MM04: Agroforestry

Background

Agroforestry refers to the practice of growing trees in crop or livestock systems. Trees have historically been present in crop and pasture land, though mechanisation of agriculture over the latter half of the 20th century has led to the decline of such mixed systems (Eichhorn et al., 2006). Formal (re)integration of trees to arable and livestock production systems is of interest, as resulting woody biomass inputs represent a key route to soil carbon sequestration (Lorenz & Lal, 2014). Tree roots modify the quality and quantity of belowground C inputs, and recover nutrients and moisture from lower soil horizons (Lorenz & Lal, 2014); additional benefits can include increased yield and agroecosystem resilience. Agroforestry systems also induce a microclimate effect, improving the climate change adaptability of vulnerable agroecosystems (Mbow et al., 2014; Lasco et al., 2014). Agroforestry encompasses several specific practices and can be applied to intercropped arable systems, as an approach to fallow management, as shelter belts for protection from wind erosion, and to grazing systems (Nair et al., 2010). Disadvantages may include the competition of trees with understory (crop and grass) species for light and water, and potential for increased bushfire incidence or severity in dry areas (Lorenz & Lal, 2014). Optimisation of agroforestry systems therefore requires matching of tree and crop types in order to maximise positive interactions.

Baseline and uptake

Baseline agroforestry uptake is difficult to define accurately; in part, this is due to definitional difficulty in separating normal agricultural practice, which may include trees, from specialist agroforestry systems (Burgess & Rosati, 2018). A recent, conservative estimate (den Herder et al., 2017) put extent of agroforestry systems in the EU27 at 8.8% of agricultural area; this is nonetheless much higher than national estimates would suggest (Burgess & Rosati, 2018). The vast majority of these systems are centred around the Mediterranean basin (Burgess & Rosati, 2018). Lower uptake in more northerly regions appears to be largely due to potential for short term yield losses, (perceived) potential for interspecific competition, and lack of robust economic advice to assuage these concerns (Martineau et al., 2017). In Europe, there is an estimated area potentially available for uptake of agroforestry of 90 and 50 million ha of pasture and arable land respectively (Aertsens et al., 2013)

Mitigation summary

Effect on GHG categories*	Rating	Notes
Enteric CH_4		
Manure CH_4		
Manure N_2O		
Soil N_2O: residue N		
Soil N_2O: applied N		
Soil N_2O: grazing		
Energy CO_2: fieldwork		
Energy CO_2: other		
CO_2 liming and urea		
CO_2 sequestration below ground	-	
CO_2 sequestration above ground	-	
Pre-farm emissions		
Post-farm emissions		
Substitution of higher C products		
Production increases by more than the emissions		
Confidence in mitigation effect	High	
Cost-effectiveness**	Moderate to high	Depending on system type
Confidence in cost-effectiveness	High	

* ”-“ GHG reduction, “+”: GHG increase, “ ”: no significant effect
** low: =< £0/tCO_2e, moderate: £0/tCO_2e< >SCC, high: >SCC

Related measures and potential interaction

Measure	Impact on other measures
1—17. All cropland management measures	Reduces planted area, light availability to understory, and crop yield
6. Agrivoltaic systems	Likely to preclude implementation of this measure due to light competition

Inclusion in other marginal abatement cost curves

UK 2008	UK 2010	UK 2015	Ireland 2012	France 2013	France 2019
No	No	No	No	No*	?

*Restoration of degraded soils (including acidified soils) was considered, but rejected owing to limited applicability.

Types of system

***Typology*.** Agroforestry systems are highly variable in form and function. A typology of agroforestry systems is as follows:

Trees in cropland are often referred to as a **silvoarable** system (e.g. Palma et al., 2007; Graves et al., 2007). Trees in grassland may be called **silvopastoralism** (Torralba et al., 2016; Feliciano et al., 2018). **Intercropping** and **alley-cropping** are terms in general use for mixed cropping systems, but may also be used to refer to agroforestry systems (e.g. Nissen et al., 1999; Lamerre et al., 2015). In such cases a tree species is specified as the second crop; this terminology is most common where the tree species is grown as a crop in short rotation (e.g. a 3-year short rotation coppice). **Windbreaks** and **shelter belts** refer to trees integrated to an agricultural system specifically to provide shelter to crops or protection from wind erosion (e.g. Frelih-Larsen et al., 2014; Posthumus et al., 2015). **Hedges** are not typically considered agroforestry systems, but are often assessed alongside as another way of integrating woody biomass into agricultural systems. (e.g. Pellerin et al., 2013).

***Implementation**.* Key variables which may change between different implementations of agroforestry systems are:

a) **Tree layout**. Trees may be 'scattered', i.e. without any specific planting pattern, or linearly planted. The former is most likely to be found in semi-natural pasture systems, while the latter is more compatible with mechanised cultivation and harvesting, and is therefore more suitable for arable systems or renovated pasture (Jose et al., 2004; Eichhorn et al., 2006).
b) **Tree spacing and density**. Where trees are linearly planted, spacing between lines affects (i) number of trees per hectare, (ii) reduction in crop or pasture area, (iii) considerations relating to mechanised harvesting of crops, if applicable, and (iv) competition between trees and understory plants for light, water etc. (Chirko et al., 1996; Eichhorn et al., 2006).
c) **Tree usage**. Trees may be planted solely for their amenity value to the existing system (see #AGF.1), or for a number of commercial uses, including (i) fruit production, (ii) fodder production, (iii) fuel production, and (iv) timber production (Jose et al., 2004; Eichhorn et al., 2006).
d) **Tree harvest type**. Trees grown for timber or fruit are typically left in place for a number of decades, and killed at harvest. Fuel or fodder harvest may be non-lethal and occur with greater frequency (Jose et al., 2004; Wolbert-Haverkamp & Musshoff, 2014).
e) **Tree-crop pairing**. A large number of different tree and crop species have been trialled; Table #AGF.1 presents a literature sample of these pairings.

Table 1. Pairings of tree and crop species in arable agroforestry systems. NS = not specified.

