5

Timber valuation

Introduction

In this chapter we assess both the social and private value of timber production. This is the major market-priced output of woodland. Furthermore, while recent policy statements from both the National Assembly for Wales (1999, 2001a; Forestry Commission (FC), 2001a,b) and UK government departments (Department for the Environment, Transport and the Regions (DETR), 2000) emphasise the need to adopt a holistic approach to managing woodlands, explicitly recognising their multipurpose nature, timber production remains, nevertheless, a key element of such a strategy, playing an important role in rural economies (FC, 1998).[1]

The economic and policy imperative to include timber production within any cost-benefit analysis of land use change involving forestry is therefore clear. However, the estimated value of this production depends crucially upon the real price of timber. Because plantation returns are long delayed, any (even small) change in real prices will have a major impact upon net present value (NPV) sums.[2] In order to assess this, the chapter opens with a brief history of commercial forestry in the UK designed to acquaint the reader with the recent, major and trend-breaking increase in domestic timber supply. In the subsequent section both the supply and demand sides of the UK market are modelled so that a balanced view on future prices can be derived. These conclusions are reinforced by time-series analyses of price movements.

[1] Note that, while this document explicitly refers to English woodlands, the recent Cabinet Office report to the Prime Minister (Cabinet Office, 2000) makes it clear that this is the first of three strategy documents.

[2] NPV is the sum of discounted net benefits (benefits minus costs) over the lifetime of a project (here a plantation). For further discussion, see Price (1987b); also, see Reed and Haight (1996) who introduce stochastic elements. In practice, felling and management decisions may be highly complex. This was recognised even in the classic optimal rotation model proposed by Faustmann (1849) (see Chang, 1998; Deegen, 2000). However, this decision becomes even more complex when forest-owners are motivated by objectives other than profit maximisation (see, for example, the discussion of owner's amenity benefits by Tahvonen, 1999; or of recreational hunting by Akabua *et al*., 2000).

Whilst timber value is clearly important, private planting decisions are often determined by the availability of shorter-term grants rather than long-delayed felling benefits, so we devote a section to reviewing the various subsidy schemes available. The next section brings together the preceding discussions regarding prices and grants and information on plantation costs and tree growth to produce the base rotation[3] models upon which our timber valuations are calculated.

The long time horizons inherent in woodland investments bring us to the vexed question of discounting. We discuss the principle of discounting and provide a brief review of the literature regarding the 'correct' discount rate with respect to both social cost-benefit analysis and private investment appraisal. We conclude that as no single, clearly correct discount rate can be identified, a sensitivity analysis approach is required.

The subsequent section provides investment appraisal results from the viewpoint of a private individual (the farmer) and this is then extended to provide a limited social cost-benefit analysis of the timber product of a plantation (i.e. ignoring those externalities dealt with elsewhere in this research). In both cases, NPV and annuity equivalent (defined subsequently) results are reported, the former being the usual fare of the forest economist while the latter are comparable with competing agricultural outputs.

Assessment of all possible woodland tree species was not feasible given both time constraints and a lack of data concerning less popular species. Furthermore, preliminary analysis indicated that costs and benefits of different conifers would be reasonably similar,[4] the same being (broadly) true of broadleaves. Therefore, two 'indicator' species were selected for analysis: Sitka spruce (conifer), and beech (broadleaf).

Historical background

Pre-1945

In terms of land use, British forestry has always been the poor cousin of agriculture. Although the prehistoric 'natural' condition of the land was primarily as forest, the influence of man has consistently been to clear-fell and convert land to agricultural use. Even by the time of the Domesday Book only 15 per cent of England remained under trees.[5] This deforestation trend continued for most of the last millennium with particularly heavy losses occurring in the sixteenth and seventeenth centuries when adoption of advanced husbandry techniques and subsequent enclosure of common land allowed agriculture to confine forestry to marginal areas and private

[3] A rotation is the full lifespan of a plantation from planting to felling.
[4] This is of course a relative statement. Differences do exist and are important at the micro level. However, the magnitudes of costs and benefits are similar enough for this to be a defensible assumption in this study.
[5] Pers. comm. Colin Price, Department of Agricultural and Forest Sciences, University of Wales, Bangor.

parklands, the latter often being operated on a non-commercial basis for private amenity values (Rackham, 1976). By 1900 only 4 per cent of England, 5 per cent of Wales[6] and 2 per cent of Scotland and Ireland was under forestry, these being by far the lowest levels in Europe (Rackham, 1976; National Assembly for Wales, 2001a).

At the start of the twentieth century the UK was almost completely dependent upon imports for its timber supply. This strategic weakness was exposed by the German naval blockade of Britain during the First World War. With timber a major input to the UK's vital coal industry it was felt that the creation of a strategic domestic timber supply was essential to the future security of the country and, in 1919, the Forestry Commission was established. Although strategic security of supply was the FC's initial objective this was quickly supplemented by further aims such as the commercial production of timber, the stimulation of employment in areas of rural depopulation and the provision of public benefits such as open-access recreation and wildlife habitats.[7]

Public sector forestry in the UK has from the outset followed an erratic course. A strong initial political will to establish a secure national timber supply ensured that the 1920s were a period of major afforestation, reversing the trend (if not the effects) of the previous millennia. However, as memories of wartime shortages receded and world timber prices slumped, the 1930s saw planting figures fall well behind the 30,000 ha annual target envisaged at the creation of the FC. This slump was offset to some extent by the Commission's own promotion of forestry as a response to rural depopulation trends and a government initiative 'to create a settled force of woodsmen and their families whose livelihood would be enhanced from their own tenanted smallholdings' (Philip, 1976). Nevertheless, the 1930s still saw an overall contraction of new planting.

Post-1945

Figure 5.1 illustrates total, FC and private sector annual planting from 1945 to 2000, providing a starting point for our discussions of the development of both public and private woodland during this period.

Public sector forestry

The end of the Second World War marked the start of the most sustained period of UK forestry expansion in recorded history (see Figure 5.1). Initially, national

[6] An overview of the history of Welsh forestry from prehistoric times to the present is given in National Assembly for Wales (2001a).

[7] In recent years the FC has also defended its existence as a source of import savings and reduction in agricultural subsidy. Pearce (1991) shows the import substitution argument to be invalid.

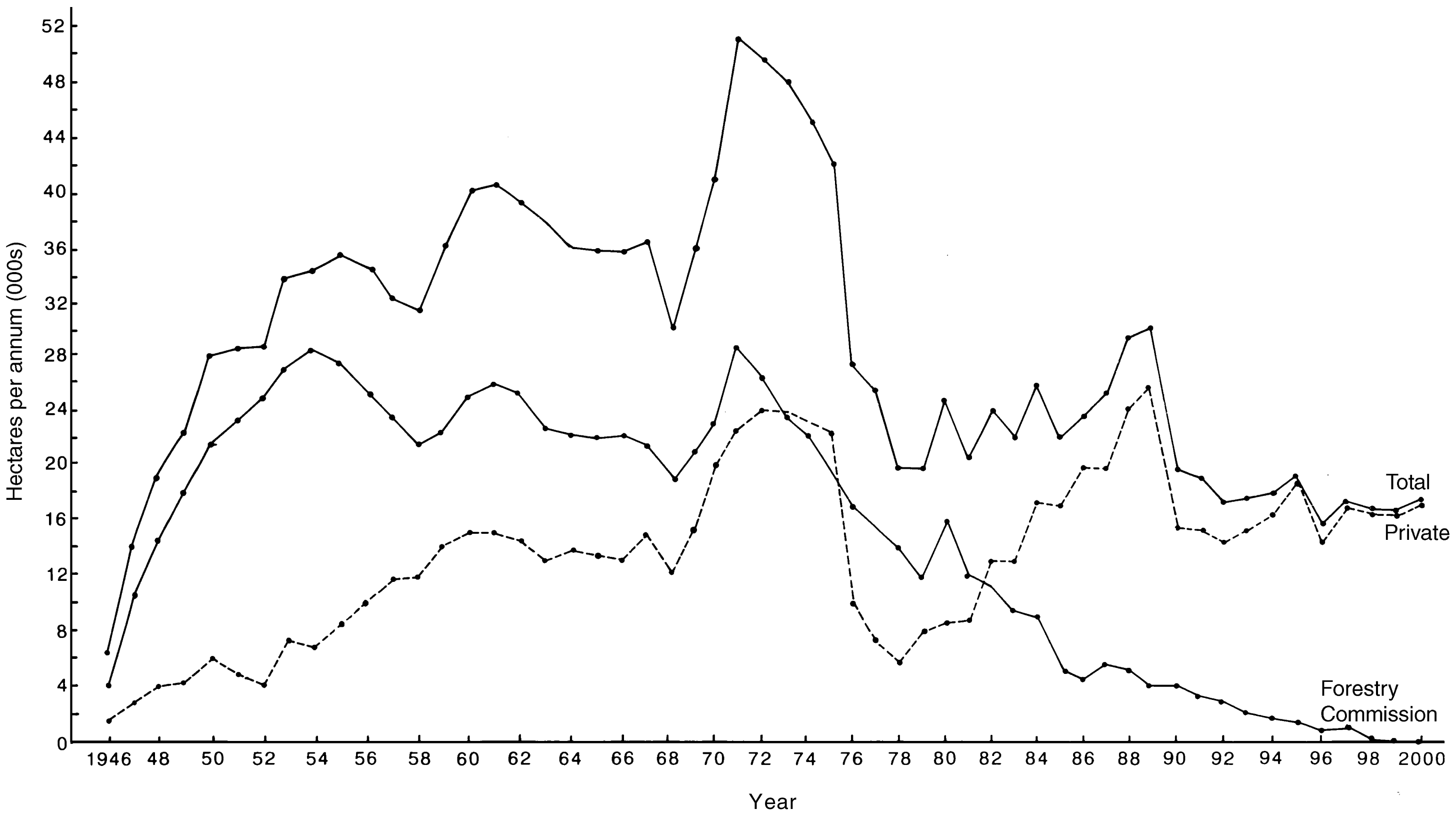

Figure 5.1. Forestry Commission, private sector and total annual forestry planting, Great Britain 1946–2000. (*Source:* Forestry Commission, (1979, 1985b, 1988a, 1989, 1990, 1993, 1994a, 1997, 2001c.)

security concerns and high prices again dominated policy objectives. The post-war adoption of a planned approach to the economy, firm prices and the expansion of the world timber trade ensured that FC planting accelerated to a peak of over 28,000 ha per annum in the decade following the war. The period from the mid 1950s to the early 1970s was characterised by fairly stable public sector planting of about 24,000 ha annually. This was helped by a government decision to allow the FC to operate at a favourably low rate of return compared to other state investments. A discount rate of only 3 per cent[8] was required of the Commission compared to rates of between 5 per cent and 10 per cent for other state-owned enterprises.[9]

The year 1971 marked a significant peak for the FC with plantings exceeding 28,000 ha for the first time since the early 1950s. However, that year also marked a turning point beginning a downward trend in FC planting which has continued for three decades to the present day. The 1970s were a difficult period for the UK economy with the oil crisis and domestic economic problems (in particular relatively high inflation and poor trade balances) leading to heavily depressed growth rates. This put pressure on all areas of public finance, to which the FC was not immune. Contractions in FC employment (Thompson, 1990) accompanied reductions in planting and by 1979 annual planting had dropped to 11,800 ha, i.e. about 40 per cent of the 1971 level.

The election in 1979 of a Conservative government, pledged to the reduction of the public sector in favour of private enterprise, meant that the decline in new planting seen in the 1970s has been extended throughout the 1980s and up to the present day, a trend which was unaffected by the election of a Labour government in 1997. New planting in Wales ceased in 1993, followed by England in 1996 and Scotland in 2000, this latter year being the first since the Second World War in which the FC had not undertaken any new planting.

Arguably, a more serious threat to the absolute scale of FC operations is the disposal of its estate into private ownership, a trend which, as Table 5.1 shows, can be traced back to the early 1980s when an extensive programme of land sales was implemented.[10] Prior to this, the overall size of the estate had grown every year since its creation to stand at 1,264,000 ha in 1981 but this has declined in every year since to stand at 1,052,900 ha in 2000, a reduction of 16.7 per cent.

[8] Even lower rates of return were required from plantings carried out in Northern Ireland. From 1989 the FC was set a target rate of return of 6 per cent but, as this is virtually unattainable without explicit valuation of non-market benefits, the Treasury initially allowed new investment decisions to be taken at a 3 per cent rate with the resultant shortfall being written off as Forestry Subsidy (H.M. Treasury, 1991: Annex G). Felling decisions remain at a 5 per cent discount rate to retain compatibility with existing FC appraisal systems.

