they did not, for example seedlots 143 and 144, productivity was poor (<1 odt ha⁻¹ yr⁻¹).

Table 5 Data for *E. gunnii* from Glenbranter 19/81 trial after 25 years

Seedlot	Diameter (cm)	Height (m)	Volume (m ³ ha ⁻¹ yr ⁻¹)	Biomass ¹ (odt ha ⁻¹ yr ⁻¹)
(a) Good survival to year 3 and good volume production to year 25³				
5	14.2	10.5	2.2	1.0
100-104 ²	20.1	17.8	18.1	8.1
106	12.2	18.2	2.7	1.2
(b) Good survival after year 3 and good volume production to year 25⁴				
1	20.7	18.7	16.5	7.4
9	19.2	14.7	11.9	5.4
11	16.2	15.8	9.4	4.2
16	20.0	21.0	16.9	7.6
100-104 ²	20.1	17.8	18.1	8.1
111	20.6	17.5	17.6	7.9
113	24.5	16.9	26.2	11.8

¹ Assuming a basic density of 0.45

² The mean from data presented by Cope *et al.* (2008)

³ Seedlots that tolerated the exceptionally cold winter of 1981/2

⁴ Seedlots that did not tolerate the cold winter of 1981/2 and were subsequently beaten-up

Table 6 Data for *E. glaucescens* from Glenbranter 19/81 trial after 25 years

Seedlot	Survival (% to year 2 ¹)	Survival (% to year 25 ²)	Volume (m ³ ha ⁻¹ yr ⁻¹)	Biomass (odt ha ⁻¹ yr ⁻¹)
142	3.0	100	29.5	13.3
143	0.8	31	0.1	0.05
144	1.3	29	2.0	0.9
145	43.1	69	7.9	3.6
146	7.0	82	12.4	5.6
147	26.4	71	21.9	9.9

¹ Data from Evans (1986) for four sites including Glenbranter

² Following beating-up after the earlier deaths

Unpublished data from trials established by the Forestry Commission in the 1980s

During the search for useful data an experiment was located, which had been replicated at four sites in southern England, and the data for *E. gunnii* and *E. glaucescens* have not previously been published. The objectives of the experiment were to: (1) to identify the fastest growing broadleaved species and evaluate their potential for use in southern

Britain², and (2) to compare the effects of close and wide spacing (1.4 x 1.4 m and 2.8 x 2.8 m). The trials were planted in 1980 and therefore survived the very cold winter of 1981/2. Beating-up of the eucalypts was only carried out at one of the four sites (Bedgebury 26) due to early establishment problems and this site produced little useful data. The sites that have produced useful data are New Forest 38 (survival, height and diameter after seven years; Table 7), Ringwood 32 (survival and height growth for 5 years; Table 8) and Neroche 17 (survival and height growth for 2 years; Table 8).

Table 7 Data for *E. gunnii* and *E. glaucescens* from New Forest 38/80 after 7 years

Species and seedlot	Survival (%) to year 6 ²	Height (m)	Volume (m ³ ha ⁻¹ yr ⁻¹)	Biomass ¹ (odt ha ⁻¹ yr ⁻¹) ³
Close spacing (1.4 m x 1.4 m)				
<i>E. gunnii</i>	89	10.5	14.1	6.3
<i>E. gunnii</i>	94	10.8	20.9	9.4
				7.9
<i>E. glaucescens</i>	66	11.5	18.5	8.3
<i>E. glaucescens</i>	40	10.4	15.1	6.8
				7.6
Wider spacing (2.8 m x 2.8 m)				
<i>E. gunnii</i>	89	9.1	3.3	1.5
<i>E. gunnii</i>	69	9.4	3.2	1.4
				1.5
<i>E. glaucescens</i>	69	11.7	6.1	2.7
<i>E. glaucescens</i>	54	9.7	5.1	2.3
				2.5

¹ Assuming a basic density of 0.45

² Analysis showed that the only significant effect was between species (p=0.022)

³ Analysis showed that the only significant effect was spacing (p=0.011)

² up to seven species were used at each site

Table 8 Data for *E. gunnii* and *E. glaucescens* from Ringwood 32/80 after 5 years and Neroche 17/80 after 2 years

Species and seedlot	Ringwood (5 years)		Neroche (2 years)	
	Survival (%) ¹	Mean height (cm) ²	Survival (%) ³	Mean height (cm) ⁴
Close spacing (1.4 m x 1.4 m)				
<i>E. gunnii</i>	92	184	78	69
<i>E. gunnii</i>	94	427	98	87
<i>E. glaucescens</i>	65	190	76	110
<i>E. glaucescens</i>	63	201	52	110
Wider spacing (2.8 m x 2.8 m)				
<i>E. gunnii</i>	98	227	90	73
<i>E. gunnii</i>	90	439	90	75
<i>E. glaucescens</i>	82	225	73	74
<i>E. glaucescens</i>	80	216	58	108

¹ Analysis showed that the only significant effect was species ($p=0.010$); spacing accounted for 13.4% of total deviance but was not significant ($p=0.092$)

² Analysis showed no significant effect but species accounted for 95.2% of total variance but was not significant ($p=0.19$)

³ Analysis showed no significant effects but species accounted for 62.7% of total deviance but was not significant ($p=0.102$); data was over-dispersed (dispersed parameter estimated to be 6.08)

⁴ Analysis showed no significant effects but species accounted for 51.7% of total variance but was not significant ($p=0.054$)

The results from New Forest 38 are the most complete and it has been possible to estimate biomass production over the seven-year period (Table 7). Survival of *E. gunnii* was high (>85%) in three of the four plots. The survival of *E. glaucescens* was significantly lower than for *E. gunnii* with the four plots having a range of 40-69%. There was a statistically significant effect of spacing on biomass production with closer spacings being more productive. At close spacing *E. gunnii* produced $7.9 \text{ odt ha}^{-1} \text{ yr}^{-1}$, which was five times higher than at wide spacing ($1.5 \text{ odt ha}^{-1} \text{ yr}^{-1}$). Similarly, *E. glaucescens* at close spacing produced $7.6 \text{ odt ha}^{-1} \text{ yr}^{-1}$, this was three times higher than at wide spacing ($2.5 \text{ odt ha}^{-1} \text{ yr}^{-1}$).

The lack of diameter data at Ringwood and Neroche has meant that it was not possible to estimate biomass production. Survival data from both sites supports the trends from the New Forest site, i.e. that *E. gunnii* (range 90-98% for both sites) was higher than *E. glaucescens* (range 52-82%). However, the mean height increment for both species at Ringwood and Neroche is much lower than at New Forest (Table 9) and indicates much lower levels of productivity.

Table 9 Comparison of annual height growth for three sites

	Mean annual height increment (cm)		
	New Forest	Ringwood	Neroche
<i>E. gunnii</i>	166	64	38
<i>E. glaucescens</i>	180	42	50

The data that have been published from these trials can be found in Pearce (1985) and were for *E. archeri* at the New Forest 38 site. This indicated that on a rotation of four years the species could produce 8.6 odt ha⁻¹ yr⁻¹ at close spacing and 2.3 odt ha⁻¹ yr⁻¹ at the wider spacing³.

Other published and unpublished information

The following list contains some useful information found whilst carrying out the review.

- Cope *et al.* (2008) state that in France plantings of *E. gunnii* and *E. x gundal* (a hybrid of *E. gunnii* and *E. dalrympleana*) can produce 7.5 odt ha⁻¹ yr⁻¹ on a 12 year rotation.
- Purse and Richardson (2001) use a variety of information sources to justify a claim that in the UK certain eucalypts could produce 10-15 odt ha⁻¹ yr⁻¹ on many sites. The authors have also established a series of trials of SRF using eucalypts but information from these is difficult to locate. One source indicates a trial of *E. nitens* at Blackmoor, Hampshire showed 12-13 odt ha⁻¹ yr⁻¹ after 6 years. A figure of 10 odt ha⁻¹ yr⁻¹ after six years is also quoted for *E. nitens* at one of the other trials in Kent (www.primabio.co.uk/bm_trials.htm) although *E. gunnii* is noted to be 'substantially' less productive but no figures are given.
- Trials of *E. archeri*, a close relative of *E. gunnii*, grown as short rotation coppice produced between 2.5 and 14.2 odt ha⁻¹ yr⁻¹ depending on site, rotation and spacing (Potter, 1990).
- A report by Forrest and Moore (2008) indicated that *E. gunnii* could produce 12.6 odt ha⁻¹ yr⁻¹ on a site in Ireland.
- A recent large-scale planting of eucalypts at Daneshill, Retford, Nottinghamshire is worthy of further investigation
- Marriage (1972) reports impressive growth of *E. gunnii* and *E. glaucescens* but it is not possible to convert the data to an area basis.

³ The published values have been reduced by 13.3% to remove the biomass of leaves, using figures from the experiment file

Species with no published yield table - others

Red alder has been evaluated in Britain and a note summarising the published literature and FR experiments by Bill Mason is attached as Appendix 1. In summary, on good sites red alder has the potential to be as productive as the other native and naturalised species considered in this report. However, it is very sensitive to unseasonal frosts and this is the main factor limiting its use in Britain. The best results with the species have been recorded in mixture experiments in which the red alder may be protected by the other species (Table 17 in Appendix 1). The best data on the distribution of biomass between shoot and root for SRF species has been published by Proe *et al.* (2002) for red alder and shows 20-25% of total plant biomass being partitioned into the root (Table 10).

Table 10 Biomass production of red alder on a 5 year rotation

Spacing	Biomass production (odt ha ⁻¹ yr ⁻¹)	Root:shoot ratio
1.0 × 1.0 m	16.3	0.21 (after 4 yrs)
1.5 × 1.5 m	10.1	0.26 (after 3 yrs)

Data from Proe *et al.* (2002)

No useful information was located for hybrid aspen or Italian alder.

Species with a published yield table

The species that have a published yield table are ash, sycamore and birch in Edwards and Christie (1981), rauli in Tuley (1980) and sweet chestnut in Maw (1912). The early yield tables have been used for sweet chestnut because they are for planted crops and this was judged to be more appropriate for SRF. Later work on sweet chestnut coppice by Rollinson and Evans (1987) and Everard and Christie (1995) was strongly influenced by coppice and singled coppice. If these five species are planted on site types applicable for the yield table using intensive silviculture then ash, sycamore, birch and sweet chestnut could potentially produce between 4.7 and 6.0 odt ha⁻¹ yr⁻¹ (Tables 11-14); figures for rauli are higher at up to 10.5 odt ha⁻¹ yr⁻¹ (Table 15).

Table 11 Indicative figures for biomass production for ash grown on a 20-year rotation

GYC	Volume (m ³ ha ⁻¹)	Biomass ¹ (odt ha ⁻¹ yr ⁻¹)	'MS' biomass ² (odt ha ⁻¹ yr ⁻¹)
6	18	0.5	0.6
8	57	1.5	2.0
10	95	2.5	3.4
12 ³	133	3.6	4.7
14 ³	172	4.6	6.1
16 ³	208	5.5	7.3

¹ assuming basic density of 0.53

² Production using modern establishment silviculture ('MS') has been estimated by assuming that the same biomass could be achieved on 75% of rotation

³ grey shading indicates lower confidence of estimates, i.e. they have been extrapolated beyond the range of GYCs in the Edwards and Christie (1981) yield table

⁴ initial spacing is 4444 trees ha⁻¹

Table 12 Indicative figures for biomass production for sycamore grown on a 20-year rotation

GYC	Volume (m ³ ha ⁻¹)	Biomass ¹ (odt ha ⁻¹ yr ⁻¹)	'MS' biomass ² (odt ha ⁻¹ yr ⁻¹)
4	18	0.5	0.6
6	57	1.5	1.9
8	95	2.4	3.2
10	133	3.4	4.5
12	172	4.3	5.7
14 ³	208	5.2	6.9
16 ³	244	6.1	8.1

¹ assuming basic density of 0.50

² Production using modern establishment silviculture ('MS') has been estimated by assuming that the same biomass could be achieved on 75% of rotation

³ grey shading indicates lower confidence of estimates, i.e. they have been extrapolated beyond the range of GYCs in the Edwards and Christie (1981) yield table

⁴ initial spacing is 4444 trees ha⁻¹

Table 13 Indicative figures for biomass production for silver birch grown on a 20-year rotation

GYC	Volume (m ³ ha ⁻¹)	Biomass ¹ (odt ha ⁻¹ yr ⁻¹)	'MS' biomass ² (odt ha ⁻¹ yr ⁻¹)
4	18	0.5	0.6
6	57	1.5	1.9
8	95	2.4	3.2
10	133	3.4	4.5
12	172	4.3	5.7
14 ³	208	5.2	6.9
16 ³	244	6.1	8.1

¹ assuming basic density of 0.50

² Production using modern establishment silviculture ('MS') has been estimated by assuming that the same biomass could be achieved on 75% of rotation

³ grey shading indicates lower confidence of estimates, i.e. they have been extrapolated beyond the range of GYCs in Edwards and Christie (1981) yield table

⁴ initial spacing is 4444 trees ha⁻¹

Table 14 Indicative figures for biomass production for sweet chestnut grown on a 20-year rotation

Site quality (~GYC)	Volume (m ³ ha ⁻¹)	Biomass ¹ (odt ha ⁻¹ yr ⁻¹)	'MS' biomass ² (odt ha ⁻¹ yr ⁻¹)
IV (6)	43	1.2	1.6
III (8)	62	1.7	2.3
II (10)	134	3.7	4.9
I (12)	164	4.5	6.0
I+ (14) ³	208	5.2	6.9
I++ (16) ³	244	6.1	8.1

¹ assuming basic density of 0.55

² Production using modern establishment silviculture ('MS') has been estimated by assuming that the same biomass could be achieved on 75% of rotation

³ grey shading indicates lower confidence of estimates, i.e. they have been extrapolated beyond the range of site qualities in Maw (1912). These early yield tables have been used because they are for planted crops; later work by Rollinson and Evans (1987) and Everard and Christie (1995) was strongly influenced by coppice and singled coppice.