Tree	Crop	Tree use	Location	Source
Populus spp.	Wheat, barley, oil seed rape	Timber (experimental plot)	United Kingdom	Burgess et al. (2003)
Paulownia spp.	Wheat	NS (experimental plot)	North China Plain	Chirko et al. (1996)
Mixed *Rosaceae*	Vegetables, grape vines, ground fruit	Fruit	France	Coulon et al. (2001) in Eichhorn et al. (2006)
Peach, walnut, olive	Grape vines	Fruit and fuel	France	
Almond	Cereals, fodder legumes, grasses	Fruit	Sicily	Cullotta et al. (1999) in Eichhorn et al. (2006)
Cherry	Fodder beet	Fruit	Saxony, Germany	Eichhorn et al. (2006)
Cork oak	Wheat	Cork production	Sardinia, Italy	
Oak	Arable/pasture rotation	NS	Castilla-Leon, Spain	
Oak	Arable	Fodder (leaves stripped annually)	Askio, Greece	
Olive	Wheat	Fruit	Lazio, Italy	
Poplar	Wheat	NS (experimental plot)	Vezenobres, France	
Walnut	Mixed vegetables	Fruit and timber	Campania, Italy	
Mixed *Rosaceae*	Maize, other cereals, vegetables, fruits	Fruit	Spain	(INE, 2002) in Eichhorn et al. (2006)
Poplar, silver maple	Maize, soy	NS	Ontario, Canada	Jose et al. (2004)
NS	Forage grasses and legumes	NS	New Franklin, Missouri, USA	Lin et al. (1998)
Populus spp. and/or Salix spp.	Wheat	Fuel	Bristol, United Kingdom	Nichols et al. (2000)
Eucalyptus	Cabbage	Timber	Philippines	Nissen et al. (1999)
Walnut, poplar, cherry	NS	Timber	France	Pellerin et al. (2013)
Hickory, walnut, oak, maple	Alfalfa, maize, soybean	Timber	Canda	
Fig	Cereals	Fruit and fodder	Crete	Schultz et al.(1987) in Eichhorn et al. (2006)
Mulberry	Maize, fodder legumes, vegetables	Fruit	Northern Greece	
Pear	Cereals, tobacco, vegetables, grape vines	Fruit and fuel	Northern and central Greece	

Whilst much recent research focuses on agroforestry in arable systems (Burgess & Rosati, 2018), silvopastoral systems (agroforestry in grazing land) are also prevalent in the literature, with examples in Spain (Eichhorn et al., 2006), France (Pellerin et al., 2013), Uruguay (Fedrigo et al., 2018), Brazil (Xavier et al., 2014) and the United States (Lin et al., 1998).

Literature estimates of abatement and cost effectiveness
Section #AGF.4.1. provides an overview of cost effectiveness estimates of abatement from agroforestry systems available in the published literature. The following sections (#AGF.4.2—#AGF.4.5.) discuss the variables leading to the derivation of these estimates.

Overview. Pellerin et al. (2013) estimated an abatement rate for agroforestry of between 0.4 – 4.97 tonnes CO_2-eq ha^{-1} $year^{-1}$ (best estimate = 3.7), at a cost of €13 – €118 tonne CO_2-eq^{-1} (best estimate = €14). This value was estimated as a finite potential linearised over a 20-year period. The hypothesised system had low tree density in both croplands and grasslands (and was not specific to an understory crop), and considered production of walnut, poplar and cherry for timber.

Frelih-Larsen et al. (2014) estimated an abatement rate of 138 kg CO_2-eq ha^{-1} $year^{-1}$ for shelterbelt agroforestry, equating to 0.51 tonnes CO_2-eq ha^{-1} $year^{-1}$. This abatement focused on soil carbon sequestration only, and excluded biomass. The authors made an assessment of cost, and classified the measure as low cost-effectiveness given high implementation and maintenance costs (€2,000 – €6,000 ha^{-1} and €220 – €270 ha^{-1} respectively).

Eory et al. (2015) estimated an abatement rate of agroforestry of 7.34 tonnes CO_2-eq ha^{-1} $year^{-1}$ in grasslands and 8.84 tonnes CO_2-eq ha^{-1} $year^{-1}$ in arable systems. This translated to a cost effectiveness of £30 tonne CO_2-eq^{-1} in grassland and £15 tonne CO_2-eq^{-1} in arable. Measure costs were lower in arable land given the lack of requirement for protecting trees from livestock, and higher assumed soil carbon sequestration increased the abatement rate.

Thomson et al. (2018) estimated GHG abatement of 2.7 Mt CO_2-eq and 5.9 Mt CO_2-eq by 2050 for 'medium ambition' (5% of agricultural land) and 'high ambition' (10% of agricultural land) scenarios respectively. The authors do not report per-hectare abatement rates (AR), but broad calculation based on reported area and potential gives an estimated AR of 7 tonnes CO_2-eq ha^{-1}. The authors note that crop yield and production may be impacted by agroforestry, but do not assess this quantitatively given its dependence on tree spacing and growth rate.

GHG abatement. A key reason for discrepancy in agroforestry abatement rates relates to abatement scope. For example, Frelih-Larsen et al. (2014) estimate a sequestration rate of 0.14 tonnes CO_2-eq ha^{-1} $year^{-1}$, while Aertsens et al. (2013) estimate a rate of 10.08 tonnes CO_2-eq ha^{-1} $year^{-1}$. A key reason for this discrepancy was a difference in the assessment scope; the larger value also included sequestration in tree biomass, which is in itself variable. Based on a review of published values, a recent estimate from Martineau et al. (2017) is a range of 0.15 – 0.88 tonnes CO_2-eq ha^{-1} $year^{-1}$. Table 2 summarises published literature values with abatement scopes.

Table 2. Abatement rates utilised in published assessments of agroforestry.