[9] From 1989 this has been set at 8 per cent for commercial public sector enterprises, with a discretionary rate of 6 per cent applied to projects with returns accruing to the public sector. This latter rate applies to forestry management decisions, with the exception of Forestry Enterprise which is permitted to use a 3 per cent discount rate as an explicit subsidy (Pearce and Ulph, 1998).

[10] See statement by the then Secretary of State reproduced in Appendix V of FC (1985a).

Table 5.1. *Forestry Commission holdings: Great Britain 1978–2000 ('000 ha)*

Year-end	FC plantation		Awaiting planting etc.[1]	Total forest	Total FC estate[2]	Annual change in total FC estate[3]
1978	862.5		83.4	945.9	1,253.2	—
1979	868.2		84.0	952.2	1,256.3	3.1
1980	884.0		78.4	962.4	1,263.4	7.1
1981	895.7		70.2	965.9	1,264.0	0.6
1982	905.5		59.4	954.9	1,258.7	(5.3)
1983	908.7		54.0	962.7	1,250.9	(7.8)
1984	901.7		47.6	949.3	1,209.2	(41.7)
1985	900.5		34.3	934.8	1,181.0	(28.2)
1986	897.5		30.2	927.7	1,165.5	(15.5)
1987	899.7		23.4	926.4	1,156.4	(9.1)
1988	898.5		20.6	919.1	1,149.4	(7.0)
1989	898.2		17.2	915.4	1,144.2	(5.2)
	Productive land	Other woodland[4]				
1990	863.5	34.3	11.2	909.0	1,139.5	(4.7)
1991	858.5	34.5	9.8	902.8	1,133.1	(6.4)
1992	855.3	34.8	5.6	895.8	1,127.5	(5.6)
1993	845.4	37.1	5.1	887.6	1,115.4	(12.1)
1994	826.6	44.0	3.2	873.8	1,099.5	(15.9)
1995	815.6	45.4	2.0	862.9	1,089.8	(9.8)
1996	804.6	46.7	0.8	852.0	1,080.0	(9.8)
1997	795.4	48.3	0.4	844.1	1,073.0	(7.0)
1998	784.7	49.7	0.7	835.0	1,061.8	(11.2)
1999	779.5	50.7	0.6	830.8	1,057.3	(4.5)
2000	774.2	51.1	0.6	825.9	1,052.9	(4.4)

Notes: [1] Between 1979 and 1984 this figure was disaggregated into 'land awaiting planting' and 'scrubland, etc.' with the latter being about 10 per cent of the former during that period.
[2] Total forest + other non-woodland (nursery land + agricultural land + unplantable + forestry workers' holdings).
[3] Numbers in brackets indicate reductions in the FC estate.
[4] Recreational land, etc.
Source: Forestry Commission (1979, 1985b, 1989, 1990, 1993, 1994a, 1996, 1997, 2001c).

Despite numerous ministerial pronouncements on safeguarding public access to land sold by the FC, in almost all cases privatisation has led to the exclusion of the public (Goodwin, 1995).[11] This is particularly serious given that it has been in areas of high population where the proportion of FC woodlands privatised has been

[11] In the period from October 1991 to November 1995, of 35,233 ha privatised only 506 ha (1.4 per cent) had freedom of access guaranteed (Goodwin, 1995).

highest (Lean, 1996).[12] Despite recent claims that the present government has 'put a stop to large-scale sales of forest land by the Forestry Commission' (Cabinet Office, 2000: p. 90), disposals have continued under the present Labour administration although arguably at a lower rate than under the previous Conservative government. Given the poor record of ensuring access to private woodlands, this does appear to run contrary to official policy initiatives to promote rural recreation and access as set out in the recent Rural White Paper (DETR, 2000) and suggests a potential failure of 'joined-up government' in this area. However, this trend of loss of open-access land in terms of a reduced FC estate is slightly ameliorated by the increased area of 'other woodland' within the estate, the majority of which is used for recreational purposes. Nevertheless, these gains are more than offset (at least in quantitative terms) by the scale of disposals, suggesting that this remains a cause for concern.

Given this background, recent policy documents promoting the growth of multipurpose woodlands (FC, 1998) can perhaps best be interpreted as indicating continued support for private sector expansion (discussed subsequently) and a reorientation of public sector forestry away from conifer monocultures and towards mixed and broadleaved woodland. This seems most clearly to be the case in our case study area of Wales. In its most substantial woodland policy document since devolution in 1999, the National Assembly for Wales (2001a: p. 45) recently stated that:

> Since substantial areas of coniferous forest will be harvested during the next 30 years, there will be an opportunity to reshape these woodlands to deliver wider benefits to society. The National Assembly's estate can play a leading role in this process since it is made up of over 80% coniferous species, compared to only half in private woodlands.

The same document continues by providing an insight into the National Assembly's definition of the multipurpose woodland they seek to promote over their chosen policy horizon of fifty years: '. . . multi-purpose woodlands managed for recreation, landscape and wildlife as well as for timber production . . . absorbing carbon dioxide and so helping ameliorate climate change' (p. 45).

Such a definition fits well with the cost-benefit analysis conducted in this volume, and this suggests that the results reported in subsequent chapters may have particular resonance within the current policy environment both across Britain and, particularly, within our study area of Wales.

Finally, while expansion of the FC has *de facto* halted, other publicly funded forests are in process of being established, most notably the National Forest currently being developed under the auspices of the Countryside Commission in central England. First proposed in 1987 and defined as a series of woodlands comprising

[12] For example, between 1981 and 1996, 91 per cent of FC woodlands in West Yorkshire were privatised; 72 per cent in Durham; 67 per cent in Kent; 53 per cent in Humberside and 43 per cent in Essex (Lean, 1996). However, one countervailing trend has been the growth of charity-funded woodlands (although these are not always open-access) such as those operated by the Woodland Trust (Smith, 1996).

some 30 million trees planted over an area of about 200 square miles (516 km^2), the National Forest is intended to bring economic and quality-of-life benefits to a relatively depressed area (Countryside Commission, 1987, 1993; Cloke *et al.*, 1996). However, examination of the National Assembly for Wales' (2001a) *Strategy for Trees and Woodlands* shows that there are no explicit plans to develop similar projects either within or near to our study area of Wales.

Private sector forestry

From the outset, direct government intervention through the agency of a state forestry service has been complemented by the stimulation of a private forestry sector through the provision of tax relief and other incentives to private individuals who invest in timber production.[13]

Despite these incentives, inexperience meant that initial private sector involvement was very restrained. However, from the late 1950s a proliferation of firms specialising in facilitating private forestry investments considerably eased the practical problems of such investment. These companies located land, arranged purchases, planting and felling, and took care of the tax liability and refunding formalities, thus allowing those for whom tax relief was an attractive proposition to become forest-owners without ever having to visit a plantation or see a tree.

In this way, post-war planting of private woodlands expanded at a steady rate from 1945 to the early 1970s (see Figure 5.1). However, as with the FC, the 1970s were a period of relative decline for the private forestry sector. As the OPEC oil-shock sent the world economy into recession, so the UK's forest-owning elite no longer had the excess taxable income to divert into forest tax-havens. However, these were just the people who benefited from the private sector boom of the 1980s and by 1989 the planting of private woodlands was at its highest ever level. In the search for cheap afforestable land[14] many sites of great ecological value were destroyed (Royal Society for the Protection of Birds, 1987). This factor, and a national outcry against such tax avoidance,[15] caused the government to act and withdraw such tax relief with effect from late 1989 (UK Parliament, 1988).

The removal of tax relief had an immediate impact upon private sector planting, which almost halved from 1989 to 1990. The reason it did not fall further was primarily the existence of a system of planting and maintenance subsidies (discussed subsequently) designed to appeal to land-owners and, to a lesser extent, farmers, rather than to those in search of tax havens. These appear to have generated a

[13] Details of these tax relief schemes are given in Bateman (1992).
[14] Unlike most other planting costs, land purchase was not tax-deductible. This led investors to plant on cheap, but often highly unsuitable, wetland areas, destroying valuable natural habitats to produce very poor but highly tax-deductible plantations (Royal Society for the Protection of Birds, 1987).
[15] Culminating in a disparaging *Observer* front-page magazine feature on the hundred largest forest-owners in Britain (Lean and Rosie, 1988). See also *The Times* (1988) and Bloom (1988).

reasonably steady expansion in British private woodlands of just over 15,000 ha per annum throughout the period 1990–2000.[16] However, unlike Forestry Commission operations, much private woodland development falls outside the scope of policy influence, making objectives such as the promotion of multipurpose woodland more difficult to achieve (Selman, 1997).

Historical background: summary

In forestry terms the UK has only recently expanded its domestic supply. Although this grew rapidly in the post-war period, new planting by the FC is now at a total standstill, superseded by private planting at a relatively constant (if, in national terms, low) rate. However, current government policy argues that a holistic assessment of the multipurpose nature of woodland suggests a strong case for further expansion (FC, 1998; DETR, 2000). Certainly, compared to its continental neighbours, the UK lags behind in terms of its forest resource. After eighty years of expansion, less than 11 per cent of the land area of Great Britain is under woodland while about 77 per cent is under agriculture.[17] This compares with EU averages of 25 per cent and 60 per cent respectively.[18] However, this disparity of itself does not constitute a valid case for continued expansion of UK domestic timber supplies. In order to assess this we need first to consider long-term market conditions, and it is to this issue that we now turn.

The UK timber market and long-term prices

Softwoods

The UK's consumption of wood products far outstrips its domestic production, the resultant shortfall being met through timber imports. Indeed, wood products are consistently within the top five import items by value. Much of the empirical work presented in this volume concerns the early 1990s, a period when the UK consumed roughly 45 million m^3 of wood products annually,[19] at a cost of £6.3 billion, of which approximately 83 per cent was softwood products (Forestry Industry Committee of Great Britain (FICGB), 1992; FC, 2001c). The past decade has seen an overall modest increase in demand to about 47 million m^3 in 1999 at an import cost of about £6.7 billion (United Nations Development Programme (UNDP) *et al*., 2000; FC, 2001c). With both global and domestic demand for lumber forecast to increase by 20–40 per cent by 2010 (Brown, 1999; Matthews and Hammond, 1999) and

[16] Of this about two-thirds was concentrated in Scotland.
[17] Authors' calculations based upon FICGB (1992), UNDP *et al*. (2000) and FC (2001c).
[18] *Ibid*.
[19] Measured in wood raw material equivalent (WRME).

potentially double from present levels by the middle of the twenty-first century (FICGB, 1992; Watson *et al.*, 1998), some commentators have forecast increases in future real prices for timber.[20]

With respect to softwood prices, we see two major flaws in this argument. First, the present level of UK production represents only the early stages of an ongoing substantial expansion of domestic supply engendered by the sustained high levels of planting in the inter-war years and the period from the late 1940s to the 1970s. This is set to continue, with production reaching an estimated peak of nearly 20 million m^3 by the early 2020s and then tailing off (as a result of the curtailing of Forestry Commission expansion since the 1970s) to a plateau of about 12 million m^3 by the 2050s. Second, and more importantly, this expansion of domestic supply has been echoed by an increase in the availability of softwood import supplies (UNDP *et al.*, 2000).[21] World coniferous roundwood production rose from 1,096 million m^3 in 1971 to a peak of 1,307 million m^3 in 1986, slipping back only slightly to a level of 1,295 million m^3 in 1991 (Whiteman, 1995)[22] since when the area of softwood felled in most developed countries has generally been exceeded by the area replanted (UNDP *et al.*, 2000). When combined with arguments regarding ongoing technical change,[23] these factors seem to suggest that real prices for softwood are unlikely to increase in the foreseeable future. Indeed, heavy felling by some Baltic nations during the late 1990s resulted in a fall in UK real timber prices for softwoods over the course of the decade (Forest Enterprise, 2001).

A number of commentators have examined the issue of whether real timber prices have changed significantly over time, the majority concluding in favour of constant real prices (Doran, 1979; Price and Dale, 1982; Pearce and Markandya, undated; Bateman and Mellor, 1990; Bateman, 1996; UNDP *et al.*, 2000). In an in-depth analysis, Whiteman (1995) undertook a time-series analysis of real softwood prices from 1870 to 1989. His best-fitting time-series model for this period indicated stable real prices (excluding shocks) prior to the Second World War, a shift to a higher level during the war and a continuation at a higher, but again constant (excluding shocks), level after the war. Whiteman's best estimate was therefore for a constant real softwood price for the foreseeable future. Our own analysis of a shorter time-series from the Second World War up to the early 1990s also supports an assumption of constant real prices, with a single shock to the system during the commodity price boom of the 1970s (Bateman and Mellor, 1990).