Table 15 Indicative figures for biomass production for rauli grown on a 15-year rotation

GYC	Volume ¹ (m ³ ha ⁻¹)	Biomass ² (odt ha ⁻¹ yr ⁻¹)	'MS' biomass (odt ha ⁻¹ yr ⁻¹)
10	90	3	4.1
12	116	3.9	5.3
14	153	5.1	7.0
16	192	6.4	8.7
18	230	7.7	10.5

¹ From the yield table in Tuley (1980) assuming no thinning and using linear interpolation for a 15 year rotation.

² assuming basic density of 0.50

With the effect of more productive sites (i.e. higher yield classes than the published yield tables) the range of biomass productivity could be increased up to between 7.3 and 8.6 odt ha⁻¹ yr⁻¹ for the first group of four species (Tables 11 to 14). More productive sites were not included for rauli because the published yield table already includes a GYC range of 10-18, which was assumed to be close to the maximum productivity (Danby, 1991). The relationship between the existing height-age curves for ash and existing data is shown in Figure 1. The rotation specified for ash in the review is 20 years (Table 3) and data in the range of 0-20 years indicate a close fit between the data and the existing relationships. However, between 20 and 30 years there is some evidence that the present maximum GYC is too conservative. Nevertheless, this evidence was not judged to be strong enough to justify increasing the productivity range for each species above that assumed in the published yield tables.

4. Discussion

The main point to emerge from this review is that the quantity and quality of growth data available at present are inadequate to underpin the development of SRF in Britain. To undertake any type of economic or environmental analysis of SRF requires the development of yield and biomass models that take account of the species and sites that are likely to be planted and the intensity of the silviculture that will be applied. The establishment of the English network trials is a tentative step in the right direction but in reality they can only partially meet the need for the development of better information on growth and biomass.

It is also clear from the available information that the relationship between productivity and risk has not been evaluated for the range of non-native species being considered for SRF. The Forestry Commission's project to evaluate eucalypts during the 1980s was

significantly affected by winter cold and unseasonal frosts (Evans, 1986). Similarly, the AFOCEL programme in France to establish SRF using eucalypts was abandoned in 1985 due to losses in the first commercial plantings caused by winter cold (Bennett and Leslie, 2003). Some work on the relationship between frost risk, productivity and economics has been attempted by Terreaux (2000) and it would seem prudent to undertake similar work in Britain.

As the figures in this review show the *potential* productivity of eucalypts and rauli in Britain is higher than the other species considered. However, this increased productivity comes at a much higher and poorly quantified risk. Both Evans (1986) and Purse and Richardson (2001) have been quoted in this report suggesting that the range of productivity of eucalypts could be in the region of 5.4-7.2 odt ha⁻¹ yr⁻¹ and 10-15 odt ha⁻¹ yr⁻¹ respectively. The minimum of either of the ranges does little to communicate the added risks of using eucalypts compared with other species, for which we have a better understanding of likely survival and growth. The new data that have been presented in this report does give some information on this point. Of the four sites that were planted with eucalypts in spring 1980 one suffered establishment problems, two showed good early survival but only satisfactory growth, and the last one showed the potential of the species in Britain.

The relationship between survival/growth and cold for eucalypts is likely to be more complex compared with many of our native and naturalised species that become fully dormant during the winter. This is clearly shown from the survival data from Thetford and Glenbranter where deaths have continued after the establishment phase. This contrasts with the general pattern when planting dormant species where the rate of death is greatest in the first year after establishment and then the rate declines to a much lower level.

An attempt has been made in this review to take account of the types of sites where SRF may be planted. It is interesting to note that during the 1980s Evans (1986) clearly thought that eucalypts would be planted on the type of land that was available for forestry during that time, i.e. marginal agricultural land in the uplands. The emphasis shifted in the 1990s to include the possibility of tree planting on more productive ex-agricultural land (Williamson, 1992). Since 2000 prices for agricultural products have soared and there has been increased emphasis on the use of woodland to regenerate land and create urban green space. It is difficult to predict what land will become available for SRF but it seems fairly safe to assume that the range of site productivities will be wider than is taken account in the existing yield tables. Some evidence has been presented in the report to support this (Figure 1).

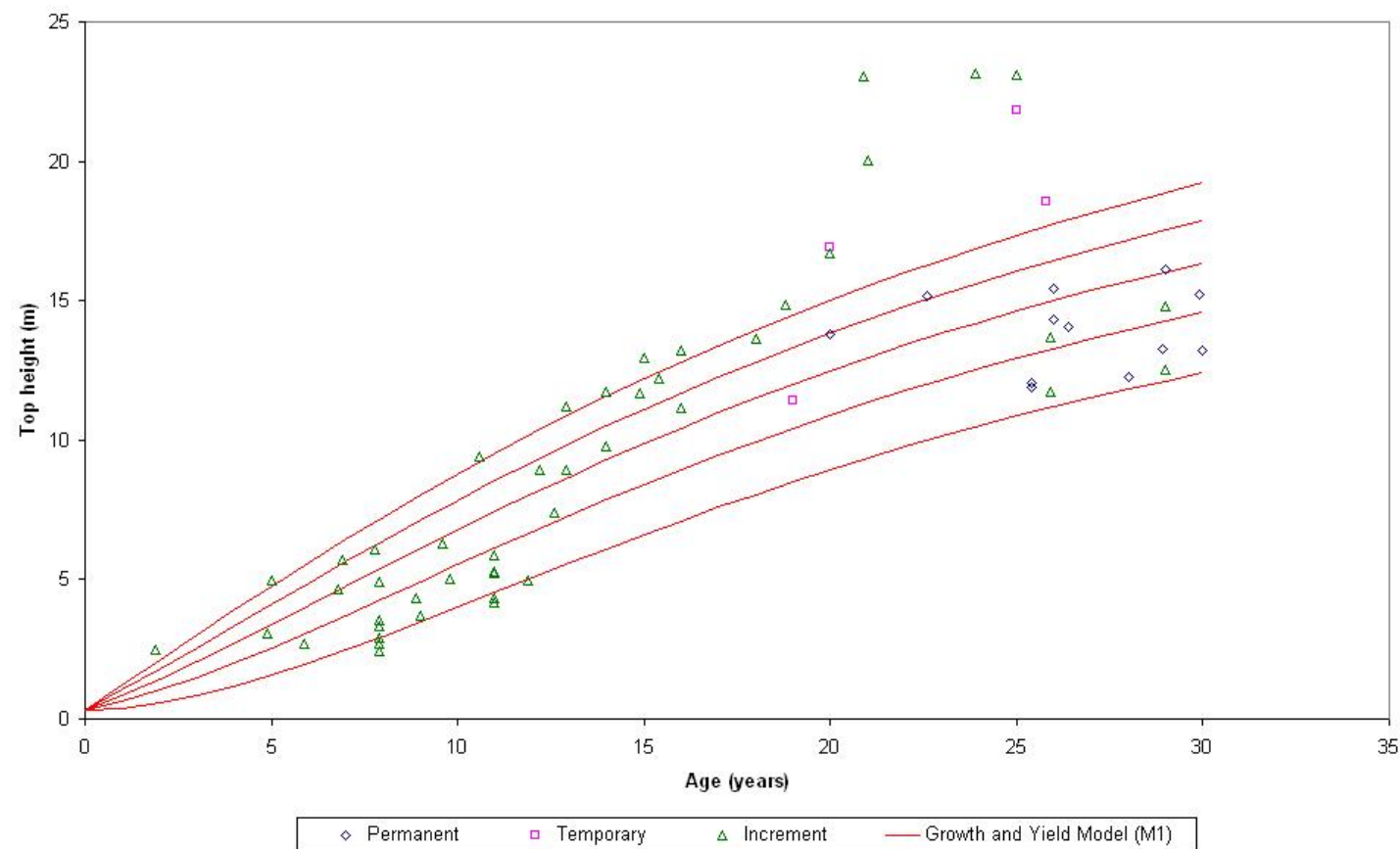


Figure 1 Relationship between top height-age curves for ash and available data from permanent, temporary and increment plots.

The review has also tried to take account of the effects of higher intensity silviculture during establishment, mainly for species with a published yield table. We have good information on the benefits of good early silviculture on survival and growth but most of the data are for a 3-5 year period after planting and longer-term benefits have not been rigorously evaluated. However, there is good evidence to support the fact that longer-term benefits will accrue in terms of growth and yield (Wagner *et al.*, 2006). It is also worth adding that there have only been a few attempts to examine the maximum potential growth of many species that could be obtained by counteracting the main constraints on tree growth on lowland sites. One example of 'maximum amelioration' by the Forestry Commission indicated substantial increases in productivity were possible (Rollinson, 1983).

Much of the information presented in this report uses (1) growth information from small plots and (2) height and diameter assessments to estimate growth. These are both fraught with difficulties when attempting to estimate biomass productivity, as discussed in detail by Cope *et al.* (2008). For example, the plots at Glenbranter each contained 9 trees arranged in a square 3 × 3 plot, so that 8 of the trees were effectively 'edge' trees. In addition the use of 0.35 as a form factor to convert diameter and height to volume is unlikely to be accurate for the range of eucalypts considered in the report, as we know that the growth pattern of the different species varies (Evans, 1986). The use of the basic densities used to convert volume to biomass is also unlikely to take account of the variety of sites and climates where the trees have been growing. However, despite these difficulties the information used in this report is the best available at the time of writing.

In addition to the above points on biomass estimation a further source of possible error is the difference between the desired rotations in Table 1 and those actually reported in this review for species without a published yield table. Data on survival and growth presented in this review are either from the early part of the rotation (<7 years) or, in the case of Thetford and Glenbranter, well beyond the anticipated rotations. This means that (1) it is difficult to assess what the productivity would have been on the rotations in Table 1, and perhaps more importantly, (2) impossible to indicate if the rotations assumed in Table 1 are the optimum for the species considered. The results also show some differences between *E. gunnii* and *E. glaucescens* despite their overall productivity being similar. The main difference is that survival of *E. glaucescens* is generally lower. The fact that *E. glaucescens* maintains a similar level of productivity to *E. gunnii* may indicate that the species may have good compensatory growth, which could be a valuable trait.

The effect of spacing on growth and yield of eucalypts has been quantified for British conditions by the results of New Forest 38. The fact that yield was much higher at closer spacing confirms relationships that have been published by others (Nielsen and Gerrand,

1999). This result emphasizes the fact that spacing is as significant as species when designing an appropriate silvicultural system for SRF. The main objective of SRF should be to capture and convert as much solar radiation into biomass whilst reducing as many limiting factors as possible. To achieve this the best silvicultural system for SRF may involve (i) close initial spacing; (ii) a systematic thinning half-way through the rotation; (iii) coppicing the stems at rotation; (iv) singling the coppice for the second rotation, and (v) mixed species stands to spread risk and increase the efficiency of conversion of solar radiation into biomass. More discussion of possible silvicultural systems for SRF is required; at present the debate is focussed too narrowly on species choice.