Study	Tree	Crop	Region /scale	Derivation	Tree density (trees ha^{-1})	Sequestration (tonnes CO_2-eq ha^{-1} year^{-1})			Abatement based on		
						Best estimate	Minimum	Maximum	Soil carbon	Biomass	Other GHG effects
Pellerin et al. (2013)	Walnut, cherry, poplar	Arable	France	Literature	30-50	3.75	0.45	5.02	Y	Y	Y
Eory et al. (2015)	NS	Arable	United Kingdom	Literature	NS	8.84			Y	Y	NS
	NS	Grass	United Kingdom	Literature	NS	7.34			Y	Y	NS
Frelih-Larsen et al. (2012)	NS	NS	United Kingdom	Literature	NS	0.51			Y	N	N
Aertsens et al. (2013)	NS	NS	Europe	Literature	NS	10.1			Y	Y	N
Martineau et al. (2013)	NS	NS	Europe	Literature	NS		0.15	0.88	Y	N	N
Hamon et al. (2009)	Poplar	NS	Vézénobres, France	Empirical	140	23.7			N	Y	N
	Black walnut	NS	Vézénobres, France	Empirical	70	2.48			N	Y	N
	Walnut	NS	Monpellier, France	Empirical	NS	3			N	Y	N
	Walnut	NS	Monpellier, France	Empirical	NS		0.1	0.5	Y	N	N
Upson & Burgess (2013)	NS	NS	United Kingdom	Empirical	NS	1.5			Y	N	N

The Forestry Commission's Carbon Lookup Tables (West & Matthews, 2012) provide species-specific biomass sequestration estimates for planted forestry. Recalibration of this data to provide estimates for agroforestry-typical tree densities enables estimation of biomass sequestration by different species in an agroforestry system (Table 3).

Table 3. Monte Carlo-simulated estimates of biomass carbon sequestration by tree species in the United Kingdom. Monte Carlo simulation assumes a) 20m row spacings in arable or pasture land, b) clearfell at between 20 and 40 years, and c) equal likelihood of biomass carbon persisting post-clearfell (e.g. used as construction timber) or being reemitted (e.g. used as woodfuel).

Species	Carbon sequestration in biomass (t CO_2-eq ha^{-1} $year^{-1}$)			
	Mean	**Std. Dev.**	**2.5% C. I.**	**97.5% C. I.**
Beech	4.5	1.6	0.8	7.2
Corsican pine	5.0	1.3	2.6	7.7
Douglas fir	6.9	1.8	3.9	10.5
European larch	4.1	1.1	2.2	6.3
Grand fir	6.7	1.7	3.8	10.2
Hybrid larch	4.2	1.1	2.4	6.6
Japanese larch	4.3	1.1	2.5	6.8
Leyland cypress	6.7	1.8	3.1	10.1
Lodgepole pine	4.1	1.3	1.6	6.5
Noble fir	5.1	1.6	1.4	8.0
Norway spruce	4.3	1.4	1.4	7.3
Oak	4.6	1.4	1.6	7.1
Scots pine	3.3	1.3	0.5	6.0
Sitka spruce	5.5	1.8	2.4	9.4
Sycamore, ash and birch	6.0	1.5	3.4	9.3
Western hemlock	6.8	1.8	3.7	10.3
Western red cedar	6.0	1.6	2.7	9.1

Investment and maintenance costs. For **short rotation coppicing** (SRC), assessed as a monoculture system, estimates of investment vary from £1,730 ha^{-1} (Glithero et al., 2013) to £1,796 ha^{-1} (Mitchell et al., 1999). Given the high investment, Mitchell et al. (1999) estimated that cash flow for a short rotation coppicing (SRC) system would be net negative until year 16 of a 26-year cycle. Lamerre et al. (2015) assessed an alley-cropped implementation of SRC with a tree density per ha approximately half that of monoculture SRC; more typical intercropped agroforestry systems utilising SRC might have 20 – 40% of the total trees per hectare (Chirko et al., 1996; Lamerre et al., 2015). While such systems may provide amenity value to the intercropped system, crucially, SRC has a lower gross margin than many arable crops (Mitchell et al., 1999; Wolbert-Haverkamp & Musshoff, 2014), hindering incentives for its implementation in agroforestry systems.
Utilisation of short rotation coppicing normally requires substantial herbicide application (typically glyphosate) to remove coppice stools (Mitchell et al., 1999; Glithero et al., 2013). If these are intercropped and rotated, during the time it takes for the associated root systems to break down, these structures may impede ploughing (Mitchell et al., 1999), limiting cultivation of crops.

Timber production. For French timber-based agroforestry systems (see #AGF.4.1 for system description), Pellerin et al. (2013) estimate annualised costs of €17 – €45 and €50 ha^{-1} $year^{-1}$ for planting investment and maintenance of trees respectively.

Impacts on crop production. For arable production under timber agroforestry, Pellerin et al. (2013) estimate a cost of €80 – €124 ha^{-1} $year^{-1}$ resulting from lost crop production. It is not clear to what extent this results from the loss of cropping area (estimated at 5% by Pellerin et al., 2013) and to what extent (if any) from loss of yield per unit cropping area. For grassland, Pellerin et al. (2013) estimate a production loss cost of €42 – €70 ha^{-1} $year^{-1}$ and the same 5% loss of production area.

Loss of yield from understory crops may also a be concern, due to interspecific competition for light, nutrients or water (Jose et al., 2004). In temperate climates, competition for light may be the greatest issue. Impact on yields is highly variable, depending on factors including tree spacing, tree and crop species, climate, and management (Jose et al., 2004). Literature reported ranges are summarised in table #AGF.4 and fig. . Photosynthetic pathway may be an important factor in determining yield impacts; Lin et al. (1998) found, in Missouri, that warm-season forage grasses and legumes (i.e. C_4 photosynthetic pathway) are significantly more likely to suffer yield loss in the presence of agroforestry than cool-season (C_3) species of similar functional groups.

Table 4. Crop yield impacts of agroforestry systems reported in the literature.