[20] This argument is reinforced by concerns regarding acid-rain damage to forests (Ewers *et al.*, 1986; Bergen *et al.*, 1992; Pearce, 1993; FC, 1994b). However, estimates indicate that this is unlikely to have any significant impact upon timber supply and consequent prices (Bateman, 1996).

[21] This trend is exemplified by the case of Sweden where, since the 1930s, timber growth has consistently outstripped cutting (Wibe, 1992).

[22] These measurements are in underbark volumes.

[23] Two forms of technical change can be identified: (i) improved plantation husbandry; (ii) increased availability of timber substitutes (particularly in the construction industry; see Leigh and Randell, 1981).

Table 5.2. *High forest by general species: Forestry Commission and private woodland in Great Britain 1947–2000 ('000 ha)*

Forest type	1947	1965	1980	1994	2000
Mainly coniferous high forest	397	922	1,317	1,516	1,584
Mainly broadleaved high forest	380	352	564	615	837
Total	777	1,274	1,881	2,131	2,421

Source: Figures for 1947, 1965 and 1980 are from the occasional Census of Woodlands (FC, 1987; reproduced in Pearce, 1993). Figures for 1994 are from FC (1994a) and include some extrapolation from the 1980 Census. Figures for 2000 are from FC (2001c).

Extending this analysis to the present day (by incorporating data up to March 2001 from Forestry Enterprise, 2001) suggests that the slump in real prices observed in the late 1990s is not statistically significant over this longer period (although of course it would eventually become so if it were sustained for a sufficiently long time).

In conclusion, the consensus view fails to support the hypothesis of future increasing real prices for UK softwoods and we therefore adopt an assumption of constant real prices in the subsequent analysis (although we do note the possibility of unforeseen shocks challenging such an assumption).

Hardwoods

While global reserves of coniferous forest have been reasonably stable or have even grown over the past two decades, the post-war era has seen some decline in temperate hardwoods and a dramatic fall in tropical hardwoods. Considering first the British case, the twentieth century saw a continuation of a centuries-old decline in the area of ancient broadleaf woodlands. In England and Wales this stood at just 142,000 ha in 1933 yet had more than halved by the mid 1980s (Nature Conservancy Council (NCC), 1984). The bulk of this loss arose from conversions to mainly conifer plantations, with the remaining losses generally attributable to agricultural encroachment (NCC, 1984; Council for the Protection of Rural England (CPRE), 1992). However, the planting of new broadleaved woodlands has meant that, since the 1960s, the overall area of broadleaved high forest has consistently risen, as shown in Table 5.2.

While newly planted broadleaved woodland does not have the ecological value of ancient woodland, it does represent an encouraging trend. However, as in the case of softwoods, the UK is far from self-sufficient in hardwoods. The present level of UK domestic hardwood (round and sawn) consumption is about 2 million m^3 per annum. This exceeds domestic production, which fell over the past decade from 1.2 million m^3 in 1991 to 0.8 million m^3 in 1999 as a result of low planting and high felling early in the century (FICGB, 1992; FC, 2001c). While this represents a much higher self-sufficiency rate than for softwoods, and production is forecast to rise to 1 million m^3 per annum in 2001 and remain at that level for at least twenty years (FC, 2001c), nevertheless the UK is highly import-dependent and consequently subject to fluctuations in the world market.

Global stocks of hardwoods have fallen dramatically in the post-war period, primarily as a result of deforestation in the developing, tropical countries of the world in which such trees are predominant. The causes of this deforestation are complex and interlinked and include increasing consumption pressures from both the developed and the developing world (Whiteman, 1995; Global Environment Facility, 1998; World Bank, 1999),[24] population and poverty pressures in the developing world (World Resources Institute, 1994; United Nations Population Division (UNPD), 1998), sustained growth in demand for fuel-wood, to the point where it is currently estimated that half of all global wood consumption is as fuel (UNDP *et al.*, 2000), and forest burning for agricultural expansion and other reasons (Myers, 1990; Elvidge *et al.*, 1999; World Commission on Forests and Sustainable Development, 1999).

The total loss of global forests to date is uncertain but may be as high as 50 per cent (Bryant *et al.*, 1997). What is more certain is that annual net hardwood extraction rates rose from about 0.8 per cent at the end of the 1970s (Doran, 1979) to 1.8 per cent a decade later (Myers, 1990) but have fallen back slightly over the course of the 1990s (Food and Agriculture Organization (FAO), 1997; UNDP *et al.*, 2000). Current losses are the subject of considerable controversy but probably exceed 130,000 km^2 annually (FAO, 1997; Matthews *et al.*, 2000; Tucker and Townsend, 2000; UNDP *et al.*, 2000), a rate which means that by 2010 only Brazil and the Democratic Republic of Congo will have any significant remaining areas of rainforest and both of these will be under unsustainable long-term pressure. Given that the rainforests represent the richest global environment for biodiversity (Davis *et al.*, 1994; Olson and Dinerstein, 1998), the potential exists for species extinction on a scale unprecedented in human history (MacNeill, 1990; Pearce and Warford, 1993; World Resources Institute, 1994; Oldfield *et al.*, 1998; UNDP *et al.*, 2000).

Setting aside the terrible ecological consequences of this destruction, the unsustainable nature of current global hardwood extraction has been seen by some as likely to lead to increases in associated real prices. Indeed, published estimates of such increases can be found, ranging from a credible 0.5 per cent to what we regard

[24] While the majority of population growth is in the developing world, a citizen of the developed world consumes up to ten times the ecosystem goods and services of a citizen in the developing world (Global Environment Facility, 1998).

as an unfeasibly high 4 per cent annually.[25] However, this is balanced by opposing views such as that of Whiteman (1995), who argues that while consumption may increase, 'it should be possible to improve forest management to meet these demands, which would then keep timber prices relatively stable'. Certainly the rate of growth in British hardwood planting detailed above should mean that, providing there is not an unforeseen sharp rise in domestic demand, the current rate of UK self-sufficiency may improve, giving something of a domestic buffer against future reductions in global supplies.

This is an area of uncertainty, disagreement and relatively little in-depth research. While we feel that there is a considerably stronger case for real price increases in hardwoods than softwoods, any such rise is likely to be some way off. Furthermore, as our wider study examines the potential for conversions out of existing agricultural land use and into forestry we prefer to adopt conservative assumptions with regard to changes in the future value of both land uses. Adopting any positive rate of real price increase for hardwoods would translate into substantial increases in projected timber values. Instead we prefer to adopt the zero real price rise assumption of Whiteman (1995) and accept that any transpiring price increases will improve the potential for land use conversion to forestry. Such an assumption also allows us readily to revise our calculations in the light of subsequent improved information.

We now turn our attention away from prices towards the other major source of timber-based revenues: grants and subsidies.

Grants

Given the long-delayed nature of forestry returns, government incentives have always played a major role in UK private sector planting decisions. The earliest incentives coincided with the establishment of the Forestry Commission when, in 1919, a scrub clearance and ground preparation grant was introduced. A second planting grant scheme, introduced in 1927, established an enduring trend for broadleaves to be given preferential subsidy rates over conifers, reflecting an early recognition of non-strategic/production objectives within forestry policy.

Following the Second World War a variety of Forestry Commission administered schemes were introduced. Examination of these reveals a gradual movement in forestry policy objectives from simply maximising timber production to initiatives giving equal emphasis to timber, environmental and recreational goals (Johnson and Nicholls, 1991; Bateman, 1996; Winter, 1996; MacFarlane, 2000; DETR, 2000; FC, 1998, 2001b). However, until the late 1980s the overriding force behind the expansion of private sector forestry was tax concessions. The scrapping of most

[25] Estimates are from Johnston *et al*. (1967), Doran (1979), Burnham (1985) and Hart (1987).

of these concessions in the 1988 Budget (Lynch, 1989) thrust the role of grants centre-stage as the main means of state support for forestry in the UK, a role which has persisted to the present day.

The majority of woodland grants are administered by the Forestry Commission although other funding bodies are also important and both are considered below. The review presented in this section focuses primarily upon the rates of grant in operation during the period of the early 1990s for which our empirical model operates. However, in comparison with the major changes introduced at the end of the 1980s, the basic structure of this grant aid has varied relatively little over the past decade although grant amounts have generally increased (Winter, 1996; MacFarlane, 2000). A number of recent policy initiatives have stressed plans for expanding forestry both in Britain as a whole (FC, 1998; DETR, 2000) and in our case study area of Wales (National Assembly for Wales, 1999, 2001a; FC, 2001a,b). However, these documents have not put forward policies for an overhaul of the grant-aiding system but rather suggest that levels of aid may be increased in the future (although no specific announcements have been made to date). Therefore, the structure of the model presented here for the early 1990s remains valid today and for the foreseeable future although rates of grant are already dated as they are continually under review.

Forestry Commission administered grants

Throughout the 1980s the FC emphasised its reorientation away from the simple pursuit of timber output and towards wider objectives (FC, 1985c). Such policy was embodied in the introduction, in 1988, of the Woodland Grant Scheme (FC, 1988b) which, alongside the stimulation of timber production and rural employment, explicitly set out to enhance landscape, create wildlife habitat, provide longer-term recreation and sporting facilities and encourage the conservation and regeneration of existing woodlands. Rates of support under the Woodland Grant Scheme (WGS) were revised in 1990 as listed in Table 5.3.

Payments under the WGS were made in three instalments: 70 per cent at planting, 20 per cent after five years and 10 per cent after a further five years (subject to satisfactory establishment). In addition to this a Better Land Supplement (BLS) was payable for planting on arable/improved grassland cultivated (including ploughing) within the previous ten years. BLS was £400/ha for conifers or £600/ha for broadleaves, all payable at planting.

Further enhancement of this package was provided in 1992 by the introduction of the Woodland Management Grant (WMG). This provided an annual addition to the WGS, payable after the first ten years of establishment in return for the setting down and execution of five-yearly management plans designed to

Table 5.3. *Woodland Grant Scheme payments (£/ha)*

Area planted	Conifers	Broadleaves
0.25–0.9 ha	1,005	1,575
1.0–2.9 ha	880	1,375
3.0–9.9 ha	795	1,175
10 ha+	615	975

Source: Johnson and Nicholls (1991).

Table 5.4. *Woodland Management Grants*

Type of WMG	Period of eligibility (age of wood in years)	Rate of grant (£/ha per annum)
Standard: conifer[1]	11–20	10
Standard: broadleaf[1]	11–40	25
Special[1,2]	11 onwards	35
Supplement for small woods[3]		
Standard: conifer	11–20	5
Standard: broadleaf	11–40	10
Special[4]	11 onwards	10

Notes: [1] All these grants are also payable as *additions* where the owner is a farmer under the Farm Woodland Scheme, as compensation for agricultural output forgone (but not for establishment costs).
[2] Higher rates are available for woodlands of special environmental value (nature conservation, landscape or public recreation). The owner will be expected to maintain the wood's character. These grants are available for any forest older than ten years. However, they may be extended to younger or even proposed forest if the Forestry Commission is satisfied that there is demand for such a provision.
[3] Available as additions for all woodlands of less than 10 ha (of correct age).
[4] Available for any woodland (over ten years) of less than 10 ha where the woodland is of special environmental value.
Source: Johnson and Nicholls (1991).

increase the environmental value of the woodlands concerned. Table 5.4 details WMG payments.

The year 1991 also saw the FC introduce the Community Woodland Supplement (CWS), a further addition to the WGS (and WMG) designed to promote recreational woodlands 'within 5 miles of the edge of a town or city and in an area where the opportunities for woodland recreation are limited' (FC, 1991). In implementation this has been interpreted very broadly so that relatively small communities of just a few thousand people are considered sufficient to justify payment of CWS. At its introduction the scheme consisted of a single payment of £950/ha payable at planting. All woodlands qualifying for CWS were allowed WGS and WMG, the latter being paid at the enhanced 'special' rate.

In addition to the above, from 1992 the FC offered a single £100 flat rate payment for each new woodland (irrespective of size) conditional on the drawing up of a management plan (FC, 1991).[26]

Other grant schemes

In 1988 the then Ministry of Agriculture, Fisheries and Food (MAFF) introduced the Farm Woodland Scheme (FWS) to provide annual income support to farmers who establish woodlands on what was previously agricultural land (MAFF, 1987a).[27] The scheme had almost identical objectives to the FC's WGS (and was payable concurrently) with the additional goal of reducing surplus agricultural production. As a consequence, higher rates of FWS were payable on better quality land. Although these rates did not distinguish between conifer and broadleaf woodlands, the period of annual support was longer for the latter.