Climate change is a major factor that will affect the prospects for SRF in Britain. The UK climate change projections have recently been revised by DEFRA (Murphy *et al.*, 2009; UKCP09). These projections indicate greater temperature changes, less extreme summer drought, and less intense winter rainfall than were predicted in 2002, all of which are likely to affect the productivity of SRF. With regard to winter cold and unseasonal frosts, the predictions (Murphy *et al.*, 2009) indicate that there will be an increase in mean temperature across the UK in all seasons, including a reduction in the number of frost days and in the diurnal temperature range in winter. This is likely to advance budbreak date but reduce frost damage in spring (Cannell and Smith, 1986; Murray *et al.*, 1994). However, in autumn although frosts will occur less frequently, they may cause more damage due to generally warmer temperatures and later hardening of shoots. Therefore in general frost will decrease as a problem (Redfern and Hendry, 2002) and the risks of using non-native species in SRF may also decrease.

Information on the distribution of biomass between shoot and root is scarce and the data from Proe *et al.* (2002) for red alder have been the only authoritative information located in the review process.

5. Conclusions

1. The quantity and quality of growth data available at present are inadequate to underpin the development of SRF in Britain.
2. The relationship between productivity and risk has not been evaluated for the range of non-native species being considered for SRF. For example, the often quoted productivity range of 10-15 odt ha⁻¹ yr⁻¹ for eucalypts does not adequately take account of the risks of frosts and winter cold, i.e. it probably has a lower minimum value.
3. It is clear from the New Forest 38 data that spacing has a significant effect on productivity of eucalypts and is equally important as species in designing a silvicultural system for SRF. More informed discussion of possible silvicultural

- systems for SRF is required including establishment methods, spacing, thinning, coppicing and singling, and species choice (including mixtures).
4. Our knowledge of provenance selection for non-native species being considered for SRF has not developed much since the 1980s and is inadequate.
 5. The *potential* productivity of eucalypts and rauli in Britain is higher than any of the other species considered in this review. However, little work has been carried out to examine the potential for improving the growth of native species in a SRF system.
 6. Based on the findings of this review the range of productivity for each of the species considered in this report when planted in SRF is given in Table 16.

Table 16: SRF biomass productivity range

Common Name	Scientific Name	SRF biomass productivity range (odt ha ⁻¹ yr ⁻¹)	Rotation (years)
Shining gum	<i>Eucalyptus nitens</i>	*	-
Tingiringi gum	<i>E. glaucescens</i>	2.5 - 7.6	<10
Cider gum	<i>E. gunnii</i>	1.5 - 8.2	<10
Italian alder	<i>Alnus cordata</i>	*	-
Red alder	<i>Alnus rubra</i>	0.9 - 4.8	-
Hybrid aspen	<i>Populus tremula x tremuloides</i>	*	-
Sycamore	<i>Acer pseudoplatanus</i>	0.5 - 5.7	20
Silver birch	<i>Betula pendula</i>	0.5 - 5.7	20
Ash	<i>Fraxinus excelsior</i>	0.5 - 4.7	20
Rauli	<i>Nothofagus alpina</i> (syn. <i>N. procera</i>)	3 - 10.5	15
Sweet chestnut	<i>Castanea sativa</i>	1.2 - 6.0	20

* Insufficient information located as basis to estimate a productivity range

6. Acknowledgements

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Appendix 1: Red alder (*Alnus rubra* Bong.) – silviculture and provenance – a north British perspective.

Bill Mason, Senior Silviculturist, Northern Research Station, Forest Research

Natural range

Red alder originates from western North America from SE Alaska to California. It is a characteristic pioneer species of disturbed sites in humid climates ($> 750 \text{ mm y}^{-1}$ rainfall).

Sites

Red alder is usually found on moist sites along streams and lower slopes. It prefers loams, gravels, sands and clays, but it can tolerate soils with restricted drainage provided these are not waterlogged during the growing season. Optimum growth is to be found on soils of moderate acidity (pH of 4.5-5.5) and it responds well to added phosphate.

British experience

1. Experiments go back to at least the 1930s. Some 75 trials/experimental plots have been planted, excluding those where the species was planted in nursing mixtures with Sitka spruce.
2. Nearly 50 per cent of trials were planted on peats or peaty gleys, which in hindsight are not soils for which the species is suited. See site preferences above.
3. Performance has been very variable in terms of survival and long-term growth. The species is very sensitive to autumn and spring frost. Sites with high survival are rare in British trials. Early height growth of surviving trees is often very fast (e.g. height increments of $> 1\text{m}$ 2-3 years after planting). However, dieback and/or crown breakage is often reported by 10-15 years and long-lived trees are quite rare. Glimerveen and McNeill (1993) quote an average height of 16.8 m from a review of FR experimental sites where red alder had survived longest.
4. Individual stands have shown high growth rates on good soils e.g. values for Lennox and Glencorse estimated at $10\text{-}12 \text{ m}^3 \text{ ha}^{-1} \text{ y}^{-1}$ (McIver, 1991), see Table 17. The species will regenerate naturally on suitable sites where it can form dense stands, similar to those reported from the Pacific Northwest. Its ability to coppice is questionable given variable performance in some British trials (Sheppard, 1993).

Provenance

1. The only provenance trial was a series initiated in 1982 and planted in 1985 on 5 sites (Bramshill, Falstone, Solway, Shin, and Glencorse near Bush). These compared some 25 origins from throughout the natural range and included material from the first generation stand at Lennox forest (near Glasgow) which has been widely used in British forestry.
2. Sheppard (1991) has reported on early results from Glencorse including frost testing among provenances. 10-11 year data are available for the three other sites in north Britain. The results show variable survival of about 50% (Shin), 60% (Falstone) and about 80% (Solway). Height of the best provenance was 7.3 m at Falstone, 6.6 m at Solway, and 3.2 m at Shin (see Table 17 for more details).
3. There was considerable between provenance variation in these trials and much interaction with site. However, the home collected Lennox provenance gave consistently good performance in all trials (i.e. ranking in the top 5).
4. Frost hardiness testing (Cannell *et al.*, 1987) suggests that red alder, when fully dormant, is cold hardy to -30°C in Britain. However, it dehardens early in the spring and hardens late in the autumn. The species is therefore vulnerable to spring and autumn frost (colder than -3°C).

Conclusions

1. Red alder is a fast growing pioneer species which can make good growth on suitable sites in Britain, especially on moist fertile soils. It is probably best suited to the more oceanic parts of the country. Peaty soils should be avoided.
2. Autumn and spring frosts are a serious risk to establishment, and also to coppice regrowth.
3. Home collected seed sources (e.g. Lennox) should give good results, within site and climatic limitations.

Table 17 Some summary data for FR experiments with red alder established since 1980

Experiment	Year planted	Soil	SNR	Age	Mean height (m)	Best height (m)	Mean dbh (cm)	Best dbh (cm)	Estimated Basal Area (m ² ha ⁻¹) ^b
Shin 112 ^a	1985	Deep peat	Very poor	11	2.5	3.2			
Solway 12 ^a	1985	Peaty podsol	Poor	10	4.7	6.6			
Falstone 14 ^{a,e}	1985	Surface water gley	Moderate/poor	11	6.2	7.3	7.2	9.4	15.3
North York Moors 79 ^c	1993	Brown earth/ironpan intergrade	Moderate (ex cultivation)	10	10.0	10.4	12.4	12.7	25-26
York 13 ^d	1998	Reclamation – mining spoil	Very poor/poor	10	6.8				

Notes:

- These experiments are part of the 1985 provenance trials. The mean value is the average of all provenances and the best is the provenance that performed best at that site.
- Basal areas are estimated by multiplying the mean dbh by the average number of trees per plot.
- This is a 1:1 mixture experiment where red alder is mixed with either Common alder or oak. Basal areas are calculated as for b for red alder and then doubled to allow for full red alder stocking i.e. without the other component. This is one of the few recent experiments where red alder shows excellent survival. Assuming 2000 trees per hectare and a basic density of 0.45 and using the Purse and Richardson (2001) volume formula gives **4.84** odt ha⁻¹ yr⁻¹.
- This is another mixtures experiment with three levels of site amelioration, but it contains pure plots of red alder. The alder performance showed little response to added sludge or greencrop. Survival here was less than 60 per cent.
- These growth data were used to calculate the minimum of the productivity range for red alder. Assuming 2000 trees per hectare and a basic density of 0.45 and using the Purse and Richardson (2001) volume formula gives **0.9** odt ha⁻¹ yr⁻¹.



TRICHIOCAMPUS VIRINALIS. Poplar sawfly on Aspen

Potential Impacts of Pests and Pathogens on Short Rotation Forestry in Britain

Joan Webber, Christine Tilbury, Heather Steele and David Rose

Summary

Increasingly, short rotation forestry (SRF) is being considered as another option for efficient biomass production and several possible species/hybrids have been suggested as suitable candidates for SRF in Britain. These comprise various alder species (*Alnus glutinosa*, *A. cordata* and *A. rubra*), ash, birch, poplar and sycamore, as well as some cold-hardy eucalypts (*Eucalyptus gunnii*, *E. nitens* and *E. gunnii* X *dalrympleana*) and *Nothofagus* species *N. obliqua* and *N. nervosa* (= *procera*). However, there is only scattered information on the range of pests and pathogens that could affect the productivity of these potential SRF species. To gauge the agents most likely to cause significant damage, records in the extensive databases compiled over decades by the Disease Diagnostic Advisory Service (DDAS) and the Pest Advisory Service (PAS) of Forest Research (FR) were analysed. These databases catalogue all diagnostic queries or reports of tree disorders and pests that come into FR each year.

The analysis indicated several pests and pathogens common to most of the SRF species, but generally these only cause limited or sporadic damage under current conditions despite being widespread. A number, however, are able to cause serious levels of damage and could become established in SRF plantations. They include root pathogens such as *Phytophthora*, which can cause very significant damage and mortality to hosts such as alder, eucalyptus and *Nothofagus*, and some foliar pathogens such as bacterial canker (*Xanthomonas populi*) and Melampora rust, both of the latter are already notable for their impact on Short Rotation Coppice poplar plantations. A number of pests and pathogens either not currently present in Britain or recently arrived could also compromise SRF plantings should they arrive and establish. They include emerald ash borer and a gall forming wasp (*Ophelimus*) which attacks sycamore.

Apart from identifying potential pest and pathogen problems for SRF crops, the interaction between climate change and some of the more serious tree pathogens is also considered. In particular, the likelihood that trees may show increased susceptibility to certain pests and diseases when under drought stress is highlighted. The need for careful selection of provenances to avoid various health disorders, as well as an

understanding of how new pests can be introduced via the plant trade are both emphasised as a first step to combating health problems on SRF plantings.

1. Introduction

The oil crisis of the 1970s was the initial impetus behind the move to grow biomass for fuel. In Britain, this interest focussed on willow and poplar as potential sources of fast growing woody biomass, raised in the form of short rotation coppice (SRC) with a rotation time of 2-5 years. These crops tended to be grown on farmland and therefore were considered as the province of agriculture rather than of forestry. However recently, short rotation forestry (SRF) has been put forward as another possible method for efficient biomass production. One of the advantages of SRF is its similarity to the historic fuelwood coppice systems that have operated in Britain for centuries, and the underlying principle is to grow trees at spacings that encourage rapid growth. After a growth period of between 8 and 20 years, the trees are felled when they reach a size that is optimum for harvesting and easy handling, usually between 10-20 cm diameter at breast height.

A recent review by Hardcastle (2006) commissioned by the Forestry Commission and Defra suggested twelve possible species/hybrids as suitable candidates for SRF in Britain. The selection of species (see Table 1), was based on site demands, stand structure and biodiversity potential, as well as possible impacts on hydrology, soils and landscape. The purpose behind the selection was to try and include those species with potential to be used on a substantial scale and to encompass native, naturalised and exotic species.

Table 1. Proposed Species for SRF in Britain

Native or naturalised in Europe	Exotic
Alder (black)	Alder (red)
Alder (Italian)	<i>Eucalyptus gunnii</i>
Ash	<i>Eucalyptus gunnii</i> X <i>dalrympleana</i>
Birch	<i>Eucalyptus nitens</i>
Poplar (cultivars)	<i>Nothofagus obliqua</i>
Sycamore	<i>Nothofagus nervosa</i> (= <i>procera</i>)

The exotics considered for SRF tend to be relatively well known. For example, *Nothofagus* is already widely planted, including some forest plots (Tuley, 1980), and its form is similar to beech. *Eucalyptus gunnii* has been extensively planted in parks and gardens, and its light canopy makes it less intrusive than the other two *Eucalyptus* under consideration.

2. Susceptibility to pests and pathogens

Concerns have been expressed over possible pest and disease issues associated with SRF. Firstly, the possibility of importing pests through seeds and propagation material. Secondly, given the likelihood that SRF species will be on ex-agricultural land, the possibility that they could harbour agricultural pests and diseases including the risk of providing alternate hosts. A third possibility is the potential for pests and diseases from other land uses and habitats to impact on SRF plantations.