Tree	Crop	Region	Crop yield range (% lone yield)	Source
Populus spp.	Wheat, barley, oil seed rape	United Kingdom	*See fig. #AGF.1*	Burgess et al. (2003)
Paulownia spp.	Wheat	North China Plain	80—97	Chirko et al. (1996)
NS	Cool-season grasses	Missouri, USA	85—124	Lin et al. (1998)
NS	Cool-season legumes	Missouri, USA	43—86	Lin et al. (1998)
NS	Warm-season grasses	Missouri, USA	46—74	Lin et al. (1998)
NS	Warm-season legumes	Missouri, USA	37—83	Lin et al. (1998)
Eucalyptus	Cabbage	Philippines	84—85	Nissen et al. (1999)
Poplar, silver maple	Maize	Ontario, Canada	75	Jose et al. (2004)
Poplar, silver maple	Soybean	Ontario, Canada	79	Jose et al. (2004)

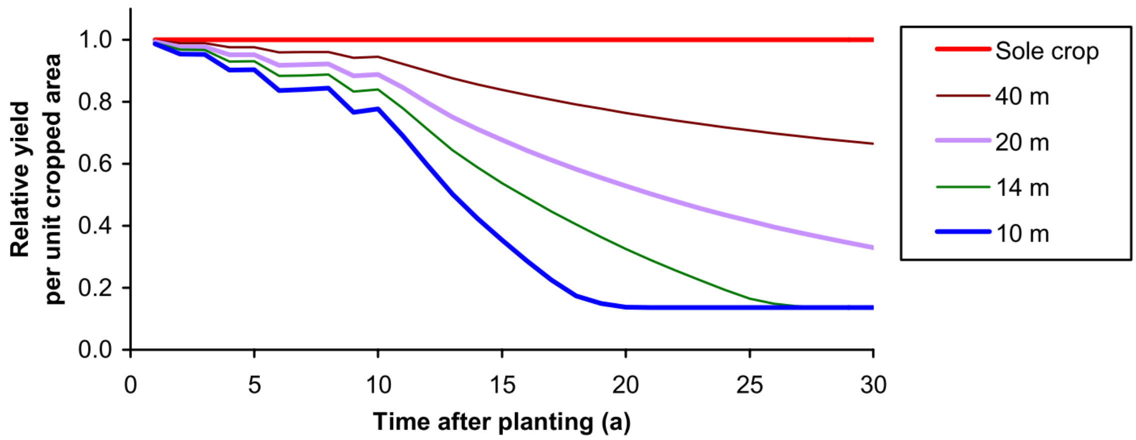

Fig.1. Modelled yield impacts in a United Kingdom-based agroforestry system comprising wheat, barley and oil seed rape rotations with poplar trees (*Populus* spp.). Yield effects are is highly dependent on row spacing and stand age. Adapted from Burgess et al. (2003).

Outputs from agroforestry. Pellerin et al. (2013) estimate revenue of €84 - €147 ha^{-1} $year^{-1}$ from timber production by agroforestry systems. Willow and poplar grown in SRC may return £42.50 odt^{-1} (oven dried tonne) with 3-year yields of around 3kg oven-dried wood per plant (Mitchell et al., 1999). With a SRC typical within-row spacing of 0.5m (Lamerre et al., 2015) and agroforestry-typical between row spacings of 5 – 10m (Chirko et al., 1996), estimated revenue of £85 – £170 ha^{-1} $year^{-1}$ from fuel wood could be achieved from the intercropped system, similar to the values suggested by Pellerin et al. (2013). Where trees are not harvested, tree leaves may also be stripped annually to provide fodder for livestock (Eichhorn et al., 2006); such practices serve to reduce the shading effect on understory species, as well as offsetting costs of fodder production or purchase.

Bottom-up assessment of agroforestry marginal abatement cost: assumptions and data sources

This fiche identified agroforestry as a potentially high-abatement measure with a number of associated complexities. This potential, coupled with the lack of certainty around the variables driving differences in literature-estimated marginal abatement costs, meant that we chose to collate this information and perform a detailed, bottom-up estimation of the marginal abatement cost effectiveness of this measure in the context of arable agriculture in the United Kingdom. This section reports on the data sources defined for this exercise.

Agroforestry tree density and row spacing. A key variable in the literature-reported agroforestry systems is tree density; this effectively scales all associated elements of the system (e.g. sequestration potential, crop yield impacts, costs, timber production, etc.). We chose to use four in-field row spacings of 10, 20, 30 and 40m. In addition, we assessed one additional treatment assuming tree planting along fencelines, using data from Carey et al. (2008) to estimate fenceline length per hectare.

***Agroforestry system type and duration*. **The reviewed literature suggested deciduous trees would be most suitable for implementation in an agroforestry setting. Based on a choice of growth data for deciduous species (Matthews et al., 2016), data relating to sycamore was selected as most appropriate. This data is also representative of species such as birch and cherry; these species have good apical dominance and light canopy shade, both advantageous traits in agroforestry systems (R. Sykes, pers. comm.). Optimal yield duration was variable, so system durations of 50, 60, 70 and 80 years were assessed to ensure this variable did not bias results.

Below-ground CO_2 sequestration potential. Estimates of below-ground sequestration potential vary considerably (Table #AGF.2), One of the major underlying causes of this variation appears to be differences in tree density. Per-tree estimates of below-ground CO_2 sequestration were derived from values reported by Aertsens et al. (2013), and standardised to tree densities for different treatments.

Biomass CO_2 sequestration potential. Estimates of biomass sequestration by trees following the SAB (sycamore, ash and birch) growth path were extracted from the Forestry Commission's Carbon Lookup Tables (West & Matthews, 2012). These values were standardised for tree density per hectare to reflect agroforestry inter- and intra-row spacings using data from Matthews et al. (2016). Agroforestry inter-row spacings were set at 10, 20, 30, and 40m; agroforestry intra-row spacings were set at the equivalent spacing at age of maximum mean annual increment (MAI) from the yield tables (Matthews et al., 2016).

Tree planting and maintenance costs. Costs associated with implementing and maintaining the agroforestry system were extracted from Burgess et al. (2003). These costs were either given per tree or per unit planted area; where systems differed in this respect the costs were scaled to reflect this.

Tree timber yield and timber sale revenue. Tree timber yield was calculated according to reported yields from Matthews et al. (2016), scaled to reflect agroforestry tree densities. A variety of yield classes are reported for the sycamore growth path; we took the median reported yield class of 8 (before tree density scaling) as the most likely scenario. Timber yield revenue was calculated according to the price-curve equation defined by Whiteman et al. (1991); the model output was scaled to reflect inflation and changes in the timber price indices since its parameterisation.