Poor uptake of the FWS led to its replacement in 1992 by the Farm Woodland Premium Scheme (FWPS) (MAFF, 1992a,b,c). Here farms first applied to the FC for planting grants under the WGS (including BLS, WMG, CWS and the single new woodland payments where appropriate). If approved the farm could then apply to MAFF for FWPS payments as shown in Table 5.5.

For woodlands with less than 50 per cent broadleaves the FWPS is payable in each of the first ten years after planting, a period which is extended to fifteen years for mainly broadleaved woodlands.[28] However, grant repayments with interest are stipulated if land is returned to agriculture within twenty years for the former or thirty years for the latter (MAFF, 1992b).[29]

With respect to our Welsh study area, the creation of the Cambrian Mountains and Lleyn Peninsula Environmentally Sensitive Areas (ESAs) in 1986 and 1987, respectively, seemed to offer the possibility of further grants for broadleaved woodland.[30]

[26] In addition to this the FC also provides certain other grant payments for general and coppice management, open spaces and grey squirrel control. Details are given in Johnson and Nicholls (1991).

[27] The FWS also pays planting grants but, since its revision in 1992, these have been identical to those offered under the WGS. Farmers may not collect both FWS and WGS planting grants.

[28] This is considerably more front-loaded than the original FWS which provided lower annual sums but over a longer period.

[29] Farms may also convert land into forestry under the Common Agricultural Policy (CAP) set-aside scheme. Set-aside woodland is not eligible for either FWPS or WGS BLS payments. Standard WGS payments may be received concurrently with set-aside in high productivity areas but as this does not apply for most of our study area we do not pursue this particular permutation any further.

[30] Further grants towards the costs of promoting landscape or countryside conservation are occasionally paid by the Countryside Commission and Nature Conservancy Council while the Agricultural Development Advisory Service (ADAS) can provide certain technical support. However, the occasional nature of such support means that it is not considered further in this study.

Table 5.5. *Payments under the Farm Woodland Premium Scheme*[1] *(£/ha per annum)*

Present use	Lowlands	Disadvantaged Area	Severely Disadvantaged Area
Arable/improved grassland	250	190	130
Unimproved	n/a	60	60

Notes: n/a = not available.
[1] The following FWPS restrictions apply: (i) not more than 50 per cent of farm eligible; (ii) not more than 40 ha of unimproved land per farm; (iii) eligibility for arable/improved grassland restricted to land under such usage within the previous three years; (iv) the FWPS as a whole is cash rather than area limited. Further details are given in MAFF (1992b).
Source: MAFF (1992b).

The Welsh Office (1989a) stressed the importance of such features within the Cambrian Mountains ESA while in a subsequent leaflet (Welsh Office, 1989b) payments of £45/ha per annum became available for management of such woodlands. These are clearly specified as additions to existing planting and management grants. However, subsequent publications regarding the Lleyn Peninsula ESA offered lower rates of grant (£15/ha per annum) restricted to existing broadleaf woodland alone (Welsh Office, 1992a,b). Conversations with both ESA authorities in the late 1990s indicated that those anomalies still persisted.

A policy and planning drive away from purely timber-orientated, monoculture conifer plantations was signalled in 1988 when EC Directive 85/337 was implemented via Environmental Assessment (Afforestation) Regulations which made applicants for FC assistance submit an environmental assessment of the proposed forest.[31] In the same year the Department of the Environment, Transport and the Regions (DETR), the ultimate national planning authority, indicated that planning permission for such large conifer plantations would not normally be granted for sites in England. As noted previously, this policy has since been reinforced in policy documents applying to both England (Forestry Commission, 1998) and, more recently, our study area of Wales, with the National Assembly for Wales explicitly promoting the concept of diverse, multipurpose woodland as opposed to conifer monocultures (National Assembly for Wales, 2001a).

[31] In practice such assessments became routine requirements for plantations of over 100 ha affecting National Nature Reserves, National Parks, Sites of Special Scientific Interest, etc.

While the policy environment appears to be favourable to further woodland expansions, the planning system may place a brake on this, particularly in the case of farm woodlands. Lloyd *et al*. (1995) report study findings suggesting that some farmers believe that conversions to woodland may be irreversible,[32] a similar result being reported by Crabtree *et al*. (1998). Indeed, our own informal contacts with the Forestry Commission indicated that felling licences would only be granted on condition that affected areas would be replanted.[33] While future policy may change, this means in effect that once farmers place land under trees they may well become legally bound to maintain an equal area of woodland on the farm in perpetuity. This irreversibility of land use may well slow the expansion of farm woodlands and we consider the potential inertia of farmers in reacting to purely financial pressures in the cost-benefit analysis presented at the end of this volume.

Grants: conclusions

Farmers considering diverting land into forestry are eligible for a variety of grants and subsidies. These vary considerably according to which scheme they register under and according to locational factors. In order to allow for this the timber valuation model developed subsequently allows flexibility across the full gamut of grant/subsidy opportunities existing in the early 1990s. Results for these various permutations were identified using the following coding system:

S = subsidy rate is (as follows)
I = rate for planting on improved grassland or arable land
U = rate for planting on unimproved land
nda = rate for planting in a non-disadvantaged area
da = rate for planting in a disadvantaged area
sda = rate for planting in a severely disadvantaged area
+CW = rate for planting, given Community Woodland supplement
−CW = rate for planting, not given Community Woodland supplement

In the following section we incorporate these subsidies within the wider costs and revenues arising from plantation management.

Plantation costs and revenues

Choice of species

Ideally one would wish to analyse all those species which are likely to be used in a conversion from agriculture to forestry. The feasibility of such an analysis

[32] See also Williams *et al*. (1994) which gives further details regarding this study.
[33] This may also adversely affect land prices.

was investigated with the FC's Forestry Investment Appraisal Programme (FIAP). However, while this is an excellent tool for the management of given stands, it was not amenable to the type of modification required to answer the questions posed by this research. Consequently it was decided that two representative species, one conifer and one broadleaf, would be chosen for study.

Among the eight major species of conifer grown commercially in the UK,[34] the Sitka spruce stands out as by far the most dominant, constituting 28 per cent of total forest area, more than double that of any other species (FICGB, 1992). Sitka spruce is capable of producing an average annual yield in excess of 24 m^3/ha over an optimal rotation, with typical UK productivity averaging 12–16 m^3/ha. The rapid growth rate means that optimal felling ages can be very short, from sixty years on poor ground to as little as forty-five years on good sites.[35] The choice of Sitka spruce as a representative conifer therefore reflects a logical and often observed timber-productivity decision. However, this species is not thought to be optimal in terms of recreation value.

Interestingly there is little empirical evidence regarding a connection between tree species and recreation value. In one of the few valuation studies to consider this, Hanley and Ruffell (1992) failed to identify a significant relationship. This may mean that all woodland recreation valuation studies are observing values for outdoor, rather than specifically woodland, activities. However, if we temporarily lurch from the empirical to the anecdotal, it is the authors' firm belief that walkers *do* recognise and appreciate the difference between the claustrophobic atmosphere produced by a species like Sitka spruce (with its dense entanglement of lower branches, tightly packed together to maximise timber yield, set in a bed of stultifying acid pine needles) and, say, the much more airy and open feel of a Scots pine woodland. An even clearer difference is evident when we then consider the gorgeous spaciousness, beautiful trunks and foliage, and verdant undergrowth of an oak or beech woodland.

To allow for this difference we decided to extend our appraisal to consider a representative hardwood. Here the choice was more difficult as the oak is the most abundant broadleaf species but is relatively slow-growing and less productive than the beech, which we selected for study as a more viable hardwood alternative.[36]

34 In descending order of total forest area, major conifers are: Sitka spruce (28%); Scots pine (13%); lodgepole pine (7%); Japanese larch (6%); Norway spruce (6%); Corsican pine (2%); Douglas fir (2%); European larch (2%). Major hardwoods are: oak (9%); ash (4%); beech (4%); birch (4%). The remaining area consists of a range of species (FICGB, 1992).

35 As discussed subsequently, optimal felling age is a function in part of discount rate rather than just of growing conditions.

36 There are some ecological arguments in favour of the oak over the beech as the latter creates less understorey and has a less penetrable canopy. However, data availability favoured the beech, which is, despite these drawbacks, ecologically strongly preferable to the Sitka spruce.

Sitka spruce costs and revenues

Costs

Irrespective of species, the majority of plantation costs occur at the start of the rotation (planting, etc.) and at felling. Here we make the common FC assumption that all cutting costs (both thinnings – the extraction of undersized trees at set points during the rotation so as to maximise long-run plantation yield – and felling) are either carried out by contractors or incur contractor-level implicit costs upon the plantation operator. This allows us to use the standing timber price–size curve discussed subsequently.

Estimates for these and other costs (maintenance, fertiliser, fencing, etc.) were obtained from the FC and are detailed in Bateman (1996). Costs may vary somewhat depending upon infrastructure, distance to sawmills, local variation in input supply prices (including labour), intensity of planting, etc. Typical values for such parameters were incorporated within the data supplied by the FC.[37]

Revenues

Four factors are key to the determination of timber revenues:

(i) the rate of growth
(ii) the price per m^3
(iii) the discount rate (the rate at which forest managers or policy-makers progressively reduce the present-day value of revenues that will be received further and further into the future)
(iv) available grants and subsidies.

We will address each of these factors in turn.

Growth rate is typically defined using the yield class (YC) measure, which is the maximum average annual increment in volume which a stand of trees can deliver. So, for example, a stand for which this value is 12 m^3/ha per annum is referred to as being YC12.

Since its inception in 1919 the Forestry Commission has collected data quantifying the characteristics of plantations growing in differing yield classes. These ‘yield models’ have been collated across varying species and management regimes (Edwards and Christie, 1981) and show how, for each yield class, tree volume increases over time. The yield models provide the basic data on tree growth used in our subsequent analysis.

[37] Note that costs were representative rather than being varied spatially. This is a potential weakness in the analysis, and other commentators such as Thompson *et al*. (1997) have used spatially sensitive costs. However, the latter study considers an area (British Columbia) forty-five times larger than Wales and uses just three cost-level zones. In effect, therefore, our own study performs well in comparison and we do feel it is reasonable to point out that, while revenues vary very substantially across Wales (as shown in Chapter 6), costs are less variable.

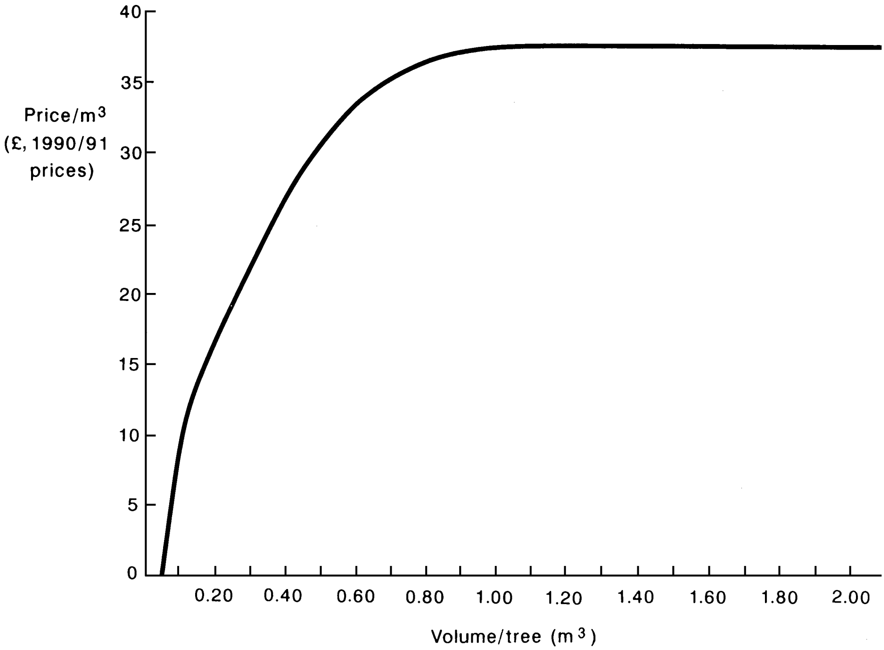

Figure 5.2. Price–size curve for conifers in England and Wales. (*Source:* Drawn from data given in Whiteman, 1990.)