In the earlier review (Hardcastle, 2006), it was concluded that the only imports of SRF material from outside the EU would be in the form of seed for *Eucalyptus* and possibly also *Nothofagus*, which should minimise plant health risks. A wider literature search did not reveal any evidence of cross-infection between agriculture and SRF plantations, nor did it reveal any dangers from pathogens able to infect SRF species which might find alternate hosts in agricultural crops, and as a consequence be already established on potential SRF sites.

Despite this, there are potential threats to SRF plantations from the pests and pathogens already present in Britain. To gauge how damaging these might be we turned to the extensive databases compiled over decades by the Disease Diagnostic Advisory Service (DDAS) and the Pest Advisory Service (PAS) of Forest Research (FR). These databases catalogue all diagnostic queries or reports about tree disorders and pests that come into FR each year. Records from the databases that related to the species listed in Table 1 were compiled, categorised and evaluated for their potential to impact on the SRF species under consideration. The DDAS database comprises reports from both FR research stations (Alice Holt, near Farnham, Surrey; Northern Research Station, Penicuik, Midlothian). The time period chosen for the data analysis study spanned 1976-2008 for the Alice Holt DDAS records, 1998-2008 for the Northern Research Station DDAS records, whilst PAS records spanned 1985-2008. In part, this reflected the depth of each database and the resources available to analyse the records.

a) Ash - *Fraxinus excelsior*

Fraxinus excelsior (ash) is common throughout Britain although it diminishes in frequency in the extreme north of Scotland. It is recorded as the third most common broadleaved species in the most recent Census of Woodlands and Trees, and is now one of the most widely planted broadleaved trees.

Diseases of ash

Combined DDAS records between 1976-2008 for ash totalled 281; about 30% of the reports were disorders caused by abiotic agents (site conditions/cultural, herbicide and

salt damage, or weather related impacts). Records for the main disorders which occurred at least four or five times over the time period analysed are shown in Table 2.

Most of the pathogen records for ash were decay fungi, with root rotting *Armillaria* species occurring frequently. Other decay fungi considered common on ash, such as *Perenniporia fraxinea* and *Inonotus hispidus*, were also recorded regularly (Table 2). However, very few foliar pathogens were noted although certain bark attacking canker pathogens were recorded consistently over three decades. These included the bacterial canker *Pseudomonas savastanoi*, which is known to be widespread although at a relatively low incidence (Strouts and Winter, 2000). Different clones of ash apparently vary greatly in their resistance to this pathogen, and susceptible individuals can suffer from many separate infections and so become very disfigured and distorted although they are rarely killed by the disease. The susceptibility of certain genotypes and seedlots may therefore be an important factor to keep in mind if ash provenances are selected for SRF with a limited genetic base.

Table 2. Common disorders of *Fraxinus excelsior* from DDAS records: 1976-2008

Disorder/Pathogen	Type of damage/symptoms	Number of reports
Biotic		
Decay fungi (various or unidentified)	Wood and root rotting	38
<i>Armillaria</i> spp. (honey fungus)	Root rotting and basal cankers	23
<i>Ganoderma</i> spp	Root and butt rot	8
<i>Perenniporia fraxinea</i>	Root and butt rot	12
<i>Pholiota squarrosa</i>	Root and butt rot	7
<i>Daldinia concentrica</i>	Top rot	4
<i>Inonotus hispidus</i>	Top rot (ash heart rot)	19
<i>Nectria</i> / <i>Cylindrocladium</i> spp.	Bark – canker	14
<i>Phomopsis</i> spp.	Bark – canker	7
<i>Pseudomonas savastanoi</i>	Bark – bacterial canker	5
Abiotic		
Weather related	Mainly drought/frost damage	17
Waterlogging	Root death	1
Chemical damage	Herbicides and salt	21
Site conditions/cultural	Multiple symptoms	20

Other relatively common bark pathogens included species of *Nectria* and *Phomopsis* (Table 2). These are usually considered to be weak pathogens capable of causing only scattered bark and twig death, although *Nectria* may occasionally cause larger perennial cankers on stems and branches of ash. However, the diseases caused by these pathogens are generally of little consequence although they are widespread and common.

Pests of ash

The PAS catalogued a total of 71 records of pests attacking ash between 1985-2008. The most common agents are listed in Table 3 and included various leaf galls and leaf or bud feeding insects. Most native species of broadleaved trees are affected by leaf galls which are mainly induced by eriophyd mites, gall midges or aphids. Although these galls are often highly conspicuous, usually they little impact on the overall health of the tree and just affect the appearance of the foliage. Likewise, the common leaf weevil (*Phyllobius pyri*) which feeds on the foliage of various broadleaved trees and fruit trees, seldom causes severe damage.

Table 3. Common pests of *Fraxinus excelsior* from PAS records: 1985-2008

Pest	Type of damage	Number of reports
<i>Eriophyes fraxinivorous</i> (Eriophyd mite)	Leaf bead gall	4
<i>Dasineura</i> spp (mainly <i>D. fraxini</i> , gall midge)	Leaf pouch gall	5
<i>Psyllopsis</i> spp. (plantlouse)	Leaf gall	6
<i>Phyllobius pyri</i> (leaf weevil)	Leaf feeding	2
<i>Prays fraxinella</i> (ash bud moth)	Bud and shoot death	7
<i>Operophtera brumata</i> (winter moth)	Bud and leaf feeding	2
<i>Hylesinus</i> spp. (mainly <i>H. varius</i> , ash bark beetle)	Beetle galleries in bark	25

Ash bud moth (*Prays fraxinella*) is also widespread and probably very common although not often reported. The damage caused by this pest is most visible in spring as it girdles shoots and causes them to wilt as well as resulting dieback of some small twigs. Symptoms are frequently visible on young ash and it has been suggested that the consequent loss of terminal buds and shoots causes forking and poor form. However, Kerr and Boswell (2001) found that ash bud moth was present in less than 1% of the 4106 buds examined and so concluded that this was unlikely to be a serious cause of forking in newly planted ash trees, at least in southern Britain.

Another insect pest commonly associated with ash is the bark beetle *Hylesinus varius*. This can cause knobbly outgrowths on branches which are known as 'ash roses'. These are produced when the ash bark beetle tunnels into the bark and wood of trees to overwinter. However, *Hylesinus* beetles also breed in the bark of ash trees previously weakened by other factors such as root disease, so although they are frequently reported as a pest, they are not the cause of major damage or tree death (Alford, 2007).

Ash also suffers from the condition known as 'ash dieback' although only one report of this listed in the DDAS database. Ash dieback is a complex or syndrome, apparently not the result of a single causal agent, but from the combined effects of several pests and pathogens. Affected trees show varying levels of decline, ranging from death of twigs through to dieback of branches and major limbs. It is most common in hedgerow trees, particularly those adjacent to arable land and its incidence is highest in eastern England where trees are more likely to be subject to drought stress (Hull and Gibbs, 1991). It has been hypothesized that root damage, for example caused by agricultural disturbance, can alter the toughness of leaves thereby making ash trees in hedgerow ecosystems more susceptible to insect herbivores such as weevils (Foggo *et al.*, 2008) and therefore more liable to the syndrome of dieback.

New threats to ash

Mortality of ash has been increasingly observed in European countries during the last ten years. Initially, studies revealed the presence of a pathogenic fungus associated with the dying trees, and this was identified as a new species of fungal pathogen named *Chalara fraxinea*. Further studies carried out in 2008 in Poland then revealed that *C. fraxinea* was another form (the anamorph) of an already described species, *Hymenoscyphus albidus*, which is considered as non-pathogenic, native, and widespread in Europe (including Britain). Therefore, the emergence of a new disease caused by this species is difficult to explain, and it is acknowledged that further studies are needed to understand why it is now causing this damaging disease on ash in some parts of Europe. So far, however, no significant changes in the health of ash have been observed in Britain that could be attributed to *H. albidus*/*C. fraxinea*.

Another threat to ash which is absent from Europe is *Agrilus planipennis*, a species of Asian buprestid beetle which has recently been introduced into North America with disastrous results as it is now causing significant damage to ash trees in both urban and forest environments. Symptoms include general yellowing and thinning of foliage, dying of branches, crown dieback, and eventually death of the tree after 2 to 3 years of infestation. In the most affected parts of the USA it is estimated that *A. planipennis* has killed millions of trees over the past few years (*F. pennsylvanica*, *F. americana* and *F. nigra*, as well as several horticultural varieties of ash). This pest is absent from Europe and phytosanitary measures are in place to prevent its accidental introduction, but it

remains a concern especially as the level of susceptibility of European ash species such as *F. excelsior* is unknown.

b) Alder – *Alnus*

Black alder – *Alnus glutinosa*

Alluvial forests on flood plains are dominated by alder *Alnus glutinosa* and willow *Salix* spp but intensive agriculture and clearance of riverine woodland have eliminated most true alluvial forests in the UK and often just narrow strips or lines of trees remain (Harper *et al.*, 1997). Therefore, although this is a native species the total UK extent probably only amounts to around 6,500 ha.

Italian alder – *Alnus cordata*

In its native range Italian alder has a very limited distribution, and is present only in southern Italy and Corsica. The species was introduced to Britain in 1820 and has since been planted widely in shelter-belts and woodland grant schemes. It thrives best on sites with high rainfall and relatively mild winters (Claessens, 2003).

Red alder – *Alnus rubra*

Red alder (*Alnus rubra*) is a native of North America but has been used as an ornamental in some European countries such as Britain and the Netherlands (Orwa *et al.*, 2009). It has also been deployed in Britain for planting on sites being rehabilitated after land-fill or mining contamination.

Pests and diseases of alder

Cech and Hendry (2003) reviewed various causes of dieback and decline in European alders. They concluded that although a number of weak pathogens were implicated in the dieback of branches and stems, they were unlikely to be very damaging unless alders are weakened by other factors such as fluctuating water tables or climatic factors. Armillaria root rot was considered to occur quite commonly on alder, but undoubtedly the most serious pathogen is the alder Phytophthora, first found in England in 1993 (Brasier *et al.*, 1995).

Phytophthora alni is now widespread in Britain, and is the cause of disease of dieback and death of around 20% of native black alders in riparian ecosystems. It can also infect Italian alder, although this species is less susceptible than black alder (Webber *et al.*, 2004). Red alder is apparently much more resistant to this pathogen (T. Jung, personal communication). Evidence from different countries indicates that this pathogen has been spread widely via the movement of infected planting stock from nurseries, although symptoms may not have been evident at the time (Jung and Blaschke, 2004; Santini *et al.*, 2001).

The combined DDAS records for alder mostly related to black (common) alder with around 100 records for *A. glutinosa*, 20 records for *A. cordata* and only 16 for *A. rubra*. The records confirmed the importance of *Phytophthora* as a pathogen of alder (Table 4); even the less susceptible Italian and red alder were both affected. The only other pathogens reported with any frequency include root rot fungi and some fungal foliar pathogens such as *Melampsorium*, *Taphrina* and the bark/shoot fungus *Melanconium*.

Table 4. Common disorders of *Alnus* spp. from DDAS records: 1976-2008

Disorder/Pathogen	Type of damage/symptoms	Number of reports		
		Black	Italian	Red
Biotic				
Decay fungi, including <i>Armillaria</i> and <i>Chondrostereum purpureum</i>	Wood and root rotting	7	0	1
<i>Phytophthora</i>	Collar rot and stem lesions	56	5	4
<i>Taphrina tosquinetti</i>	Leaf death	4	0	0
<i>Melampsorium betulinum</i>	Leaf rust	2	0	0
<i>Melanconium</i>		2	1	0
Abiotic				
Weather related	Drought damage	14	1	4
Chemical damage	Herbicides	4	0	0
Site conditions/cultural/other	Multiple symptoms	8	11	8

Records of pests affecting alder in the PAS database were sparse, although like ash and birch, alder was found to be frequently affected by leaf galls and leaf beetles such as *Chrysomela acena*, all of which are wide-spread and common but cause no serious long-term damage (Strouts and Winter, 2000). Other occasional pests also included willow scale (*Chionaspis salicis*), sawflies (eg *Croesus petrionalis*) and buff-tip moth (*Phalera bucephala*).

c) Birch - *Betula pendula*

There has been increased planting of silver birch (*Betula pendula*) in Britain since the 1980s, as a broadleaved element of native woodland restoration schemes or as amenity trees along roadsides.