Crop area and yield impacts. The impact on crop area was calculated based on the in-field row spacings of 10, 20, 30 and 40m, and an assumed 2m alley below each tree row (Burgess et al., 2003). Impacts to crop yield in the planted area resulting from the presence of trees in the cropping system were estimated based on the raw data reported by Burgess et al. (2005);

this was scaled to reflect the differing row width treatments according to scaling factors derived from data reported by Chirko et al. (1996). Application rates of agrochemicals and pesticides to the crop were also adjusted to reflect reduced crop area.

Bottom-up assessment of agroforestry marginal abatement cost: Modelling approach

Given the wide uncertainties associated with aspects of the measure cost and abatement rates, a Monte Carlo simulated-based uncertainty modelling approach was used to ensure that the estimate was not biased by unreasonable assumptions. The following were considered as a source of uncertainty in the modelled marginal abatement cost:

- **Below-ground C sequestration.** Minimum and maximum values for the ranges reported in Aertsens et al. (2013) were utilised.
- **Biomass C sequestration and timber yield.** Uncertainty in the yield classes reported in Matthews et al. (2016) was assumed, and this directly related to uncertainty in biomass C sequestration reported by West & Matthews (2012).
- **Planting and maintenance costs.** Burgess et al. (2003) report a range of values for most cost categories; these ranges were used as stochastic variables in the simulation.
- **Timber revenue.** Variation in timber yield per tree and per hectare (resulting from uncertainty in yield class) was incorporated into the Whiteman et al. (1991) price curve equation. In addition, the variability in timber price index in the period 1991—2017 was used to derive a stochastic scaling factor for data derived from the price curve, producing a randomly-placed estimate of the relative value of the timber in a fluctuating market.
- **Crop yield impacts.** Variability in the yield data reported by Burgess et al. (2005) was used to scale the estimate of the crop yield impact induced by the agroforestry system.

All financial data were converted to GBP 2017 for parity. The agroforestry system was assumed to be applied in a UK wheat system with a grain yield of 8.4 tonnes ha^{-1} and a gross margin of £796 ha^{-1}. All one-off costs and revenues were annualised and discounted using a discount rate of 3.5%. A Monte Carlo simulation (10,000 samples, Mersenne seed = 2605) was run to estimate the impact of the defined uncertainties on the marginal abatement cost effectiveness of the specified agroforestry systems.

#AGF.8. Bottom-up assessment of agroforestry marginal abatement cost: Results

The results of the Monte Carlo simulation identified marginal abatement costs of between £38—152 tonne CO_2-eq^{-1} for the specified systems (range is across 95% C. I. for all systems). Abatement rate was highly variable depending on system type, with mean estimates from 0.33 tonnes CO_2-eq ha^{-1} $year^{-1}$ (fenceline planting) to 11.09 tonnes CO_2-eq ha^{-1} $year^{-1}$ (10m rows). The cheapest system to implement was fenceline planting, although low tree density meant that this system also had the lowest per hectare abatement rate. Row planting was most cost-effective at 40m spacings, although the difference between different row spacings was relatively slight. The full ranges calculated for the different implementations are presented in Fig.2.

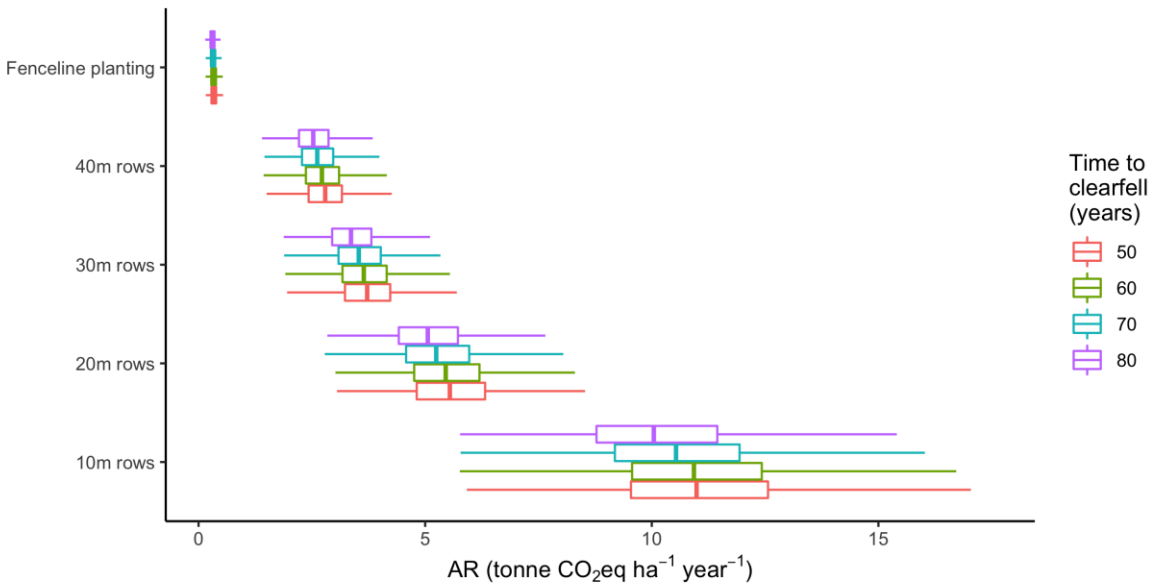

Fig.2. Estimated per-hectare annual GHG abatement rates for the agroforestry systems described in sections #AGF.6—#AGF.7. Abatement is annualised over the duration of the system (50—80 years).