From the perspective of the timber producer, price is far from constant and is instead a function of the mean volume per tree. Simply put, when trees are thin they are of limited use and so their price/m^3 is low. As trees increase in volume so their usefulness, and therefore price/m^3, rises. This continues (at a diminishing rate) to the point where a tree's girth is such that it can be used for sawn wood, telegraph poles and myriad other products. After this point the price/m^3 remains fairly constant and the value of a stand increases only as much as volume does.

Estimation of this 'price–size curve' has been the subject of repeated statistical investigation by the FC (Mitlin, 1987; Whiteman, 1990; Sinclair and Whiteman, 1992). In this study we adopted the findings of Whiteman (1990), primarily because this research uses the same base year as our wider study, but also because this analysis recognises that prices are higher in England and Wales than in Scotland and produces separate models which provide a substantially better fit to the data ($R^2 = 87.5\%$) than the unified analysis for the whole of Britain reported by Sinclair and Whiteman subsequently ($R^2 = 74.7\%$). Figure 5.2 illustrates the price–size curve used in our analysis.

As a stand of trees grows, so thinner trees (known appropriately as thinnings) are removed to help the remainder flourish. The thinning process typically starts about twenty years after planting and then occurs at regular intervals of, say, five years up to felling. The value of thinnings can be calculated via the price–size curve, although for obvious reasons this value is substantially less than that of the main crop. By relating the price–size curve to the timber volume information contained in the yield model we can calculate the timber revenue generated each year from planting to felling and compare this with per annum costs to obtain a value for annual timber net benefit.

The optimal felling age therefore emerges as a key factor in determining the overall value of a stand. As already mentioned this will vary according to the yield class concerned, for, as the FC yield models show (Edwards and Christie, 1981), the faster a tree grows the sooner it reaches its age of maximum annual average product. Therefore, as yield class increases, optimal felling age falls. Early felling is encouraged by the practice of discounting, which we now consider.

Discounting is the process by which revenues and costs occurring in the future are converted into present-day values. By this process, different projects that have returns occurring at different times can be compared on a common footing and investment decisions made. The general result is that costs and benefits arising in the future are not valued as highly as those which occur in the present day. This is for a variety of reasons including a simple preference for earlier rewards ('positive time preference') and the fact that money invested in a project, such as forestry, is no longer available for investment elsewhere, say in a bank, and so there is a lost opportunity, in terms of the interest forgone, which increases over time (the 'opportunity cost of capital'). The further into the future that costs and benefits occur, the more they are discounted. By looking at how this effect increases into the future we can observe the underlying 'discount rate' that is being employed.

The factors determining discount rates are complex and while we return to consider some of these in Chapter 7 we do not attempt to provide a full account of their determinants in this volume.[38] However, what is pertinent here is that changing the discount rate can dramatically alter the present value of a given investment. This is particularly true for forestry where (with the exception of felling expenditures) the bulk of costs occur early in a rotation while felling revenues can be long delayed. This impact can be demonstrated by examining the discount factor (DF) implied by each discount rate. The DF shows the proportion (from 1 to 0) by which future costs or benefits should be multiplied to obtain their equivalent present-day value. Figure 5.3 illustrates the DF for four discount rates (each of which has some relevance for our analysis and which we discuss subsequently) as time progresses from the present (year 0) into the future. As can be seen, for any discount rate, the further into the future that a cost or benefit occurs, the lower is the DF and consequently the

[38] Introductions are given in Pearce (1986), Hanley and Spash (1993) and Perman *et al*. (1999). Other useful sources include Lind (1982a) and Markandya and Pearce (1994).

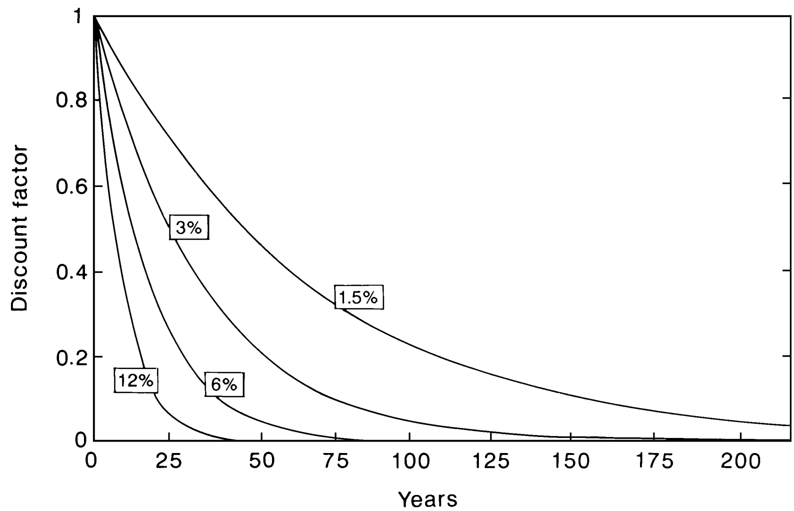

Figure 5.3. Discount factor curves.

lower is the present value of that cost or benefit. Furthermore, we can also see that changes in the discount rate imply very substantial alterations in the speed at which the DF declines. For example, for net benefits (benefits minus costs) occurring fifty years from planting we can see that at a 1.5 per cent discount rate we have DF = 0.5 (i.e. the present-day value is about half the net benefit received in year 50) whereas with a 12 per cent discount rate, DF is virtually zero (i.e. the present value of net benefits received fifty years hence is almost nil).

We discuss the choice of discount rate subsequently, but for now the important message is that discounting can very substantially affect the present-day value of long-term investments like forestry, and the higher the discount rate the lower that present-day value. As stated previously, this increases the impact of YC upon felling date. Just as a higher growth rate results in an earlier optimal felling date, so does a higher discount rate. Discounting makes it in the interest of forest managers to obtain timber-felling revenues earlier rather than later, to the extent that they trade off gains in timber volume against the discounting-induced reduction in the value of that delayed timber volume.

The impact of varying yield class and discount rate upon optimal felling age was calculated using the FIAP software mentioned previously. FIAP operates by maximising the net present value of a stand subject to several user-determined parameters. Results from this analysis are given in Table 5.6 which shows the extent to which felling age declines as both yield class and discount rate increase.

Application of the FIAP software indicated that it was insufficiently flexible to conduct our required analyses concerning variation of grant levels. Therefore, yield

Table 5.6. *Optimal felling age for various discount rates: Sitka spruce, YC6–24*

Discount rate (%)	Yield class 6	8	10	12	14	16	18	20	22	24
2	80	78	74	70	69	68	66	66	66	65
3	73	72	69	63	60	58	57	57	56	56
4	68	67	64	58	54	51	50	50	49	48
5	64	62	60	56	52	49	46	44	44	44
6	60	58	56	54	50	47	43	42	41	40
8	54	53	51	50	47	44	42	40	37	35
10	50	48	47	46	44	42	40	38	36	34
12	47	44	43	42	41	40	38	36	34	33

Notes: Optimal felling age (in years from planting) maximises NPV given the relevant discount rate (r) and yield class combination. The above figures treat the planting year as year 0. The table was calculated using FIAP running at the Forestry Commission headquarters at Edinburgh (except for the row for $r = 3\%$ which was interpolated). The authors are obliged to Jane Sinclair and Roger Oakes at Edinburgh for assistance.

The table uses the following FIAP settings: spacing = 2.00 × 2.00; thinning = line, MTT; delay on first thinning = none; stocking = 85%; successor crop NPV = 0; price–size curve = GB conifer 1992; thinning price differential (£ 1992/93) = 0.30/m^3; charge per m^3 (£ 1992/93) = 3.68/m^3.

models for YC6–24 Sitka spruce (from the tables given in Edwards and Christie, 1981) were transferred into a database for use in association with a statistical package (Minitab, 1994) allowing the authors to write macros for repetitive data analysis. In this manner, desired grant scenarios (discussed subsequently) could be specified and their net present values (NPV) calculated.[39] This model allows us to convert yield class estimates into NPV equivalents taking into account any desired grant scenario. Our yield class predictions and their estimated values are given in Chapter 6.

Beech costs and revenues

Costs

Information on hardwood planting costs is far less readily available than for conifers. Data were collected both from interviews with managers of broadleaf woodlands[40] and from certain published sources (Lewis, pers. comm., 1988; Hart, 1987 and pers. comm., 1990). However, the distributions of revenues and costs are similar to

[39] Examples of the year-by-year revenue and cost streams derived from such models are detailed in Bateman (1996).

[40] Notably Fred Lewis, Kerswell, Exminster and Cyril Hart, Chenies, Dean.

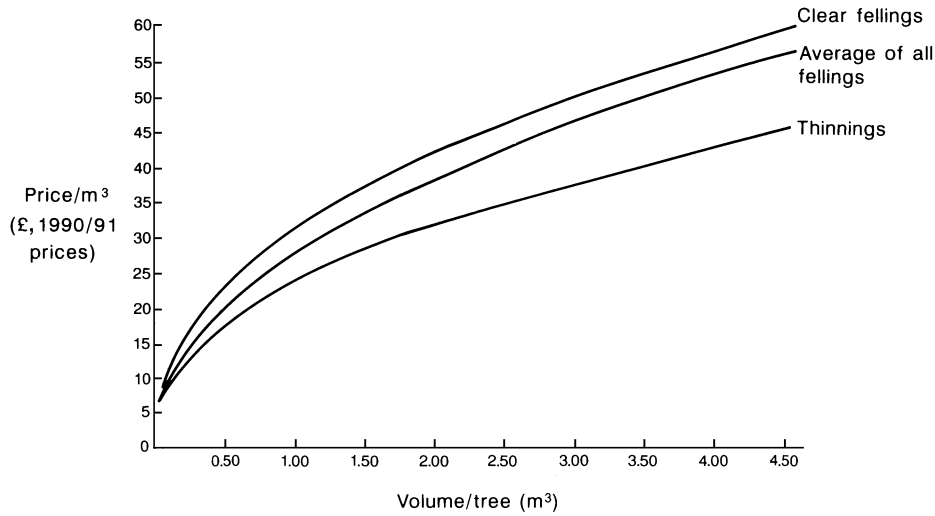

Figure 5.4. Price–size curves for beech in Great Britain. (*Source:* Drawn from data given in Whiteman *et al*., 1991.)

those for conifers, with high costs at planting and felling, mainly maintenance and thinning costs at other times, and revenues at felling and to a minor degree from thinnings (details are given in Bateman, 1996).

Revenues

The factors affecting broadleaf timber revenues are the same as those relevant to conifers, with price varying positively with tree volume. In their study of price–size curves for broadleaves, Whiteman *et al*. (1991) show that, because thinnings have relatively high extraction costs per m^3, standing prices for thinnings are on average 24 per cent below the price per m^3 paid for clear fell timber. Consequently, two price–size curves are estimated (with a third, average curve being reported for ease of generalised account rather than for individual plantation assessment). As hardwood timber values vary considerably among species, price–size curves are estimated for individual species (unlike the generalised conifer relationship), with those for beech being illustrated in Figure 5.4.

With timber volume and the price–size curve defined, we can calculate annual timber revenues, subtract costs per annum and derive the net timber benefit values for each year from planting to felling. As before, the optimal felling age declines as the yield class and discount rate increase. This relationship was analysed as described previously, results being given in Table 5.7.

Finally, hardwood yield class models and associated revenue and cost streams were linked via spreadsheet to a statistical analysis package to allow them to be integrated with available grants and subsidies. Net present values for any user-defined

Table 5.7. *Optimal felling age for various discount rates: beech, YC4–10*

	Yield class			
Discount rate (%)	4	6	8	10
2	125	120	119	118
3	105	99	95	93
4	91	85	80	78
5	81	75	71	69
6	75	69	65	62
8	65	59	56	53
10	58	52	48	47
12	53	47	43	42

Notes: Optimal felling age (in years from planting) maximises NPV given the relevant discount rate (r) and yield class combination. The above figures treat the planting year as year 0. The table was calculated using FIAP running at the Forestry Commission headquarters at Edinburgh (except for the row for $r = 3\%$ which was interpolated). The authors are obliged to Jane Sinclair and Roger Oakes at Edinburgh for assistance.

The table uses the following FIAP settings: spacing = 1.20 × 1.20; thinning = broadleaved, intermediate thin; delay on first thinning = none; stocking = 85%; successor crop NPV = 0; price–size curve = broadleaves for 1989/90 T.R.; thinning price differential (£ 1992/93) = 0.30/m^3; charge per m^3 (£ 1992/93) = 3.68/m^3.

yield class/discount rate/subsidy scenario could then be derived as detailed previously for the conifer models.