Diseases of silver birch

Over 200 combined DDAS records listed disorders affecting birch, with a significant proportion due to wood and root rotting fungi (Table 6). *Betula* is rated as very susceptible to *Armillaria* and the number of records that reported honey fungus causing damage reflected this. Birch leaf spot (*Gleosporium betulinum*) was another commonly reported disease. Although the latter can cause severe defoliation on occasion, it is not thought to cause any lasting damage to affected trees except in combination with other stress factors (Phillips and Burdekin, 1982; Strouts and Winter, 2000). The only other pathogens reported more than once included leaf rusts and *Phytophthora*, although birch is much less susceptible to *Phytophthora* compared with some of the other species recommended for SRF such as alder and sycamore. In contrast, *Melampsorium betulinum* is considered one of the most important foliar diseases of birch in several European countries, and is associated with retarded height growth and increased mortality. Helander *et al.* (1998) found that individuals of *B. pubescens* differed in resistance to this rust pathogen, but infection levels of susceptible compared to resistant trees was also dependent on environmental conditions.

Table 6. Common disorders of *Betula pendula* from DDAS records: 1976-2008

Disorder/Pathogen	Type of damage/symptoms	Number of reports
Biotic		
Decay fungi (various or unidentified)	Wood and root rotting	38
<i>Armillaria</i> spp. (honey fungus)	Root rotting and basal cankers	28
<i>Piptoporus betulina</i>	Top rot	3
<i>Phytophthora</i> spp	Root and collar rot	2
<i>Anisogramma virgultorum</i>	Leaf spot and shoot lesions	2
<i>Gleosporium (Discula) betulinum</i>	Leaf spot	19
<i>Melampsorium betulinum</i>	Leaf rust	4
<i>Marssonina betulae</i>	Leaf spot and shoot lesions	12
Abiotic		
Weather related	Mainly drought, lightning	13
Waterlogging	Root death	2
Chemical damage	Herbicides	8
Site conditions/cultural/other	Multiple symptoms	21

Green (2005) reported dieback as widespread in birch included in new native woodland grant scheme (NWGS) plantings in Scotland. Between a quarter and two thirds of the birch in NWGS plantings that were surveyed had 40% or more crown dieback. Affected

trees showed a steady deterioration in crown condition 5-10 years after planting. Three fungi were particularly associated with the dieback – *Discula betulina*, *Marssonina betulae* and *Anisogramma virgultorum*, with the latter two having the most impact. Green suggested that *M. betulae* is much more damaging to birch than the current literature suggests, demonstrated by the way it caused secondary stem cankers which continued to expand months after the inoculation of young birch plants. Surveys of NWGS in Scotland also revealed that *M. betulae* was usually associated with *B. pendula*, whereas *A. virgultorum* occurred more frequently on *B. pubescens*. Green (2005) concluded that birch provenance could be an important factor in determining susceptibility to these diseases, although climatic variables also played a part.

Pests of birch

Only a small number of pest reports (13) were recorded by the PAS on silver birch (*B. pendula*), although more reports of pests (43) were attributed to the genus *Betula* spp (see Table 7). Most of the common pests are widely distributed and feed on other deciduous tree genera, and usually only cause limited or localised damage. Both the buff-tip and vapourer moth caterpillars are reported as voracious leaf feeders but seldom defoliate entire plants; both were occasionally found feeding on birch (Table 7).

Table 7. Common pests of *Betula* species from PAS records: 1985-2008

Pest	Type of damage	Number of reports	
		<i>B. pendula</i>	<i>Betula</i> spp.
Aphids (<i>Betulaphis</i> , <i>Glyphina</i>)	Leaf feeding	2	3
Eriophyid mites	Leaf gall	1	2
<i>Croesus septentrionalis</i> (sawfly)	Leaf feeder	3	8
<i>Elasmucha grisea</i> (shield bug)	Leaf feeder	2	2
<i>Phyllobius pyri</i> (leaf weevil)	Leaf feeding	1	15
<i>Phalera bucephala</i> (buff-tip moth)	Leaf feeding	2	2
<i>Orgyria antiqua</i> (vapourer moth)	Leaf feeding	0	3

d) Poplar – *Populus*

In Britain there are two native members of the poplar family - the aspen *P. tremula* and the black poplar *P. nigra* var *betulifolia*; grey poplar (*P. x canescens*) may be native in southern Britain too. Poplars are considered to be at the northern limit of their natural distribution and less likely to thrive when grown outside southern Britain although they are planted widely for screening, shelter or ornament. Planted forms include numerous cultivated hybrids, varieties and clones of poplar, many of which have been produced

from plant breeding programmes to offer rapid growth as well resistance to various diseases: particularly rust fungi of the genus *Melampsora* and bacterial canker *Xanthomonas populi*.

There has been a growing interest in planting poplars throughout the 1990s. Between 1989 and 1998 many poplar plantations were established, mainly on land previously used for arable purposes and pasture. In addition, recent estimates of land suitability suggest that as much as 3 million ha of agricultural land in Britain is suitable for planting with poplar (Williams and Thomas, 2006) and this would include SRF. However, currently about 14,000 ha of poplar are under cultivation with approximately half in East Anglia (Thomas *et al.*, 1998). The most common clones planted are the varieties Beaupré and Boelare, although loss of resistance to disease organisms such as rust (*Melampsora larici-populina*) appears to be one of the main current threats to successful poplar exploitation (Lonsdale and Tabbush, 2002).

Diseases of poplar

Over 300 records of disorders affecting the many different clones and varieties of poplar have been collected by the DDAS between 1976-2008, but these include records for species as diverse as *P. alba*, *P. balsamifera*, *P. nigra* and *P. trichocarpa*, as well for various clones of *P. trichocarpa* X *P. deltoides* and *P. deltoides* x *P. nigra* varieties. The different clones and species can vary greatly in their susceptibility to the many diseases that affect poplar. However, overall the records do emphasise the frequency with which *M. larici-populinum* and bacterial canker (*Xanthomonas populi*) affect poplar (Table 8), although decay and root rot fungi (including *Armillaria*) are common, as well as the leaf spot fungi *Marssonina populi* and *Taphrina populina*. *Melampsora* rust can be particularly damaging to some poplar species and clones, affecting trees in different ways. Leaves on affected trees shrivel and fall prematurely but the infection can also interfere with the frost tolerance characteristics of some host species. A combination of these effects can lead to seriously reduced increment, shoot dieback or even tree death with severe infection, depending on the timing and severity of the infection (Tubby, 2005).

The vulnerability of poplar to certain diseases, when grown for example in SRC plantations, comes from the clonal nature of many of the varieties and the ease with which new pathogen genotypes evolve to overcome resistance in selected clones. Thus, in 1994 a new pathotype of *M. larici-populina* (pathotype 'E4') arrived in Britain and infected all commercial varieties that were previously totally rust tolerant (Tubby, 2005). Currently, with no approved pesticide for use against poplar rust, a site location at least 500 m away from the alternative host (*Larix* spp.) of the rust fungus is recommended, to prevent rust populations from establishing and cycling from host to host. Other recommendations to combat poplar pathogens include planting mixes of genetic material of disease-approved varieties.

Table 8. Common disorders of poplar from DDAS records: 1976-2008

Disorder/Pathogen	Type of damage/symptoms	Number of reports	
		<i>Populus</i> spp.	<i>Populus</i> clones*
Biotic			
Decay fungi (various or unidentified)	Wood and root rotting	15	1
<i>Armillaria</i> spp. (honey fungus)	Root rotting and basal cankers	4	0
<i>Phytophthora</i> spp	Root and collar rot	2	1
<i>Chondrostereum purpureum</i>	Decay	4	2
<i>Ganoderma</i> spp	Top rot	5	0
<i>Melampsora larici-populinum</i>	Leaf rust	13	9
<i>Marssonina populi</i>	Leaf spot	8	0
<i>Taphrina populina</i>	Leaf blister	7	0
<i>Xanthomonas populi</i>	Bacterial canker	11	1
Abiotic			
Weather related	Drought, frost, wind	7	3
Chemical damage	Herbicides	9	1
Site conditions/cultural/other	Multiple symptoms	10	1

* clones of 'interamerican' hybrids (*P. trichocarpa* X *P. deltoides*)

In contrast to the many records of diseases of poplar, relatively few poplar pests were reported to the PAS. These included the smaller poplar leaf beetle (*Phyllodecta laticollis*) and the brassy willow beetle (*P. vitellinae*), both of which are considered to be common pests of poplar and willow and able to cause significant damage to leaves as they feed. Although cases of heavy infestation are rare, they can lead to reduced yield and crop death. However, because defoliation is usually periodic and late in season it usually does not cause a reduction in crop yield. Sawflies (*Nematus melanaspis* and *Trichiosoma lucorum*) can also cause significant levels of defoliation but again the damage is likely to be transient rather than year on year. More details about pests of poplar grown for biomass are available on the FRh SRC webpages <http://www.forestresearch.gov.uk/srcsite/infd-5ldkv9>

e) Sycamore - *Acer pseudoplatanus*

The British and Irish Hardwoods Improvement Programme (BIHIP) reports that there are approximately 67,000 ha of woodland in Britain in which sycamore (*Acer pseudoplatanus*) is the dominant species (<http://www.bhip.org/index.html>). This comprises 49,000 ha in England, 11,000 ha in Scotland and 7,000 ha in Wales. Sycamore is also widespread as a street and amenity tree.

Diseases of sycamore

Pathogens problems affecting this species are reported with relatively high frequency to the DDAS. More than 300 disorders affecting sycamore were recorded between 1976-2008 although many related to abiotic disorders such as mechanical and herbicide damage, pollution, and weather related impacts such as drought, frost or even lightning strikes (Table 9). The reports of damaging biotic agents included more than 30 genera, but certain root rot and decay fungi were consistently associated with sycamore, particularly *Armillaria* and *Kretzschmaria*. A number of foliar pathogens also frequently infected sycamore. These included: *Cristularia depraedans*, which causes striking white spots on leaves and leads to early leaf fall; *Rhytisma acerinum*, the cause of large, black leaf spots; and *Ophiognomonia pseudoplatanus* (giant leaf blotch of sycamore). However, all are considered to be conspicuous but unimportant diseases that cause little if any lasting damage to affected trees (Strouts and Winter, 2000).

Table 9. Common disorders of *Acer pseudoplatanus* from DDAS records: 1976-2008

Disorder/Pathogen	Type of damage/symptoms	Number of reports
Biotic		
Decay fungi (various or unidentified)	Wood & root rotting	28
<i>Armillaria</i> spp.	Root rotting and basal cankers	15
<i>Kretzschmaria (Ustulina) deusta</i>	Root and butt rot	7
<i>Ganoderma</i> spp	Root and butt rot	3
<i>Bjerkandera adusta</i>	Top rot	2
<i>Cristulariella depraedans</i>	Foliar – white leaf spot	17
<i>Ophiognomonia pseudoplatani</i>	Foliar – giant leaf blotch	7
<i>Rhytisma acerinum</i>	Foliar – tar spot	11
<i>Unicula aceris</i>	Foliar – powdery mildew	4
<i>Cryptostroma corticale</i>	Wood – canker stain/wilt	16
<i>Phytophthora</i> spp.	Root rot	7
<i>Verticillium</i> .spp.	Wood – vascular wilt	4
<i>Xylemella fastidiosum</i>		4
Abiotic		
Weather related	Mainly drought damage	22
Waterlogging	Root death	4
Chemical damage	Herbicide and salt	25
Site conditions/cultural	Multiple symptoms	38

Potentially much more damaging agents include the fungal pathogens *Cryptostroma corticale*, *Phytophthora* and *Verticillium* (Table 9). *C. corticale* is a latent pathogen able to exist in the wood of healthy sycamore for many years without harmful effect. However, if prolonged dry weather prevails, the pathogen can spread quickly throughout the wood causing a disease known as sooty bark, which leads to partial or total wilting of the crown and mortality (Young, 1978). Strouts and Winter (2000) comment that the number of tree deaths caused by *C. corticale* can be alarming on a local scale, but mostly disease epidemics are short-lived and geographically restricted because of the climatic conditions required for the disease to be expressed.

Sycamore is also highly susceptible to *Verticillium* wilt and the disease is often fatal, especially to recently planted trees. *Verticillium* can be common in ornamental tree nurseries and is usually introduced onto new sites via infected planting stock. Once present, *Verticillium* is difficult to eradicate as it can persist in soil and roots for years and reinfect new plantings. Similarly, *Phytophthora* pathogens are also common in nurseries, soil borne, and difficult to eradicate once established on a site. Although sycamore has not previously been considered to be particularly susceptible to *Phytophthora*, reports of *Phytophthora citricola* causing aerial bleeding lesions have become more common over the last five years, at least in southern England (Brasier and Jung, 2006). In addition, *P. cambivora* has recently been found causing collar rot in young plantations of sycamore in northern Germany (Hartmann *et al.*, 2005).