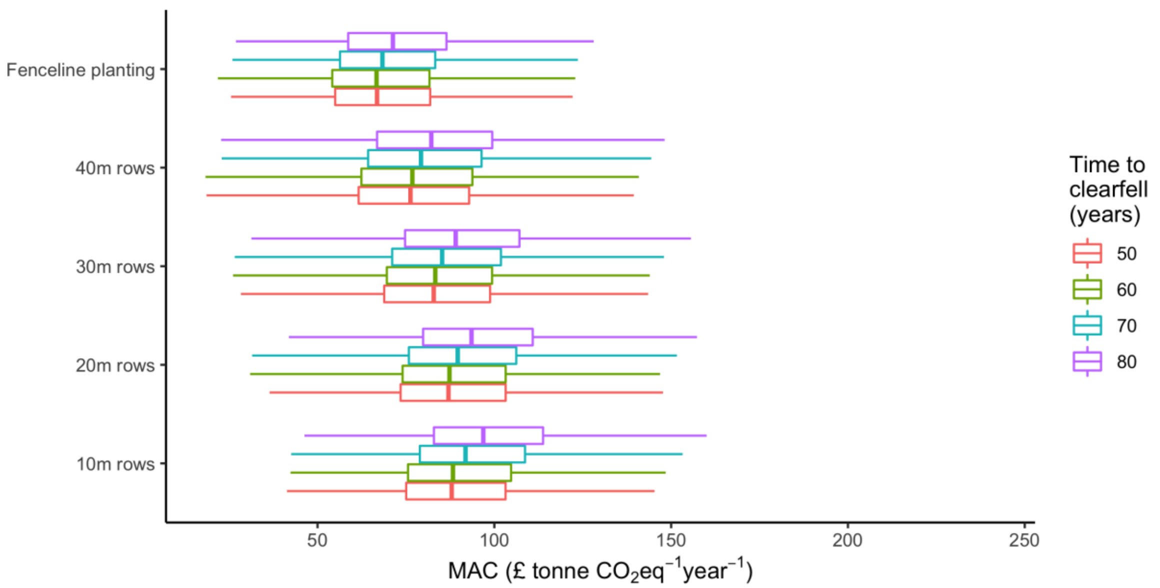

Fig.2. Calculated marginal abatement costs (in discounted 2017£) for agroforestry systems described in sections #AGF.6—#AGF.7.

While the greatest overall sequestration was achieved by the longer-duration systems, the highest annual rate was achieved over the shorter duration. Owing to this, and to the discounting of timber revenue (DR = 3.5%), the most cost-effective duration for all systems was 50 years. However, the 60-year duration was very similar for all systems, indicating that this time period strikes a balance between maximum abatement from biological systems and economic efficacy. Table 5. summarises the abatement cost effectiveness for 50-year-duration systems.

Table 5. Abatement cost effectiveness for 50-year-duration agroforestry systems.

System type	Marginal abatement cost (2017£ tonne CO_2-eq^{-1})			
	Mean	**Std. Dev.**	**2.5% C. I.**	**95.5% C. I.**
10m rows	91	21	56	140
20m rows	90	23	54	141
30m rows	85	23	48	138
40m rows	79	24	40	132
Fenceline planting	70	21	38	117

Crop yield was most influenced by the narrower row spacings; for the 10m system, average crop yield reduction was 25%. This decreased disproportionately with wider spacings, to a minimum of 4% for the 40m system. No yield impact was assumed for the fenceline planting system.

Aside from variations in estimated abatement rate, the cost of labour (best estimate = 15.28 £ ha^{-1}, range = 7.39—16.80 £ ha^{-1}) was an important driver of variation in cost effectiveness for the agroforestry system. Establishment and maintenance of the trees is quite labour intensive by comparison to the wheat crop, meaning the cost effectiveness is strongly affected by the cost of labour. The uncertainty in this variable largely stemmed from uncertainty as to whether the work would be carried out by the owner-occupier (the more expensive scenario) or contractors charging lower rates. Given the long-term nature of the investment in timber, the discount rate (3.5%) was also a key assumption in determining the cost effectiveness of the system. Lower discount rates improve the CE of the measure, while higher rates render it more expensive.

Eory et al. (2015) assumed uptake rates of 0.1%, 1%, and 10% for agroforestry on arable land in the United Kingdom, and the same uptake rates are assessed here. These are similar to the 'ambition levels' assessed for agroforestry by Thomson et al. (2018), which were low (~0%), medium (5%) and high (10%). In order to upscale the calculated rates for row-based systems, we use an of croppable area from Defra (2018) equal to 6,203,000 ha (Table #AGF.6). For fenceline agroforestry, the estimated total fenceline length as reported Carey et al. (2008) is used to scale the estimate of abatement potential. The 50-year duration systems, being the most cost-effective in each case, are utilised.

Table 6. Estimates of total abatement potential for 50-year duration agroforestry systems.

System type	Total abatement potential (kt CO2-eq year^{-1})		
	0.1%	**1%**	**10%**
10m rows	46.4—93.8	463.6—938.2	4,636.2—9,382.2
20m rows	23.5—47.0	234.6—470.2	2,345.9—4,702.4
30m rows	15.7—31.7	156.7—317.3	1,567.4—3,172.9
40m rows	11.7—23.6	116.7—236.5	1,167.2—2,365.0
Fenceline planting	0.3—0.7	3.2—7.2	32.1—71.6

Ancillary impacts of agroforestry systems

A number of additional agroecosystem and management impacts may result from integration of trees into the arable or pastoral systems:

Shelter effects: Trees in the cropping systems are often introduced specifically as windbreaks (Posthumus et al., 2015). In general, agroforestry systems will offer shelter to crops and soil from wind effects and erosion (Lasco et al., 2014).

Water storage and management: Agroforestry has been shown to enhance water use, storage and efficiency in arable and pastoral systems (Lasco et al., 2014).

Nitrogen fixation and offset of nutrient requirements: Substantial nitrogen fixation may be provided by leguminous tree species, with rates from 20 – 500 kg N ha^{-1} $year^{-1}$, depending largely on species, reported in the literature (Jose et al., 2004). Agroforestry-generated rates of up to 350 kg N ha^{-1} $year^{-1}$ have been reported in temperate US pastures (Sharrow, 1999). This additional soil nitrogen may offset crop fertilisation requirements. The benefits of this addition of fixed N may take time to realise, however, as many soils will require several years before concentrations are raised to the extent that plant growth is affected (Jose et al., 2004).

Control of pests and diseases: Agroforestry systems have potential for positive impacts on the spread of pests and diseases in cropping systems (Lasco et al., 2014).

Reclaiming degraded land: Agroforestry systems may facilitate the reclamation and rehabilitation of degraded land, e.g. mine sites (Nair & Garrity, 2012).

References

Aertsens, J., De Nocker, L. & Gobin, A. (2013) Valuing the carbon sequestration potential for European agriculture. *Land Use Policy* 31, pp. 584–594. Available at: http://dx.doi.org/10.1016/j.landusepol.2012.09.003.