Discount rates

Any investment in forestry essentially trades off initial costs against delayed benefits. This is conventionally achieved by calculating the NPV of the investment via a discount rate (r) which is influenced by positive time preference, the opportunity cost of capital and other factors discussed subsequently. Now, this research sets out to examine two perspectives on whether agricultural land should be converted to forestry: that of the farmer; and that of society. However, there is good reason to suppose that these two will have differing discount rates.[41] Put at its simplest, if we consider time preference, farmers are mortal while society is, at very least, much

[41] For further discussion on the divergence of social from private discount rates see Baumol (1968), Goodin (1982), Sen (1982), Sagoff (1988), Pearce and Turner (1990), Markandya and Pearce (1994) and Pearce and Ulph (1998).

longer lived (we hope!). Therefore, society is likely to place relatively more weight on delayed returns than is an individual farmer. Accordingly we might expect society to have a lower rate of positive time preference. A similar result is obtained when we consider discounting based on the opportunity cost of capital. For a risk-averse society this should imply a relatively low social discount rate dictated by the rate of return on riskless investments (government bonds, etc.). However, for the private individual the opportunity cost of capital should be relatively high on account of the rates of return available from alternative investments.[42] Both arguments suggest that private (agricultural) discount rates might be higher than social discount rates.

In this section we examine evidence regarding agricultural and social real rates of discount. However, before turning to this we need to address one further complication, the comparability of agricultural and forestry investments. Farmers commonly make decisions on an annual cycle whereas the time horizon of a forester is usually a full rotation of a stand, which typically varies from a minimum of four decades for conifers to over a century for hardwoods. Comparison of annual gross margin with rotation NPV is therefore problematic. Two approaches exist. First, agricultural margins can be assessed and discounted over at least a rotation length. Second, woodland NPV can be converted to an annual equivalent, i.e. the constant annual return (or 'annuity') which, over the length of a rotation, would be valued equally with the standard NPV sum. After discussion with relevant experts[43] it was decided that the former option lacked credibility as farmers (who are the relevant decision-makers) are used to annual rather than rotational decision-making. Therefore, the calculated NPVs for all our yield models (using the relevant agricultural or social discount rate) were converted to annuity equivalents.[44]

Farmers' discount rates

Literature review

A priori we would expect that the relatively lower rates of return exhibited by the agricultural sector (compared to the industrial and commercial sectors) would result in somewhat lower real discount rates than those implied by the government's 8 per cent average rate of return required of public sector agencies selling commercially or the 6 per cent rate used for pure public good activities (H.M. Treasury, 1991; Pearce and Ulph, 1998).[45] However, little explicit work has been published

[42] This may be a less strong argument if re-investment is restricted to the agricultural sector where rates of return are historically low.
[43] Notably Colin Price and Rob Willis, University of Wales, Bangor.
[44] The conversion process and related formulae are discussed in Bateman (1996).
[45] The 8 per cent estimate is 'based on average returns on assets achieved in the private sector for activities with low cyclical year by year variability' (H.M. Treasury, 1991). In 2002 H. M. Treasury published consultations signalling a reduction of the pure public sector discount rate from 6 to 3.5 per cent.

in this area, with most commentators examining real rates of return or agricultural interest rates rather than discount rates *per se*.

The early work in this latter area is predominantly American, dating back to Melichar (1979) who proposed that real rates of return were determined by expected rents and actual and expected inflation rates. Feldstein (1980) modified this theory by suggesting that such a mechanism may ultimately be driven by inflation acting upon land prices, while Tanzi (1980) extended this by proposing a further link to the business cycle. However, in an empirical test of these theories, Alston (1986) failed to find a long-run link between inflation and land prices, and Burt (1986) rejected such complex models in favour of a simple long-run equilibrium land price approach which yields an estimate of the real rate of return of 4 per cent per annum.

Turning to the UK, similar results are reported by Cooper (1992) who uses a real interest rate approach, based on the work of Brase and La Due (1989), to estimate a mean value of 4.5 per cent for UK agriculture for the period 1964–90. While agricultural interest rates are highly variable,[46] such a result seems to be roughly echoed by lending practice during our study period. In correspondence with the authors, the National Westminster Agricultural Office (a major source of farm finance) quoted an average real agricultural interest rate of 4 per cent over base rates.[47]

A lower interest rate, averaging 2.44 per cent above base rate, is reported by Cunningham (1990) in a study of MAFF surveys, while MAFF itself employed an agricultural interest rate risk premium of 2.78 per cent above base rate during our study period.[48] However, there are several problems with extrapolating from interest rates to discount rates. First, if base rates change frequently, lags in the adjustment system may confound the analyst. Second, interest rates vary significantly across farms, projects and time.[49] Third, the link between interest rates and discount rates may be weak in that the former relate to returns on new investments rather than on total assets (which are likely to be lower).

In addressing this latter point, Harrison and Tranter (1989) analysed the period 1978/79 to 1986/87, reporting a mean real rate of return on all assets of 2.56 per cent.[50] Positive time preference would suggest that the real discount rate might be somewhat higher than this. Such an argument would support the findings

[46] Annual averages range from −13.01 per cent (1976) to +10.08 per cent (1990) in Cooper (1992).

[47] Pers. comm. Sue Train, National Westminster Agricultural Office (NWAO), and letters from Brian Montgomery, Senior Executive, NWAO, July 1993. However, this correspondence highlighted the variation in rates across farms and projects. For example, a range of real rates of 0–5 per cent was given for differing projects and times by Charles Morgan of Chris Grote Farms, Norfolk.

[48] Pers. comm. Douglas Cooper, MAFF, 1993.

[49] This point was made in correspondence with NWAO (see above) and Paul Hill (Wye College) who both stated that while interest rates were roughly 2 per cent above base rates for good risks, they could be much higher for risky investments.

[50] Sample extends across Great Britain. Rates are quite consistent, only ranging from 1.87 per cent to 3.90 per cent.

of Lloyd (1993) who uses a capital asset pricing model of agricultural land prices in England and Wales for the period 1946–89 to empirically derive a long-run real discount rate of 3.6 per cent.

These latter studies provide what we feel is the best evidence on agricultural real discount rates. However, none of these studies is specific to our Welsh study area and so our own rate of return analysis was undertaken.

Empirical work

Two studies of agricultural rates of return in Wales were undertaken: a short time-series analysis of the period 1987–92; and a cross-sectional study of the 1989/90 base year. In both cases data were provided by the Welsh division of the Farm Business Survey (FBS, 1988, 1989, 1990, 1991, 1992) which defines the nominal return as farm income expressed as a percentage of tenants' capital.[51]

Rates of return in Wales, 1987–92

Table 5.8 presents nominal rate of return ($\mathrm{RoR_n}$) statistics for various categories of farm identified during FBS surveys for the years 1987/88 to 1991/92. These categories are further subdivided by farm size.

Statistical analysis was undertaken for all farm categories except pig, poultry and cropping farms as these are minor activities in Wales and were not separately classified after 1989. This showed that specialist or mainly dairy farms achieved significantly higher $\mathrm{RoR_n}$ than did other farms. Subsequent analysis also isolated a quadratic relationship with size, measured in British stocking units (BSU), showing that $\mathrm{RoR_n}$ rose with size but at a diminishing marginal rate. $\mathrm{RoR_n}$ also fluctuated annually although only one year (1988/89) was found to be significantly different from all others. A variety of further variables taken from the FBS database (see discussion in Chapter 8) were also tested and found to be insignificant in predicting $\mathrm{RoR_n}$.

A model was constructed and tested across a variety of functional forms. Our best-fitting model is reported as Equation (5.1). Tests for interactions, multicollinearity, autocorrelation and heteroscedasticity failed to isolate any significant problems with this model.

$$\begin{aligned}\mathrm{RoR_n} = {} & -18.62 + 7.68\,\mathrm{DAIRY} + 9.57\,\mathrm{HIYEAR} + 1.13\,\mathrm{BSU_t} - 0.0105\,\mathrm{BSU_t^2} \\ & (-9.06) \quad (6.32) \qquad (6.53) \qquad (8.38) \qquad (-6.33)\end{aligned} \tag{5.1}$$

[51] The Farm Business Survey (FBS) was an arm of MAFF (operating in Wales under the auspices of the Welsh Office) which conducted annual surveys of a representative sample of farms throughout the country. The sample size averaged 734 farms per annum over our 1987–92 study period; however, many farms are retained in the sample for about three years. The number of distinct farms in the time series is 2,867.

Table 5.8. *Agricultural nominal rate of return (RoR) on tenants' capital: Wales 1987/88–1991/92*

Farm type and size (BSU)	1987/88 *n*	1987/88 mean size (BSU)	1987/88 RoR (%)	1988/89 *n*	1988/89 mean size (BSU)	1988/89 RoR (%)	1989/90 *n*	1989/90 mean size (BSU)	1989/90 RoR (%)	1990/91 *n*	1990/91 mean size (BSU)	1990/91 RoR (%)	1991/92 *n*	1991/92 mean size (BSU)	1991/92 RoR (%)	1987–92 *n*	1987–92 mean size (BSU)	1987–92 RoR (%)
Specialist dairy																		
Up to 15.9	30	11.87	10.04	30	11.85	13.89	28	11.37	4.84	20	10.42	−0.13	17	10.15	−6.25	125	11.29	5.96
16–23.9	26	19.57	10.21	26	19.32	13.02	18	19.98	14.29	14	19.27	4.27	20	19.48	9.27	104	19.52	10.64
24–39.9	35	30.82	13.76	35	31.23	26.52	38	30.95	17.81	34	31.63	13.30	28	31.13	15.24	170	31.15	17.44
40 and over	27	67.13	25.10	27	69.03	36.06	31	67.10	27.37	36	63.21	19.69	31	60.70	20.65	152	65.22	25.32
All sizes	118	31.83	18.11	118	32.33	27.77	115	34.21	21.16	104	36.82	15.56	96	34.54	16.25	551	33.85	20.01
Mainly dairy																		
Up to 23.9	14	14.14	6.65	14	14.14	4.78	14	15.38	0.01	25	16.31	−1.12	15	14.15	−2.99	82	15.02	1.06
24–39.9	15	31.45	13.41	15	31.79	18.32	13	31.91	13.72	9	34.61	13.60	11	34.52	13.68	63	32.61	14.72
40 and over	18	56.08	15.55	18	54.48	19.36	18	59.89	16.08	15	73.32	10.05	16	72.21	13.70	85	62.63	15.15
All sizes	47	35.73	13.83	47	36.37	17.31	45	37.96	13.24	49	37.12	8.11	42	41.20	11.67	230	37.59	12.81
Hill sheep																		
Up to 15.9	24	10.13	−3.84	24	10.55	1.34	25	10.03	−18.04	22	9.69	−16.54	21	11.36	−3.15	116	10.33	−8.11
16 and over	27	32.57	13.96	27	31.67	20.06	24	33.68	6.14	32	31.14	−1.04	32	34.28	11.14	142	32.65	9.78
All sizes	51	22.01	10.14	51	21.73	15.99	49	21.62	0.54	54	22.40	−3.84	53	25.20	8.76	258	22.62	6.26
Hill cattle & sheep																		
Up to 15.9	39	10.30	3.91	39	10.64	9.49	35	10.87	−8.81	34	11.52	−11.80	25	10.44	−5.86	172	10.75	−1.94
16–23.9	29	19.07	5.58	29	19.52	12.21	32	18.87	−2.55	36	19.65	−4.06	23	19.99	4.70	149	19.40	2.66
24–39.9	26	30.14	12.87	26	30.33	17.70	28	29.82	5.72	29	31.37	2.57	25	31.41	8.29	134	30.61	9.23
40 and over	14	57.77	20.84	14	57.36	20.12	15	70.36	7.22	18	76.13	−3.53	20	74.04	6.70	81	68.13	9.29
All sizes	108	23.59	12.11	108	23.82	15.85	110	26.13	2.38	117	28.88	−2.85	93	32.12	5.37	536	26.79	6.43
Upland cattle & sheep																		
Up to 15.9	16	9.33	−3.66	16	8.65	3.53	16	9.29	−7.42	19	8.29	−17.65	19	7.56	−15.09	86	8.58	−8.64
16 and over	20	26.21	4.64	20	27.43	7.52	18	23.29	−2.07	23	25.66	−3.53	25	30.43	2.14	106	26.82	1.68
All sizes	36	18.71	2.71	36	19.08	6.60	34	16.70	−3.57	42	17.80	−6.57	44	20.55	−0.81	192	18.65	−0.51
Lowland cattle & sheep																		
All sizes	13	12.64	−1.50	13	12.68	1.38	17	18.14	−5.05	31	22.84	−1.59	26	17.90	−0.06	100	18.11	−1.38
Pig & poultry																		
All sizes	6	29.77	3.96	6	22.64	12.94	*	*	*	*	*	*	*	*	*	12	26.20	8.45
Cropping farms																		
All sizes	11	44.84	10.96	11	42.89	1.54	*	*	*	*	*	*	*	*	*	22	43.87	6.25
Summary statistics[1]																		
Total no. of farms	390			390			370			394			353			1,897		
Mean		28.07	9.54		27.81	14.06		28.45	4.91		29.91	0.61		30.16	5.40		29.23	7.07
Trimmed mean		27.11	9.43		26.75	13.61		27.26	4.93		28.61	0.57		29.04	5.67		28.37	6.95
Standard deviation		15.92	7.40		15.93	8.99		17.85	11.17		19.40	10.03		18.98	8.94		17.05	8.45
S.E. mean		3.32	1.54		3.32	1.87		3.90	2.44		4.23	2.19		4.14	1.95		3.56	1.76
Minimum		9.33	−3.84		8.65	1.34		9.29	−18.04		8.29	−17.65		7.56	−15.09		8.58	−8.64
Lower quartile		14.14	3.96		14.14	6.60		16.04	−3.06		17.06	−3.95		16.02	−1.90		18.11	1.06
Upper quartile		32.57	13.83		32.33	19.36		33.94	14.01		35.72	9.08		34.53	12.68		33.85	12.81
Maximum		67.13	25.10		69.03	36.06		70.36	27.37		76.13	19.69		74.04	20.65		68.13	25.32