Pests of sycamore

Between 1985 - 2008 the PAS accumulated just over 30 records of pest species associated with sycamore, although it is recognised that this tree species supports huge numbers of aphids which provide an important food supply for organisms as varied as birds and fish. The most common pest species are listed in Table 10.

Table 10. Common pests of *Acer pseudoplatanus*: from PAS records: 1985-2008

Pest	Type of damage	Number of reports
Eriophyid mite (eg <i>Artacris macrorhyncus</i>)	Leaf gall	8
<i>Drepanosiphum platanoides</i> (sycamore aphid)	Sap sucking	2
<i>Phyllobius</i> spp (leaf weevil)	Leaf feeding	3
<i>Pulvinaria regalis</i> (horse chestnut scale)	Leaf and twig feeding	16
<i>Euproctis chrysorrhoea</i> (brown tail moth)	Leaf feeding	2
<i>Orthosia stabilis</i> (common quaker moth)	Leaf feeding	2
<i>Pammene regiana</i> (moth)	Seed feeding	2

Pulvinaria regalis (horse chestnut scale) is one of the most conspicuous pests of sycamore, and therefore recorded frequently. It is visible as a white spotted appearance on branches during the summer and these white spots are the dead bodies of the female scale insect surrounded by a waxy wool. High *P. regalis* populations can have a significant impact on the growth of affected trees but this pest is generally considered to be a problem only on urban trees, especially trees under stress due to lack of water or nutrients.

Other pests recorded occasionally on sycamore included various leaf gall-causing mites and various moths – all with larval stages that feed on leaves or even seed. However, although most are fairly common and in some years can become locally abundant, none are likely to have a major impact on the health of sycamore. The brown tail moth is of some note as the caterpillars are well-known for their urticating hairs; they cause extreme irritation if in contact with human skin. They feed in a communal web on the leaves of many broadleaf species, including sycamore but also species such as beech, ash and poplar.

f) Eucalyptus species

The two Eucalyptus species which are candidates for SRF are both mountain/sub-alpine species in their native areas of Australia, and so are fairly hardy, able to withstand moderate frost and winter cold. *E. nitens* operates at a temperature range between -3°C to 29 °C growing best between 9-18°C; *E. gunnii* grows under similar conditions (Australian National Botanic Gardens, 2007; Evans, 1986).

Diseases of Eucalyptus

In countries where eucalypts are either native species or introduced and grown commercially (eg Australia, New Zealand, South Africa and South America) there has been increasing recognition over the past fifty years that they are vulnerable to a wide range of diseases. This is illustrated by some of the destructive epidemics suffered by some eucalypt species, particularly caused by root attacking *Phytophthora cinnamomi* in native forests, but also other epidemics of foliar and canker diseases are frequent in plantations, or cause dieback of remnant trees on agricultural and grazing land. This has stimulated intensive research and management of diseases of eucalypts summarised in Keane *et al.* (2000). Various *Mycosphaerella* pathogens which attack the foliage of some Eucalyptus species have proved to be highly damaging to some plantation grown species, including *E. nitens* in Tasmania (Dungey *et al.*, 1997). *Mycosphaerella juvenis* has also proved to be so such a serious disease of *E. nitens* in South Africa only certain provenances can be grown because of the disease (Crous and Wingfield, 1996). However, the eucalypt species under consideration for SRF are generally not amongst those considered to be especially susceptible to *Phytophthora* or to some of the most damaging foliar pathogens, and most of the latter are not present in Britain. Instead, the most common disorders of eucalypts seen in Britain tend to be common decay fungi (see Table 11), along with damage caused by frost and other winter related injuries. The numerous pathogens of eucalypts elsewhere in the world highlights the impact that some introduced pathogens could have on SRF plantations should they ever arrive in Britain. It also underlines the need for effective plant health measures if seeds or cuttings are imported from major Eucalyptus growing regions around the world such as Australia, South America or even elsewhere in the EU.

Table 11. Common disorders of *Eucalyptus* spp from DDAS records: 1976-2008

Disorder/Pathogen	Type of damage/symptoms	Number of reports between 1976-2008	
		<i>Eucalyptus</i> spp.	<i>Eucalyptus gunnii</i>
<i>Armillaria</i> spp.	Root rot/decay	4	2
<i>Chondrostereum purpureum</i>	Decay	3	2
<i>Hypholoma fasciculare</i>	Decay	1	0
Weather related	Winter cold damage	4	0
Water excess oedema	Over-watering damage	4	2
Cultural/miscellaneous	Multiple symptoms	8	4

Pests of Eucalyptus

Few records of eucalypt pests were available from PAS, with the exception of a species of gall forming wasp, new to the UK. This insect, a species of *Ophelimus*, has been found in a few private gardens in the southeast of England on *Eucalyptus* sp. (Tilbury and Jukes, 2006) and the same species, or similar closely related species, was also found at a plant nursery in Yorkshire (MacLeod, 2007). The identity of the organism has not been confirmed but it is very similar to *Ophelimus maskelli*, an Australian species and pest of *Eucalyptus* that has established in Israel and various southern EU Member States, e.g. France, Greece, Italy and Spain. Very high densities of galls can occur on foliage, such that mature trees can lose almost all leaves. *O. maskelli* is most likely to establish more widely in southern Europe where *Eucalyptus* hosts occur widely as forestry and amenity trees. Damage is likely in Britain but probably less than that currently observed in southern European countries (MacLeod, 2007). However, once again this finding emphasises the danger accidentally introduced pests and pathogens could pose to SRF crops, in addition to the impact of pests already present.

g) Nothofagus species

Stands of *Nothofagus* species can be found on sites scattered through Britain, with the earliest experimental plots planted in the 1950s (Danby, 1991). The most widely grown species are *Nothofagus procera* and *N. obliqua*. Despite this, relatively few records for *Nothofagus* spp have been compiled, and those that are available come only from the Alice Holt DDAS (see Table 12); no records of any insect pests are available.

Table 12. Common disorders of *Nothofagus* spp. from DDAS records: 1976-2008

Disorder/Pathogen	Type of damage/symptoms	Number of reports	
		<i>Nothofagus</i> spp.	<i>Nothofagus nervosa</i>
<i>Botrytis cinerea</i>	Damping off	5	0
Decay fungi	Wood & root rotting	2	6
<i>Phytophthora</i> spp.	Root rot, bleeding canker	2	10
<i>Phomopsis</i> spp.	Twig/branch cankers	0	4
Weather related	Winter cold damage	4	11*
Waterlogging	Root death	1	0
Cultural/miscellaneous	Multiple symptoms	5	7

*3 records of drought damage

Not surprisingly, winter cold damage is reported with moderate frequency as a problem on *Nothofagus*, along with wood rotting fungi and common twig and branch canker fungi (Table 12). However, *Phytophthora* pathogens are also frequent, particularly on *N. nervosa* (= *procera*). More recently, a previously unrecognised species of *Phytophthora*

only described in 2003, *P. pseudosyringae* (Jung *et al.*, 2003), has proved to be highly damaging in some plantations of *N. obliqua* in Britain and therefore could have an impact on this species in SRF plantings. Attack by *P. pseudosyringae* can cause extensive bleeding cankers on the trunks of mature and semi-mature trees, leading to girdling and death of trees (J. Webber, unpublished data). In some locations in southern England up to 70% of planted trees have been infected or killed by this pathogen, leading to premature felling and commercial losses (Ben Jones, personal communication).

3. Interactions with climate change

Typically, the two most important factors in the development of plant disease epidemics are temperature and moisture. Inevitably therefore, climate change is likely to have direct and also indirect effects on the activity of tree pests and pathogens affecting SRF.

In general, all fungi have an optimum temperature range for activity, and this largely determines their potential for survival, growth and range, thus defining their geographical distribution in relation to their hosts. However, unlike root and wood infecting fungi, foliar pathogens are more immediately responsive to climate change than most other pathogens. They are directly exposed to fluctuations in air temperatures and UV radiation without the mitigation supplied by surrounding soil or woody tissue. Thus, foliar diseases are probably linked most strongly to changes in temperature and precipitation (Peterson, 1967). This means that foliar pathogens which require free moisture for host infection, sporulation and spore dissemination are likely to become more damaging in western parts of the UK with predicted increased spring rainfall as a result of climate change.

Such pathogens would include those in the genera *Marssonina* and *Melampsora* and the latter are some of the most damaging diseases of commercially grown poplar in the UK, particularly *Melampsora larici-populina*. In addition, the warm-temperature *Melampsora* species such as *M. allii-populina*, which currently has a geographical range in southern and central Europe, are predicted to become more problematic in the south of the UK due to climate change (Lonsdale and Gibbs, 2002). Threats posed by northward shifts of existing rust species, together with evolution of new pathogenic races of these diseases, have potentially serious implications for poplar grown in short rotation.

In addition, it has long been recognised that interactions exist between drought stress and fungal diseases of forest trees, and drought-stressed trees tend to be more susceptible to pathogen attacks (Bier, 1959; Hepting, 1963; Schoeneweiss, 1975). With predictions of increased frequency and severity of drought in parts of the UK due to climate change, trees growing on drought-prone sites or shallow rooted species such as

birch are likely to suffer increased attacks by root infecting fungi. Drought stress in trees is generally considered to act as a predisposing factor to infection by *Armillaria*, especially the more weakly pathogenic species which can be opportunist and only able to attack trees that have been weakened through other biotic and abiotic agents (Gregory and Redfern, 1998; Desprez-Loustau *et al.*, 2006). In addition, root infecting Phytophthoras, require moist soil conditions (even periods of flooding) for infection and spread, but the damage they cause tends to be most visible in the summer especially if trees are drought stressed with a reduced capacity to transport water to other parts of the tree. A build up of *Phytophthora* results in the death of fine feeder roots, even root and stem girdling, so trees may die suddenly when under water stress or show signs of marked decline. The predicted warmer climate will result in milder winters in the UK, and so may well provide more suitable conditions for many of these root-attacking Phytophthoras.

Another disease which very relevant to climate change is sooty bark disease of sycamore (*Cryptostroma corticale*). This fungus can survive as latent infections in wood, developing disease in response to water stress and high temperatures (around 25°C) (Dickenson and Wheeler, 1981). Disease outbreaks occur in the years immediately following hot, dry summers, particularly when the mean monthly temperature of more than one summer month equals or exceeds 23°C (Young, 1978). Currently, the disease tends to occur predominantly in the south east of England (Gibbs, 1997) although it was reported as far north as Yorkshire in 1996 following the 1995 drought (FR DDAS database). *C. corticale* is therefore not only expected to cause damage more frequently, but also extend its northern range into Scotland under expected climate change scenarios.

For the likely common pests of SRF species, climate change will influence their distribution, abundance and impact. Increases in temperature will affect the development rate of most insects, and in consequence the number of generations per year is likely to rise with associated damage to host plants. The northern distribution of many tree pests is also determined by climatic factors, so extension to range are probable, especially for recently arrived insect pests from southern Europe. Moreover, as with pathogens, climate change will not only directly influence pest populations but also operate indirectly. Thus, Wainhouse (2008) suggests that one major effect of changes in rainfall patterns is likely to be an indirect one, as drought stress makes trees less resistant to pest attacks.

4. Conclusions

Reports made to the DDAS and PAS provide a valuable record of passive surveillance of the pests and diseases that affect many tree species in Britain. However, because the process depends on self reporting of any visible symptoms or signs on affected trees, many common and well recognised disorders often not reported as their cause is already known and therefore these disorders are frequently unrepresented in the statistics. Despite this, reviewing the DDAS and PAS records highlighted a number of current and potential pests and pathogens likely to impact on tree species under consideration for SRF, with several able to cause serious levels of damage. Some of these are also strongly associated with certain tree species and therefore likely to become common in SRF plantations. Apart from identifying potential pest and pathogen problems for SRF crops a number of other issues need to be kept under consideration to minimise health problems on any SRF plantings.