Burgess, P., Incoll, L., Hart, B. & Beaton, A. (2003) The impact of silvoarable agroforestry with poplar on farm profitability and biological diversity. *Final Report to DEFRA. …*. Available at: http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:The+Impact+of+Silvoarable+Agroforestry+with+Poplar+on+Farm+Profitability+and+Biological+Diversity:+Final+Report+to+DEFRA#0.

Burgess, P.J., Incoll, L.D., Corry, D.T., Beaton, A. & Hart, B.J. (2005) Poplar (Populus spp) growth and crop yields in a silvoarable experiment at three lowland sites in England. *Agroforestry Systems* 63(2), pp. 157–169.

Burgess, P.J. & Rosati, A. (2018) Advances in European agroforestry: results from the AGFORWARD project. *Agroforestry Systems* 92(4), pp. 801–810. Available at: https://doi.org/10.1007/s10457-018-0261-3.

Carey, P.D., Wallis, S., Chamberlain, P.M., Cooper, A., Emmett, B.A., Maskell, L.C., McCann, T., Murphy, J., Norton, L.R., Reynolds, B., Scott, W.A., Simpson, I.C., Smart, S.M. & Ullyett, J.M.. (2008) Countryside Survey: UK Results from 2007. NERC/Centre for Ecology & Hydrology, 105pp. (CEH Project Number: C03259). , p. 105.

Chirko, C.P., Gold, M.A., Nguyen, P. V. & Jiang, J.P. (1996) Influence of direction and distance from trees on wheat yield and photosynthetic photon flux density (Q(p)) in a Paulownia and wheat intercropping system. *Forest Ecology and Management* 83(3), pp. 171–180.

Coulon, F., Dupraz, C., Liagre, F. & Pointereau, P. (2001) *Etude des pratiques agroforestieres associant des arbres fruitiers de haute tige a des cultures ou des patures. Solagro/Inra, Ministere de l'Amenagement et du Territoire et de l'Environnement*. Toulouse, France.

Cullotta, S., La Mantia, T. & Barbera, G. (1999) Descrizione e ruolo dei sistemi agroforestali in Sicilia. In: *econdo Con- gresso Nazionale di Selvicoltura, Venezia, Italy, 24–27 June 1999, Consulta Nazionale per le Foreste ed il Legno. Minis- tero delle Politiche Agricole e Forestali, Accademia Italiana di Scienze Forestali, EdAs*. Italy, pp. 429–438.

Defra (2018) Farming Statistics: Provisional crop areas, yields and livestock populations. (October), p. 23. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/747210/structure-jun2018prov-UK-11oct18.pdf.

Eichhorn, M.P., Paris, P., Herzog, F., Incoll, L.D., Liagre, F., Mantzanas, K., Mayus, M., Moreno, G., Papanastasis, V.P., Pilbeam, D.J., Pisanelli, A. & Dupraz, C. (2006) Silvoarable systems in Europe - Past, present and future prospects. *Agroforestry Systems* 67(1), pp. 29–50.

Eory, V., Macleod, M., Topp, C.F.E., Rees, R.M., Webb, J., McVittie, A., Wall, E., Borthwick, F., Watson, C., Waterhouse, A., Wiltshire, J., Bell, H., Moran, D. & Dewhurst, R. (2015) Review and update the UK Agriculture Marginal Abatement Cost Curve to assess the greenhouse gas abatement potential for the 5th carbon budget.

Fedrigo, J.K., Cruz, R.S., Benitez, V., V, C., Ferreira, G., Posse, J.P. & Vinoles, C. (2018) Dynamics of forage mass, air temperature and animal performance in a silvopastoral system of Uruguay. *Agroforestry Systems*.

Feliciano, D., Ledo, A., Hillier, J. & Nayak, D.R. (2018) Which agroforestry options give the greatest soil and above ground carbon benefits in different world regions? *Agriculture, Ecosystems and Environment* 254(December 2017), pp. 117–129. Available at: https://doi.org/10.1016/j.agee.2017.11.032.

Frelih-Larsen, A., MacLeod, M., Osterburg, B., Eory, A. V, Dooley, E., Katsch, S., Naumann, S., Rees, B., Tarsitano, D., Topp, K., Wolff, A., Metayer, N., Molnar, A., Povellato, A., Bochu, J.L., Lasorella, M. V & Longhitano, D. (2014) *Mainstreaming climate change into rural development policy post 2013*.

Glithero, N.J., Wilson, P. & Ramsden, S.J. (2013) Prospects for arable farm uptake of Short Rotation Coppice willow and miscanthus in England. *Applied Energy* 107(2013), pp. 209–218. Available at: http://dx.doi.org/10.1016/j.apenergy.2013.02.032.

Graves, A.R., Burgess, P.J., Palma, J.H.N., Herzog, F., Moreno, G., Bertomeu, M., Dupraz, C., Liagre, F., Keesman, K., van der Werf, W., de Nooy, A.K. & van den Briel, J.P. (2007) Development and application of bio-economic modelling to compare silvoarable, arable, and forestry systems in three European countries. *Ecological Engineering* 29(4), pp. 434–449.

den Herder, M., Moreno, G., Mosquera-Losada, R.M., Palma, J.H.N., Sidiropoulou, A., Santiago Freijanes, J.J., Crous-Duran, J., Paulo, J.A., Tomé, M., Pantera, A., Papanastasis, V.P., Mantzanas, K., Pachana, P., Papadopoulos, A., Plieninger, T. & Burgess, P.J. (2017) Current extent and stratification of agroforestry in the European Union. *Agriculture, Ecosystems and Environment* 241(April), pp. 121–132.

INE (2002) *Censo Agrario de Espana 1999. Instituto Nacional de Estadıstica*. Madrid.

Jose, S., Gillespie, A.R. & Pallardy, S.G. (2004) Interspecific interactions in temperate agroforestry. *Agroforestry Systems* 61, pp. 237–255. Available at: http://download.springer.com/static/pdf/900/art%253A10.1023%252FB%253AAGFO.0000029002.85273.9b.pdf?originUrl=http%3A%2F%2Flink.springer.com%2Farticle%2F10.1023%2FB%3AAGFO.0000029002.85273.9b&token2=exp=1460620245~acl=%2Fstatic%2Fpdf%2F900%2Fart%25253A10.1.