Notes: [1] The summary statistics are calculated by omitting the 'All sizes' category means (except where this is the only entry for the category).
[2] The 1987–92 mean rate of return is weighted by annual numbers of farms as is the average BSU size.
[3] * = not available.
[4] *n* = number of farms in sample.
[5] rate = nominal rate of net return on tenants' capital, calculated as follows:

MII = Output − Inputs
and RoR = (MII/TC) * 100

where:
(i) Output = All returns from an enterprise, plus the market value of any of its products transferred out to another enterprise, plus the market value of any production from the enterprise given to workers or consumed on the farm. In the case of livestock enterprises, the value of purchased livestock and the market value of livestock transferred in from another enterprise are deducted. All totals are adjusted for changes in valuation.
(ii) Inputs = Feeds (purchased concentrates, home-grown concentrates, purchased bulk) + tack and stock keep + veterinary and medicines + other livestock costs + fertilisers + seeds (purchased and home-grown) + other crop costs + labour (farmer and spouse, paid, unpaid, casual) + machinery (contract, repairs, fuels, depreciation) + general farming costs + other land expenses + rent/rental value + rates. (Note that as a nominal farmer/spouse labour cost is included, we are calculating net rather than gross returns.)
(iii) MII = Management and Investment Income; the MII represents the reward for the farmer's (and spouse's) management and interest on the tenants' capital employed on the farm.
(iv) TC = tenants' capital: the value of livestock, machinery, crops (including cultivations) and stores. In the Farm Business Survey tables, tenants' capital is expressed as the average of the opening and closing valuations for these items.
Sources: Data taken from FBS (1988, 1989, 1990, 1991, 1992).

where:

$$\begin{aligned} \mathrm{RoR_n} &= \text{nominal net rate of return on tenants' capital (per cent)} \\ \mathrm{DAIRY} &= \text{1 for dairy farms (FBS specialist or mainly dairy categories);} \\ &\quad \text{0 otherwise} \\ \mathrm{HIYEAR} &= \text{1 for 1988/89; 0 otherwise} \\ \mathrm{BSU_t} &= \text{average size of farm type in year } t \\ \mathrm{BSU_t}^2 &= \mathrm{BSU}_t * \mathrm{BSU}_t \end{aligned}$$

$R^2 = 77.9\%$; R^2 (adj.) $= 76.7\%$; $F = 66.10$; $p < 0.001$; figures in brackets are t-statistics.

Average $\mathrm{RoR_n}$ for dairy and non-dairy farms (denoted $\mathrm{RoR_n^D}$ and $\mathrm{RoR_n^{ND}}$ respectively) over the study period can now be evaluated by substituting each group's mean values for explanatory variables into Equation (5.1).[52] For dairy farms this gives $\mathrm{RoR_n^D} = 12.68$ per cent, while for non-dairy farms this gives $\mathrm{RoR_n^{ND}} = 1.62$ per cent. This large difference between dairy and non-dairy farms is highly significant and indicates the very considerable positive impact which CAP milk quotas have had upon dairy farm incomes and the parlous state of the non-dairy, agricultural sector within our study area. We return to this theme in Chapter 8.

Conversion to real rates of return ($\mathrm{RoR_r}$) was achieved using retail price indices published by the Central Statistical Office (1993b).[53] These show an average inflation rate for the period 1987–92 of 5.81 per cent implying $\mathrm{RoR_r^D} = 6.86$ per cent and $\mathrm{RoR_r^{ND}} = -4.18$ per cent.

Rates of return in Wales, 1989/90

We were particularly interested in $\mathrm{RoR_n}$ during our study base year of 1989/90 and for the sample of farms that formed the basis of our analysis of agricultural values (presented in Chapter 8). The finding presented above suggests that this year may be typical of a longer time period. Furthermore, the representative sample of 240 farms provided by FBS for our agricultural analysis included grid reference locations which allowed us to consider a wider range of explanatory variables than previously. These included data covering the environmental attributes of the farm (soil type, altitude, etc.) obtained from the LandIS database discussed in Chapter 6. However, while many such variables were significant predictors of $\mathrm{RoR_n}$ they proved to be collinear with the DAIRY and BSU variables considered previously,

52 The assumption of normality implicit in the use of means is relaxed in further testing reported in Bateman (1996) and is shown not to have a significant impact upon findings.

53 Use of the RPI rather than some farm price index reflects the fact that, ultimately, investment funds could be moved out of the agricultural sector.

and these latter variables provided a superior degree of explanation. Following tests of functional form, our best-fitting model for these 240 farms was:

$$\underset{}{RoR_n} = \underset{(-9.66)}{-39.37} + \underset{(6.50)}{12.12}\,\text{DAIRY} + \underset{(9.51)}{13.21}\ln \text{BSU} \qquad (5.2)$$

where:

$$\begin{aligned} RoR_n &= \text{nominal rate of return, 1989/90} \\ \ln \text{BSU} &= \text{natural log of farm size in BSU} \\ \text{DAIRY} &= \text{1 if dairy farm; 0 if non-dairy farm}^{54} \end{aligned}$$

$R^2 = 43.3\%$; R^2 (adj.) $= 42.8\%$; $n = 240$; figures in brackets are t-statistics.

Substituting variable means into Equation (5.2) allows us to calculate the RoR_n for dairy and non-dairy farms, $RoR_n^D = 15.27$ per cent and $RoR_n^{ND} = -2.70$ per cent respectively. Adjusting for inflation (which averaged over 9 per cent in 1989/90) implies $RoR_r^D = 5.81$ per cent and $RoR_r^{ND} = -12.2$ per cent. These results reiterate our previous conclusion regarding the gulf between dairy and non-dairy farms in Wales. Indeed, here we see the latter group making negative nominal and real rates of return, a situation which is clearly unsustainable in the long run and has been evident in disastrously low income levels in the Welsh non-dairy sector during the 1990s (see Chapter 8).

Farm discount rates: summary

While data are scarce, available information suggests that discount rates for agriculture will be low relative to those in other sectors of the economy. Our survey suggests that estimates of general rates as low as 3 per cent in real terms are quite defensible. However, our analysis of rates of return highlights the great variability which exists in the performance of different sections of the agricultural community and in particular, with reference to Wales, the disparity between dairy and non-dairy farms. As Table 5.8 indicates, during our study period the elite of dairy farms consistently recorded nominal (and sometimes real) rates of return in double figures, while, as subsequent analyses have shown, Welsh non-dairy farms regularly showed negative real rates of return. These latter rates were clearly unsustainable and the exodus from Welsh hill-farming throughout the years of our study period has continued up to the present day.

The link between rates of return and discount rates is not simple, involving as it does consideration of time preference. This may raise discount rates above rates

[54] 'Non-dairy' is defined as less than 20 per cent of farm output being milk (n (non-dairy) $= 126$ of which 124 had zero milk revenue, 1 had 3 per cent milk revenue and 1 had 7 per cent milk revenue (the next farm had 24 per cent milk revenue)).

of return, although studies such as Lloyd (1993) suggest that this will not be by a particularly large amount. In the case of dairy farms, rates of 12 per cent and 6 per cent were selected to provide, respectively, an upper-bound and majority best estimate of real discount rates for Welsh dairy farms during our study period. A 6 per cent rate is also the government's specified discount rate (H.M. Treasury, 1991) for 'returns accruing to the public sector from projects in the public sector' (Pearce and Ulph, 1998: p. 268), a rate which has applied from 1989 to the present. For non-dairy farms, rates were clearly significantly lower, with negative rates of return being the reality for many farms in the sector. However, given the unsustainability of negative rates, we felt that a real discount rate sensitivity range from 1.5 per cent to 3 per cent would both embrace the majority of such farms and provide results which were of more relevance to those non-dairy Welsh farms which have survived the traumas of the 1990s.

Social discount rates

We can now formalise and extend our analysis of the factors influencing discount rates as per Equation (5.3) which draws upon the notation of Pearce and Ulph (1998):

$$r = \delta + \mu g \tag{5.3}$$

where:

r = discount rate
δ = rate of time preference (the rate at which utility is discounted)
μ = elasticity of the marginal utility of consumption schedule
g = expected growth rate of average consumption per capita

Economic commentators have long acknowledged that the values of the variables identified in Equation (5.3) which are appropriate to decisions affecting just the individual investor may differ from those values appropriate to decisions affecting the whole of society. The official UK social rate is derived from empirical data averaged over a wide variety of sectors giving values of about 2 for each of the elements of the basic discount rate formula detailed in Equation (5.3), i.e. $r = \delta + \mu g = 2 + (2 * 2) = 6$ per cent. However, a wide variety of views exists regarding the value of each of these elements.

Perhaps most controversial is the value of δ, the pure rate of time preference in the social discount rate (r_s). If society is immortal (or aspires to be) then, as very many eminent commentators have pointed out, δ should be very low or zero (Ramsey, 1928; Pigou, 1932; Solow, 1974b, 1992; Broome, 1992; Cline, 1992a, 1993; Fankhauser, 1993, 1995; Price, 1993; Arrow *et al*., 1996; Pearce and Ulph, 1998). Such arguments have been reinforced by the debate surrounding sustainable development. This has centred upon notions of Rawlsian equity (Rawls, 1972) wherein, to be truly equitable, decisions regarding the use of resources (be they involving man-made, human or natural capital)[55] should be made behind a 'veil of ignorance' with respect to their temporal impact. Such a view is fundamental to the often quoted Brundtland Commission definition of sustainable development as 'development that meets the needs of the present without compromising the ability of future generations to meet their own needs' (World Commission on Environment and Development, 1987). Price (1993) sees this as only interpretable as an abandonment of discounting for global-level social decision-making.

A more 'conventional' view is given by Fankhauser (1993) who sees the requirements of sustainable development as implying that $\delta = 0$, but not necessarily that $r_s = 0$. Pearce and Ulph (1998) review an extensive literature on social δ, reporting a range from 0–1.7 per cent but favouring (for empirical reasons) a relatively high best estimate of $\delta = 1.4$ per cent.

Turning to consider the elasticity of the marginal utility of consumption (μ), Price (1993) reports a wide spread of private sector rates, generally ranging from 0.5 (Squire and van der Tak, 1975) to 3 (Little and Mirlees, 1974).[56] Stern (1977) finds many values in the region of 2.[57] However, we would expect the social preference value of μ to be somewhat lower than that found in the market. This is borne out by Pearce and Ulph (1998) who report a best estimate of social μ of 0.8 with a range of 0.7–1.5.