- The various *Armillaria* species that cause root rot and decay are very common and widespread on wooded or previously wooded sites. *Armillaria* species are also known to establish disease foci in plantations of other tree species grown on former arable sites (Rishbeth, 1988). All of the species under consideration for SRF have some susceptibility to *Armillaria* root rot and some, such as *Betula*, are highly susceptible. Therefore, site selection and host matching to minimise the impacts of *Armillaria* should be considered when establishing SRF. The many other decay fungi which often occur on the various tree species may only be significant over longer time scales than those required for SRF, although if coppicing systems are used some of the common decay fungi may start to have an impact, reducing the vigour of the coppice stools and causing break-out of new stems.
- Provenance selection may be critical for some SRF species in order to avoid genotypes/provenances that may be especially susceptible to some pathogens. This is already accepted for poplar clones in order to minimise the impact of *Melampsora* rust and bacterial canker, although plantations of single-stem poplars grown at the wider spacings required in SRF may prove less susceptible to rust infection as air movement within more open canopies should prevent the build-up of disease levels seen with SRC. Other examples of diseases where provenance selection could be vital to the success of SRF include *Marssonina betulae* on birch and *Pseudomonas savastanoi* on ash, as well as the need for disease resistant provenances of *Eucalyptus* spp. That said, in many cases there is only sparse information on what provenances have the best performance or need be avoided in relation to some of the most damaging pests and pathogens.

- There is widespread evidence that many new and even endemic pests and pathogens are readily moved globally and locally via infected but often symptom free nursery plants (see reviews by Brasier, 2009; Webber, 2010). Phytophthora pathogens are particularly suited to movement in this way, but there are also many examples of introduced insect pests that arrive unnoticed on imported plants and trees (Evans, 2010). Therefore, selection and inspection of new planting stock is a critical part of minimising the impact of pest and disease problems in newly established SRF plantations. If pathogens are associated with planting stock in nurseries, they then have the opportunity to transfer to a new site and build up over time and spread to other individual trees, sometimes with serious impact on the growing trees although symptoms can take several years to become apparent. This process has apparently occurred in planted shelterbelts of alder (including species of *A. glutinosa*, *A. cordata* and *A. incana*) which were initially symptom free but later proved to be infected with *Phytophthora alni* (Gibbs *et al.*, 2003).
- Apart from the care needed to ensure that new pathogens are not introduced onto sites via infected SRF planting stock, regular surveillance of the plantations (especially during the early stages) is important. Build up of certain pests and pathogens can be rapid and could potentially threaten the productivity of short rotation forestry, especially if trees are all of a similar age and sometimes of limited genetic diversity. In this context, climate change is also likely to play an increasing role in the build up of certain organisms such as insect herbivores and foliar pathogens.

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Deer fraying to young birch tree

Potential impacts of mammals on short rotation forest biomass crops

R.M.A. Gill and R.C. Trout

Summary

Novel silvicultural techniques and tree species are being proposed for short-rotation biomass forests in the UK. We review evidence to ascertain the risks of damage by mammals if these crops become more widely established in Britain. There are a number of aspects of the proposed silviculture which gives cause for concern. Planting container stock, especially in cleared sites, will make the young trees particularly vulnerable to damage by deer, rabbits and hares. Further, the short rotation and low-value product may make the use of tree protection uneconomic. Crops grown on rotations longer than 10 years will become increasingly vulnerable to damage by grey squirrels. Plantations established close to existing woodland are likely to suffer more damage from deer and squirrels than sites well away from woodland cover. In contrast, rabbits and voles are more likely to be a problem in former agricultural sites.

Amongst the tree species proposed, hybrid aspen is likely to be the most vulnerable to deer and sycamore to squirrels. The *Eucalyptus* species are the least likely to suffer damage. Given the novelties of both species and silviculture, it is recommended that trials comparing growth and performance between protected and unprotected plots of each species in several sites where the target mammalian pest species are present should be established as a priority. Once crops are established, regular monitoring of damage is recommended, to provide warning and evidence of any change in the levels of damage that may warrant additional management action.

1. Introduction

Browsing damage by mammals is likely to present some unique challenges for short rotation forestry. Although trees are most vulnerable to British mammals when very young, this represents a higher proportion of the rotation length than in conventional forestry, with the result that a greater proportion of short rotation crops will be vulnerable to damage. Further, since short rotation crops are intended for producing a

relatively low-value product, the use of high cost fencing specifications or other protection measures might prove uneconomic. The management of browsing impacts is likely to depend on species choice, wildlife management, and possibly also habitat management. This review focuses on these aspects of damage management, drawing on evidence that is likely to be relevant for the chosen tree species in British conditions.

2. Impacts of mammal damage

The most common form of damage caused by mammals to young trees is through browsing. Shoots, buds and leaves of most tree species are readily consumed by herbivores. The herbivores present in Britain include deer (six species: red deer *Cervus elaphus*, sika deer *C. nippon*, roe deer *Capreolus capreolus*, fallow deer *Dama dama*, muntjac deer *Muntiacus reevesi*, and Chinese water deer *Hydropotes inermis*), rabbits *Oryctolagus cuniculus*, hares (*Lepus europaeus* and *L. timidus*), squirrels (*Sciurus carolinensis* and *S. vulgaris*) and voles (*Microtus agrestis* and *Clethrionomys glareolus*). Besides browsing, deer will also damage trees using their antlers ('thrashing' or 'fraying') and can break stems to reach tips or fruit. Deer may cause some damage by trampling or pulling newly planted seedlings out of the ground,

Many of the mammals mentioned above can also damage trees by stripping bark. Rabbits and voles can remove bark, usually in winter, from small diameter stems or snip off almost the entire stem above ground. Squirrels remove bark from sapling and pole stage trees in spring or summer, most frequently from the upper crown, but less often also near the base of the stem. The larger species of deer (red, sika or fallow deer) can also strip bark from sapling and pole sized trees at any time of year.

For various reasons, bark stripping damage will usually be less serious than browsing damage. Timber quality is unlikely to be a concern for short rotation crops and unless ring-barked or extensively damaged, trees normally survive bark damage and will continue to grow (Gill, 1992a;b;c). Further, trees will not be vulnerable to bark damage by deer or squirrels until near the end of the rotation. However stands grown on a rotation beyond 10 years, (or less on a particularly favourable site) will become increasingly vulnerable to damage by squirrels. Squirrels typically select the fastest growing trees, and damage near the crown may be followed by stem breakage during a storm, potentially resulting in serious yield losses (Mayle *et al.*, 2009).

With the exception of voles, which sometimes cut the roots of recent transplants, most damage by mammals normally leaves the roots intact, a factor which gives trees some capacity for recovery from above-ground damage.

Mammals are always selective in their feeding and characteristically browse some species, or parts of the tree, more than others. Further, tree species also differ in their capacity to recover from damage. It is the combination of these effects that is of consequence to forestry. The amount of damage inflicted by mammals on young plantations can be extremely variable, and on occasions, all trees can be damaged and may be killed by browsing. Several factors contribute to this variation in damage, ranging from landscape – scale to choice of species and silvicultural techniques.

a) Factors affecting damage

Herbivore density

Estimates of deer populations in lowland habitats indicate that densities are typically in the range of 10-40 km⁻² (Gill *et al.*, 1997; Gill and Morgan, 2010). Deer populations are not naturally limited by disease or predators in Britain, so in favourable habitats, or in the absence of culling (or both), higher densities can occur, sometimes in excess of 100 km⁻² (Ward *et al.*, 1994; Cooke *et al.*, 1996). Favourable habitats include areas offering a combination of shelter and forage, such as mixed woodland/arable landscapes, or woodlands with a high proportion of young, establishment phase habitats (Welch *et al.*, 1990; Gill *et al.*, 1996). The establishment of short rotation crops is itself likely to enhance habitats and increase deer populations.

Studies aimed at exploring relationships between deer densities and their impacts have suggested that in upland environments there needs to be between 4-8 deer km⁻² or less to allow sufficient tree regeneration to survive (Beaumont *et al.*, 1995; Miller *et al.*, 1998, Scott *et al.*, 2000). In the lowlands, densities can be a little higher, around 14 deer km⁻² (Gill and Morgan 2010). Further, several recent studies have suggested that the relationship between impacts and deer densities is curvilinear, with impacts increasing with density more rapidly at low rather than high densities (Tremblay *et al.*, 2006; 2007, Ward *et al.*, 2008, Gill and Morgan 2010). These studies have focussed mainly on natural regeneration. There is little direct evidence of acceptable deer densities for plantation forests. However for reasons discussed later, newly planted young stands can sometimes be more susceptible than naturally regenerated stands to deer, so target densities for short rotation forests may need to be lower than the densities quoted above. Since the densities to keep impacts within acceptable limits are well below average, they emphasise the need to manage deer populations to prevent excessive damage.

There is plenty of evidence that damage by the smaller mammal species is also strongly linked to population density (Gill 1992b). As discussed below, habitat conditions that favour each species are often linked to higher levels of damage. In young plantations, as many as 40% of unprotected transplants have needed replacement following rabbit damage even at a relatively modest density of 10 ha⁻¹ (Gill *et al.*, 1995).

Influence of landscape and habitat

For all mammals, the proximity of existing woodland, or other suitable habitat is likely to have an important influence on damage (Moore *et al.*, 1999; Bulinski and McArthur 2000). Existing habitat may make a plantation more attractive by providing nearby cover and alternative food, as well as increasing local densities. For squirrels, damage has been found to be higher in stands close to mature stands of large seed-bearing trees (Kenward *et al.*, 1988; 1990), and for rabbits, damage is likely to be worse near established hedgerows and field margins (Gill 1992b). Amongst deer, there are distinct differences depending on their foraging requirements, for example damage by roe deer has been found to be greater closer to cover, but this is less evident for fallow and red deer, which forage further from cover (Thirgood and Staines 1989; Kay 1993).

The extent of vegetation cover is likely to affect the damage done by mammals, though in different ways for each species. Both voles and rabbits are very dependent on vegetation cover, and voles often achieve high densities in former agricultural land following cessation of grazing or cultivation (Trout *et al.*, 2004). Damage to young trees can be sharply reduced by treatments such as mulching or mowing, which reduce vegetation cover around the tree (Davies and Pepper 1989; MacVicker and Trout, 1994). Rabbits can benefit from the cover provided by brash or post-harvest stump windrows, making damage for the subsequent crop more severe.

Habitat conditions can influence browsing rates on seedlings in a variety of other ways, for example by providing an alternate food source, or by concealing seedlings making them difficult for animals to find (Miller *et al.*, 1982; 2006). Seedlings planted on recently cleared re-stock sites are particularly conspicuous and vulnerable to damage by deer. As vegetation recovers however, habitat conditions become ideal for deer, and survival and growth of seedlings depends more on the relative palatability in comparison to surrounding vegetation and the ability of seedlings to survive and compete after damage.

Seedling type

Seedlings can be prepared in a variety of ways before planting and these treatments, as well as the site conditions in which they grow can influence susceptibility to damage. In general, larger or older seedlings are more likely to survive damage than younger or smaller seedlings (Gill 1992c). Several studies have shown that container-grown plants are more prone to damage than bare-root stock (Bergstrom and Bergqvist 1997; 1999; McArthur and Appleton 2004), although some authors have found no difference (Myers *et al.*, 1989).

One study has found higher browsing rates on seedlings given fertiliser in the nursery than on seedlings fertilised on site after planting (Close *et al.*, 2004), suggesting that some of the differences commonly observed between seedling types may arise because

nursery treatments make seedlings more attractive to herbivores which is reflected in higher rates of damage once planted out. Some studies have reported that planted seedlings are more prone to damage than naturally regenerated seedlings (Ballon *et al.*, 1999; Saniga 2003; Cooke and Mutze, pers comm), which may be for the same reasons, or that seedlings obtain some protection from other plants.

Tree species and clonal differences

The observation that mammals browse some tree species more than others has led to attempts to rank species according to feeding preference and in doing so identify species that are most or least likely to be damaged (Mitchell *et al.*, 1977; Gill 1992a). Comparisons between ranks suggests that while some species appear to have consistent relative palatability, others are less consistent, and for example may be noted as being vulnerable in some cases or avoided in others (Gill and Beardall 2001). There are a number of reasons why this may occur, related both to the environment or characteristics of the tree.

Several studies have revealed genetic differences in palatability to mammal damage, between either clones or provenances. Trials in Britain have revealed sharp differences amongst clones of poplar and willows to rabbits (Trout, R. unpubl). In Finland, differences in palatability of birch clones to both hares *Lepus timidus* and moose *Alces alces* (Rousi *et al.*, 1991; 1996; Jia *et al.*, 1997), have been linked to the concentration of terpenoids (papyriferic acid) and phenols in the resin of young shoots (Tahvainen *et al.*, 1991; Palo *et al.*, 1997).