Lamerre, J., Schwarz, K.U., Langhof, M., von Wühlisch, G. & Greef, J.M. (2015) Productivity of poplar short rotation coppice in an alley-cropping agroforestry system. *Agroforestry*

Systems 89(5), pp. 933–942.

Lasco, R.D., Delfino, R.J.P., Catacutan, D.C., Simelton, E.S. & Wilson, D.M. (2014) Climate risk adaptation by smallholder farmers: The roles of trees and agroforestry. *Current Opinion in Environmental Sustainability* 6(1), pp. 83–88. Available at: http://dx.doi.org/10.1016/j.cosust.2013.11.013.

Lin, C., McGraw, R., George, M. & Garret, H. (1998) Shade effects on forage crops with potential in temperate agroforestry practices. *Agroforestry Systems* 44, pp. 109–119. Available at: http://www.springerlink.com/index/w5g7358081407w17.pdf.

Lorenz, K. & Lal, R. (2014) Soil organic carbon sequestration in agroforestry systems. A review. *Agronomy for Sustainable Development* 34(2), pp. 443–454.

Martineau, H., Wiltshire, J., Webb, J., Hart, K., Keenleyside, C., Baldock, D., Bell, H. & Watterson, J. (2017) Effective performance of tools for climate action policy - meta - review of Common Agricultural Policy (CAP) mainstreaming. (340202), p. 287.

Matthews, R.W., Jenkins, T.A.R., Mackie, E.D. & Dick, E.C. (2016) Forest Yield: A handbook on forest growth and yield tables for British forestry. *Forestry Commission, Edinburgh*, p. 92.

Mbow, C., Van Noordwijk, M., Luedeling, E., Neufeldt, H., Minang, P.A. & Kowero, G. (2014) Agroforestry solutions to address food security and climate change challenges in Africa. *Current Opinion in Environmental Sustainability* 6(1), pp. 61–67. Available at: http://dx.doi.org/10.1016/j.cosust.2013.10.014.

Mitchell, C.P., Stevens, E.A. & Watters, M.P. (1999) Short-rotation forestry - Operations, productivity and costs based on experience gained in the UK. *Forest Ecology and Management* 121(1–2), pp. 123–136.

Nair, P.K.R. & Garrity, D. (2012) *Agroforestry - The Future of Global Land Use*.

Nair, P.K.R., Nair, V.D., Mohan Kumar, B. & Showalter, J.M. (2010) Carbon sequestration in agroforestry systems. *Advances in Agronomy* 108(C), pp. 237–307.

Nichols, A.R., Kendall, D.A. & Iles, D.R. (2000) The agronomic and environmental implications of a combined food and energy system. *Aspects of Applied Biology* 58, pp. 363–372.

Nissen, T.M., Midmore, D.J. & Cabrera, M.L. (1999) Aboveground and belowground competition between intercropped cabbage and young Eucalyptus torelliana. *Agroforestry Systems* 46(1), pp. 83–93.

Palma, J., Graves, A.R., Burgess, P.J., van der Werf, W. & Herzog, F. (2007) Integrating environmental and economic performance to assess modern silvoarable agroforestry in Europe. *Ecological Economics* 63(4), pp. 759–767.

Pellerin, S., Bamière, L., Angers, D., Béline, F., Benoît, M., Butault, J.P., Chenu, C., Colnenne-David, C., De Cara, S., Delame, N., Doreau, M., Dupraz, P., Faverdin, P., Garcia-Launay, F., Hassouna, M., Hénault, C., Jeuffroy, M.H., Klumpp, K., Metay, A., Moran, D. & Pardon, L. (2013) How can French agriculture contribute to reducing greenhouse gas emissions? Synopsis of the study report. (July), p. 92.

Posthumus, H., Deeks, L.K., Rickson, R.J. & Quinton, J.N. (2015) Costs and benefits of erosion control measures in the UK. *Soil Use and Management* 31(September), pp. 16–33.

Schultz, A.M., Papanastasis, V.P., Katelman, T., Tsiouvaras, C., Kandrelis, S. & Nastis, A. (1987) *Agroforestry in Greece. Laboratory of Range Science, Department of Wildlife and Range Science, Aristotle University*. Thessaloniki, Greece.

Sharrow, S.H. (1999) Silvopastoralism: Competition and facilitation between trees, livestock, and improved grass-clover pastures on temperate rainfed lands. In: *Buck L.E., Las-soie J. and Fernandez E.C.M. (eds), Agroforestry in Sustainable Agricultural Systems.* pp. 111–130.

Thomson, A., Misselbrook, T., Moxley, J., Buys, G., Evans, C., Malcolm, H., Whitaker, J., Mcnamara, N. & Reinsch, S. (2018) *Quantifying the impact of future land use scenarios to 2050 and beyond - Final Report for the Committee on Climate Change*. CEH, Penicuik, Scotland.

Torralba, M., Fagerholm, N., Burgess, P.J., Moreno, G. & Plieninger, T. (2016) Do European agroforestry systems enhance biodiversity and ecosystem services? A meta-analysis. *Agriculture, Ecosystems and Environment* 230, pp. 150–161. Available at: http://dx.doi.org/10.1016/j.agee.2016.06.002.

Upson, M.A. & Burgess, P.J. (2013) Soil organic carbon and root distribution in a temperate arable agroforestry system. *Plant and Soil* 373(1–2), pp. 43–58.

West, V. & Matthews, R. (2012) *Carbon Look up table*. Forestry Commission.

Whiteman, A., Insley, H. & Watt, G. (1991) Price-size curves for broadleaves. *Forestry Commission Occasional Paper*, p. 32.

Wolbert-Haverkamp, M. & Musshoff, O. (2014) Is short rotation coppice economically interesting? An application to Germany. *Agroforestry Systems* 88(3), pp. 413–426.

Xavier, D.F., da Silva Lédo, F.J., de Campos Paciullo, D.S., Urquiaga, S., Alves, B.J.R. & Boddey, R.M. (2014) Nitrogen cycling in a Brachiaria-based silvopastoral system in the Atlantic forest region of Minas Gerais, Brazil. *Nutrient Cycling in Agroecosystems* 99(1), pp. 45–62.