Environment and soil fertility

Recent studies have shown that palatability can be mediated by environmental conditions, for different possible reasons. Studies on *Betula pendula* and *Eucalyptus globulus* have found that clonal differences in palatability can change or disappear if grown on a more fertile soil (Laitinen *et al.*, 2002; O'Reilly-Wapstra *et al.*, 2005). In the latter case, the concentration of tannins decreased but essential oils increased in response to fertilisation, suggesting that the effect of fertilisation has potentially complex effects that may have different implications for different herbivore species. In a separate investigation on silver birch, the selection of different clones planted at one site by moose revealed a decreasing preference with increasing latitude of origin, indicating that palatability was determined by the timing of growth and bud burst (Vihera and Heikkilä 2006).

b) Evidence of damage to candidate species of interest:

Studies of browsing damage in one forest over a period of several years has shown that preferences amongst some browse species appear to change over time (Boulanger *et al.*, 2009). This could arise from changes in the relative abundance of each food species

(Crawley 1983). Attempts to investigate whether browsing pressure on trees varies for this reason has produced differing results. One study, concluded that selection by roe deer was largely frequency-independent (Verheyden-Tixier *et al.*, 1998), however Eiberle and Bucher (1989) found that browsing on some palatable species was affected by the abundance of other species. Nonetheless there is plenty of evidence that species selection is consistent enough to be of practical benefit in forest management. The following section highlights evidence for each of the chosen species.

Eucalypts: shining gum (*Eucalyptus nitens*), Tingiringi gum (*Eucalyptus glaucescens*), cider gum (*Eucalyptus gunnii*)

Eucalyptus species have not yet been widely used as commercial forest trees in Britain, however *E. gunnii* has been planted extensively as an amenity tree. Anecdotal reports generally indicate that *E. gunnii* is relatively unpalatable to deer, but is eaten by rabbits (Coles 1997). *E. nitens* is widely reported to be damaged by rabbits in Australia (O'Reilly and McArthur 2000; Bulinski 2000), but apparently not by red deer in Portugal (Coles 1997). *Eucalyptus* trials in Oregon by Hunt (1983) have revealed the *E. gunnii* is very susceptible to damage by deer (84.6% trees severely browsed by black-tailed deer - *Odocoileus hemionus*) but *E. nitens* and *E. glaucescens* in contrast were relatively undamaged (95% and 69% undamaged respectively).

Field studies on browsing by rabbits and possums to first-year seedlings of *E. nitens* have indicated that browsing can adversely affect growth, but that subsequent survival is usually unaffected (Bulinski and McArthur 1999; Bulinski 1999; Mar and McArthur 2005). On the other hand experiments based on artificial damage have indicated that survival can be affected, and that yield losses are disproportionately affected by simulated browsing: removal of 50% of the crown for example resulted in a 25% yield loss after seven years, but complete crown removal resulted in yield losses of 71-97%, through a combination of mortality and reduced growth (Wilkinson and Neilsen 1995).

Rauli (*Nothofagus procera*)

Very little information is available on the susceptibility of *N. procera* to mammals. In response to some concerns about damage, a brief survey was carried out by Forest Research in 1978 to assess its vulnerability in Britain. However, very few reports of damage were obtained, possibly because many of the stands were protected or already too tall for browsing at the time of the survey. Nonetheless, there were some reports of browsing by rabbits and both browsing and fraying by deer, and one forester noted that damage to *Nothofagus* species in general appeared to be increasing, but *N. procera* was damaged less than a nearby stand of *N. obliqua*.

In native Southern beech forests, there are many reports of browsing by deer and other ungulates, indicating that damage can be sufficient to limit regeneration of *N. dombeyi*, *N. antarctica* and *N. pumilio* in S. America (Veblen *et al.*, 1989; 1992; Cavieres and

Fajardo 2005) and *N. solandri* in New Zealand (Husheer *et al.*, 2006). In both countries however, reports suggest that browsers have a relatively low preference for *Nothofagus* seedlings in comparison to other native tree species (Veblen *et al.*, 1989; Forsyth *et al.*, 2003). Sika deer however appear to damage *N. solandri* more than red or fallow deer (Husheer *et al.*, 2006). There is an absence of information referring specifically to the effects of herbivores on regeneration of *N. procera*.

Alders: Italian alder (*Alnus cordata*) and red alder (*Alnus rubra*)

In common with the other non-native species, there is little information available on the susceptibility of these two species to mammals in Britain, although Italian alder planted on landfill sites has been reported to suffer from deer browsing (Anon 2000). Native alders *Alnus glutinosa*, have been reported to be of either low, or moderate-low preference for deer (Gill 1992a; Kay 1993; Putman 1994; Moore *et al.*, 1999; Stone *et al.*, 2004), but susceptible to damage by rabbits (Trout, R. pers comm). Looking beyond the UK, black-tailed deer in the USA have been reported to have relatively low preference for red alder (Radwan and Crouch 1974), however other authors have found that it may be severely browsed, and is vulnerable to damage (Niemiec *et al.*, 1995; Courtin and Brown 2001). In the USA, red alder is known to be browsed by elk (*Cervus elaphus*) more in autumn than other times of the year (Nelson and Leege 1982), which has been suggested to be due to a seasonal decline in the concentration of phenolic compounds (Gonzalez-Hernandez *et al.*, 2000). In apparent contrast to red alder, studies in Italy suggest that Italian alder is browsed more in early summer than winter (Casanova and Sonago 1988).

Hybrid aspen (*Populus tremula x tremuloides*) (*x wettsteinii*)

Although hybrid aspen has been grown for some years, the trials were limited in scale and in some cases also fenced, therefore yielding relatively little information about browsing at least by larger mammals.

Some authors however note that hybrid aspen is browsed by voles (Lieseback *et al.*, 1999) and roe deer (Tullus *et al.*, 2007). Both parent species are however well known to be susceptible to browsing by many ungulate species, and it is therefore likely that the hybrid will also prove to be very palatable. In North America, *P. tremuloides* is widely reported to be heavily browsed by deer *Odocoileus hemionus* and elk (*Cervus elaphus*), to the extent that there are concerns that the species may be eliminated in some areas (Kay 1997; Kay 2001; White *et al.*, 2003; Weisberg and Coughenour 2003). Similarly, *P. tremula* is widely reported to be preferred in several European studies by both deer as well as voles *Clethrionomys glareolus* and hares *Lepus timidus* (Uerkermann 1960; Armani and Franzoi 1998; Hjalten *et al.*, 2004; Zakrisson *et al.*, 2007; see also refs in Gill and Beardall 2001).

By virtue of having rapid early growth, good recovery from damage may be achieved provided initial damage is not too severe.

Sycamore (*Acer pseudoplatanus*)

Most investigations carried out in Britain suggest that sycamore is amongst the less palatable species to deer, either suffering moderate, (Chard 1966, Kay 1993) or very low levels of damage (Moore *et al.*, 1999; Harmer *et al.*, 2001). In contrast several studies from continental Europe describe it as one of the most vulnerable species (Kammerlander 1978; Eiberle and Wenger 1983; Ammer 1996; Cermak and Mrkva 2006). Sycamore is one of the most susceptible species to damage by grey squirrels. It is usually reported as being most vulnerable between 10-40 years of age, (Shorten 1957; Rowe and Gill 1985) although damage to younger saplings can occur (Gill *et al.* 1995). . Damage should therefore be expected to be most serious towards the end of the rotation, especially if extended to 15-20 years or more.

Sweet chestnut (*Castanea sativa*)

Sweet chestnut has traditionally been used mainly for coppicing in the UK. Most reports suggest that it is relatively palatable, experiencing moderate or high levels of browsing in coppice woodland (Kay 1993; Putman 1994). On the other hand, relatively low levels of leader damage were reported by Moore *et al.*, 1999 to planted trees and Downes and Whelan (1992) reported relatively low levels of damage by rabbits in Ireland. Sweet chestnut is vulnerable to damage by grey squirrels.

Silver birch (*Betula pendula*)

The majority of reports suggest that silver birch is a relatively unpalatable species (Kay 1993; Putman 1994; Kullberg and Bergstrom 2001), and it is able to recover readily from any damage that does occur (Miller *et al.*, 1998). Most browsing occurs in summer rather than winter (Cummins and Miller 1982; Gill 1992a). It is nonetheless clear that there is considerable variation in the effects of browsing on birch. In a review of 10 European enclosure studies, the density of birch seedlings was lower in control plots (i.e. unfenced) in 7 cases but higher in the remaining 3, indicating that the presence of deer can sometimes favour birch regeneration (Gill and Beardall 2001). In the UK anecdotal reports suggest that birch is browsed less by deer in the lowlands than the uplands, although neither the extent nor reasons for this have been investigated. Rabbits will damage small birch trees of many clones, though shoot terpene capsule density may be related to unpalatability.

Ash (*Fraxinus excelsior*)

The evidence for the susceptibility of ash to browsing is somewhat varied. There are several reports that ash is one of the most preferred tree species to deer (Bows 1997; Gill and Beardall 2001; Modry *et al.*, 2004); rabbits (Downes and Whelan 1992) and voles (Commarmot 1981). In addition, browsing reduces survival rates (Eiberle and Nigg 1987; Harmer 2001) and can result in a substantial loss of both seedlings as well as

coppice stools, (Cooke 1998; Mountford and Peterken 1998). Conversely there are also a number of reports that suggest that it is a relatively unpalatable species (Kay 1993; Putman 1994; Moore *et al.*, 1999; Kullberg and Bergstrom 2001) and it is unclear why such an apparent discrepancy exists. One study has however found differences in palatability between provenances of ash to both voles and rabbits (Moraal and Goedhart 1999). It is possible that some of the variation in palatability has arisen because of differences in alternative food sources. However, at least two studies that have investigated this effect have concluded that ash remained the most preferred species to deer, in spite of differences in the abundance of other palatable species (Kossak 1976; Chevalier-Redor *et al.*, 2001).

3. Anticipated effects of climate change

Deer and rabbit populations are likely to benefit directly and indirectly from many of the effects of climate change. Warmer winter weather and longer growing seasons are likely to improve their juvenile growth and survival, while elevated CO₂ will increase vegetation productivity (Irvine *et al.*, 2007). The effects of climate change on nutritional chemistry appear so far to have been investigated only in silver birch. The concentration of papyriferic acid increased in seedlings grown in elevated temperatures, but only in the upper, not lower, part of the stem leading to the conclusion that palatability to hares may decrease, but voles, which feed on the lower part of seedlings, will be unaffected (Kuokkanen *et al.*, 2004).

4. Conclusions

On the basis of published evidence of browsing on these species, it is possible to suggest a relative palatability for most of the intended tree species (Table 1). However, given that evidence of damage is scarce or contradictory for some species in British conditions, it is recommended that trials comparing growth and performance between fenced and unfenced plots of each species in several sites where the target mammalian pest species are present should be established as a priority. This will provide evidence at an early stage in the establishment of short rotation crops of the levels of damage to be expected, the capacity of each species for recovery and hence the need for protection or wildlife management.

Table 1. Relative palatability amongst tree species to mammals.

	Palatability			
	Most ←			→ Least
Deer	Hybrid Aspen	Sweet Chestnut, Ash, Sycamore	Italian & Red Alder, Rauli, Silver Birch	<i>Eucalyptus</i> spp
Rabbits	Insufficient evidence for consistent species selection			
Voles	Sycamore	Ash	Silver Birch	
Grey Squirrels	Sycamore	Sweet Chestnut Ash, Silver Birch	Hybrid Aspen, Rauli, Italian & Red Alder	<i>Eucalyptus</i> spp

It is usual practice in plantation forestry in Britain to attempt to establish the less palatable tree species (e.g. Sitka spruce *Picea sitchensis* and Corsican pine *Pinus nigra*) without fencing or other forms of tree protection. Nonetheless, deer management is usually applied to prevent damage becoming too severe. Equally, squirrel control is recommended where red squirrels are not present. Fencing or tree guards are usually only used for the most sensitive species, or where wildlife management is impractical. It is suggested that a similar approach will be possible for short rotation forests, with establishment of the less palatable species being possible without the need for protection, although it is likely that some level of wildlife management will be needed in most sites. Forest management based on planting is potentially more risky than natural regeneration, particularly for container grown seedlings planted into unprotected sites. It is possible that browsing on some species may increase over time, once they cease to be novel species for the mammals present in any particular site. Regular monitoring of damage is recommended, to provide warning and evidence of any change in the levels of damage that may warrant additional management action.

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Contact details

Forest Research main addresses:

Alice Holt Lodge, Farnham, Surrey, GU10 4LH, UK

Tel: +44 (0) 1420 22255

Fax: +44 (0) 1420 23653

Northern Research Station, Roslin, Midlothian, EH25 9SY, UK

Tel: +44 (0) 131 445 2176

Fax: +44 (0) 131 445 5124

research.info@forestry.gsi.gov.uk

www.forestry.gov.uk/forestresearch

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