The social value of g (the expected rate of growth of average consumption per capita) is typically taken as being the real rate of growth of national income. Following such an approach, Lind (1982b, 1982c) argues for a maximum rate of $g = 2$ per cent.[58] However, the sustainable development debate has highlighted the problem that accounting measures such as GDP often ignore changes (frequently losses) in the natural and other non-market capital base of the economy (Repetto *et al*., 1989).[59] Taking account of these, Pearce and Ulph (1998) suggest a best estimate for g in the UK of 1.3 per cent with a range of 1.3–2.2 per cent.

Taking best estimates from Pearce and Ulph (1998) gives a central estimate of r_s for the UK of about 2.4 per cent ($= 1.4 + (0.8 * 1.3)$). While this may seem

[55] For an introduction to the role of capital types in notions of sustainability, see Turner and Pearce (1993) or Pearce and Barbier (2000). While radical from a neoclassical perspective, more extreme (but very interesting) views are given in the work of Herman Daly (Daly, 1977; Daly and Cobb, 1990; Daly, 1995).

[56] μ is negative but we report modulus values following the convention of Pearce and Ulph (1995).

[57] Stern (1977) reports one extreme value of $\mu = 10$.

[58] Turner *et al*. (1994) point out that real growth in GDP in less developed countries is often much lower or even negative.

[59] Repetto puts forward an adjusted, sustainable national income measure. See also Pearce *et al*. (1989), Pearce and Warford (1993) and Pearce (1993).

low with respect to the Treasury's rate,[60] it is higher than that put forward by certain other commentators, particularly with respect to the discounting of global warming damages (perhaps the most potent challenge to intergenerational equity in the history of man). While not stating any particular rate, Arrow *et al.* (1996) make explicit reference to the range of 0–2 per cent used by Cline (1992a) in his economic analysis of long-run climate change models. Similarly, in his evaluation of the social costs of greenhouse gas emissions, Fankhauser (1993) uses a central (mode) estimate of $r_s = 0.5$ per cent with a range of 0–3 per cent (the upper end being mainly for comparison with other studies).

A further complication arises from the issue of multiple discount rates: the notion that social preferences may diverge radically between projects to the extent that a single discount rate is something of an oversimplification. As Arrow *et al.* (1996) and many earlier commentators have pointed out, the key factor here is substitutability, i.e. the extent to which development benefits (often in terms of man-made capital, K_m) can be traded off against costs (generally in terms of natural capital, K_n). Assuming that sustainability is socially desirable and that both sets of capital can be measured in some comparable numeraire (presumably money), then perfect substitutability would mean that any project would simply have to pass a standard Hicks–Kaldor hypothetical compensation test to be sanctioned.[61] In the literature of sustainable development this has been termed the 'very weak sustainability' rule (Turner and Pearce, 1993), which states that, provided total net benefits (total capital) are non-declining, a project may be sanctioned. This perfect substitutability assumption may be more acceptable for some K_m/K_n swaps (e.g. Sitka spruce plantations into paper, thence into money and so back to new plantations) than for others (e.g. the destruction of SSSIs to make way for motorways[62]), i.e. some K_n destruction is irreversible.

Pearce and Turner (1990) define a continuum of capital types from money (the purest form of K_m), through various types of K_n (trees, land, etc.) to 'critical natural capital' (K_n^c),[63] the last being those services of the planet vital to life support (climate and atmosphere control, ozone layer, etc.). As we move away from money along this continuum, the potential for substitution, rather than staying constant, falls until it reaches zero with K_n^c.

Such a view causes problems for cost-benefit analysis if we feel that the accumulation of K_m does not adequately compensate for the loss of K_n. This is the view of the 'weak sustainability' rule (Turner and Pearce, 1993) which argues that stocks of K_n^c should be inviolate, while K_n should be subject to some safe minimum standard

60 Pearce and Ulph (1998) suggest that for policy purposes the Treasury should use a range of 2–4 per cent.
61 See almost any cost-benefit text, for example Pearce (1986).
62 As in the case of the M3 Twyford Down extension in southern England.
63 The term is borrowed from Pearce and Turner (1990).

(SMS) below which use should be prohibited.[64] A further interpretation, the 'strong sustainability' rule, in effect argues that such an SMS has already been breached and that any further use of K_n should be offset by actual physical compensation in terms of shadow projects restoring, transplanting or recreating levels of any K_n used in future projects.[65]

The divergence between best estimates of r_s given by Pearce and Ulph (2.4 per cent) and Fankhauser (0.5 per cent) or Arrow *et al*. (implicitly 0–2 per cent) can therefore be viewed as comparing a general rate of K_m/K_n substitutability with that of a non-substitutable good: global climate. The implication of such an analysis is that, because of the various rates of substitutability and irreversibility inherent in the differing capital bases of each project, society will have different discount rates for different projects. Furthermore, we could extend this line of reasoning to the individual costs and benefits of a single project so that, in our forestry case study, UK timber (for which losses are reasonably reversible) might attract a higher r_s than recreation benefits (which arguably belong to a more depleted set of K_n), which in turn is more discounted than carbon storage (which contributes to the K_n^c stock of global climate services). Following this argument we examine the impact of using differing discount rates in our forestry case study.

In practice, the variance of r_s within a project is clearly a decision-making nightmare and opens up the potential for discount rate 'management' abuses. Indeed, the avoidance of abuse may be the most cogent argument for adopting a single rate policy. Henderson and Bateman (1995) report numerous examples from around the world of both inter- and intraproject multiple discount rates.[66] However, these appeared to be almost exclusively motivated by policy objectives rather than empirical evidence regarding underlying preferences. Unfortunately, the manipulation of discount rates to give policy-favoured projects a spurious sheen of financial respectability is widespread if invalid.

The desirability of a single rate is therefore clear. The Pearce and Ulph (1998) results (central estimate $r_s = 2.4$ per cent; range $= 2$–4 per cent) are useful here but we have to recognise that probably the recreation benefits, and almost certainly the carbon sequestration benefits, of woodland would attract a lower than average rate of public pure time preference. Accordingly, we have chosen a sensitivity analysis

64 Under weak sustainability, further use of K_n up to the SMS must still be compensated for by re-investment (savings) of the appropriate level of K_m proceeds from each project (Turner and Pearce, 1993).

65 Under strong sustainability an individual project must compensate K_n both in terms of K_m savings and by appropriate contributions to an offset physical compensation, shadow project fund. Such physical compensation must be actual rather than hypothetical (rejecting the Hicks–Kaldor rule). A still stronger view (very strong sustainability) states that each project must have its own actual physical K_n compensation shadow project (see Turner and Pearce, 1993).

66 Henderson and Bateman (1995) examine theoretical and empirical arguments in favour of hyperbolic discount rates. Bateman (1996) reassesses all of the analyses presented in this volume using a hyperbolic discount rate and shows that this further increases the potential for transfers from agriculture to multipurpose woodland.

for r_s which includes one rate (1.5 per cent) below the Pearce and Ulph range[67] and another at the centre of that range (3 per cent). For comparative purposes we have also employed the Treasury's 6 per cent public sector discount rate throughout, although we echo the sentiments of Pearce and Ulph that this seems 'very difficult to justify'.

Discount rates: conclusions

Given the major impact which variations in the discount rate will have upon long-delayed forestry returns, we feel that our discussion highlights the need to adopt a sensitivity analysis approach. We feel that real social discount rates of 1.5 per cent and 3 per cent are well justified as a reasonable range here. The Treasury's 6 per cent rate is also included for comparative purposes. Turning to consider farmers' real private discount rates, the 1.5 per cent and 3 per cent rates are useful for assessing decisions in the Welsh non-dairy agricultural sector.[68] Conversely, rates of 6 per cent and 12 per cent roughly describe reasonable limits to apply to dairy farms in Wales.[69]

The private value of timber production

The discussions in this chapter show that the private value of a productive plantation is determined by a variety of factors including species, plantation costs, timber yield, timber price (where both the price–size curve and assumptions regarding future real prices are important), grants and subsidies, and the discount rate. All these factors were brought together by integrating data from the FC yield models (Edwards and Christie, 1981) for Sitka spruce (YC6–24) and beech (YC4–10) within a series of spreadsheets. This allowed easy manipulation of all assumptions (e.g. grant schemes, discount rates, optimal felling age,[70] etc.) to produce a full range of private NPV and annuity equivalent values. Results from this exercise are reported in full in Bateman (1996) which details a variety of permutations, ranging across all species, yield class, discount rate and subsidy scheme scenarios. Results are also calculated for a single, optimal rotation of trees and for a scenario of perpetual replanting after each felling. As this produces a plethora of permutations, we reproduce results for just one scenario here. Figure 5.5 graphs

[67] This also reflects the lower-range estimates of Fankhauser (1993) and Arrow *et al*. (1996).
[68] We recognise that a number of these farms may not be attaining rates of return of even 1.5 per cent. However, such farms are unlikely to keep operating in the long term. The 3 per cent rate also provides an assessment under conditions similar to those likely to apply if discount rates are cut as recently proposed by H. M. Treasury.
[69] Discussion of results obtained using hyperbolic discount rates is given in Bateman (1996).
[70] Set as per Tables 5.6 (for conifers) and 5.7 (for broadleaves).

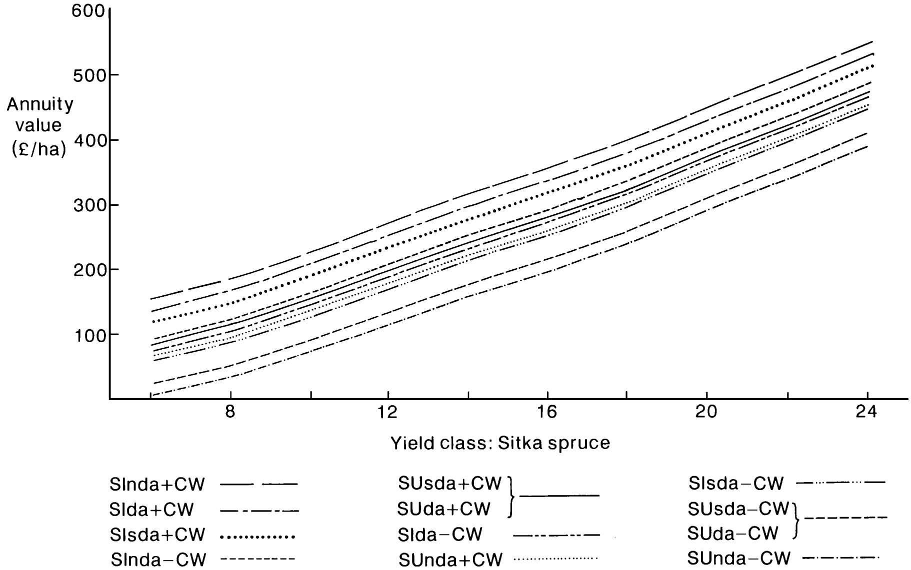

Figure 5.5. Farmers' private timber values for Sitka spruce (annualised equivalents of a perpetual series of optimal rotations: $r = 3\%$). Various yield classes and subsidy types.

annuity equivalents for the full range of Sitka spruce yield classes and all feasible grant scheme registrations (using the abbreviations developed at the end of the earlier section on grants (p. 128) assuming that a system of perpetual replanting is used and a 3 per cent discount rate is applied. Figure 5.6 repeats the analysis for beech.

For both Sitka spruce and beech we see that, as expected, annual equivalent values rise with yield class (just as they fall with discount rate; see subsequent results). As subsidy schemes are not linked to timber productivity the difference between scheme payments is constant across yield classes. Comparison between Sitka spruce and beech is interesting as it shows that, holding yield class constant (i.e. YC 6, 8 or 10), returns from broadleaves are higher than for conifers. This is due to higher prices and subsidy levels for broadleaves and occurs despite the lower felling age of conifers. However, because conifers are capable of much higher yield classes than broadleaves on almost any given site and, more importantly, because such high-yield plantations have much lower felling ages (thus avoiding the severe discounting that occurs with long-rotation broadleaves), they provide much higher annual equivalents than broadleaves.

An overview of discounting impacts is given in Table 5.9 (full details for all yield class/species combinations are presented in Bateman, 1996). Here annualised

Table 5.9. *Farmers' private timber values for high-output Sitka spruce and beech across various discount rates (annualised equivalents of a perpetual series of optimal rotations)*

Discount rate (%)	Farmers' private value (annualised equivalent, £/ha)	
	Sitka spruce (YC24)	Beech (YC10)
1.5	496.30	103.54
3	388.46	80.68
6	219.36	31.21
12	19.45	9.59

Note: Subsidy option for all cases is SUnda−CW = subsidy for previously unimproved grassland, not in a disadvantaged area and without Community Woodland Supplement.

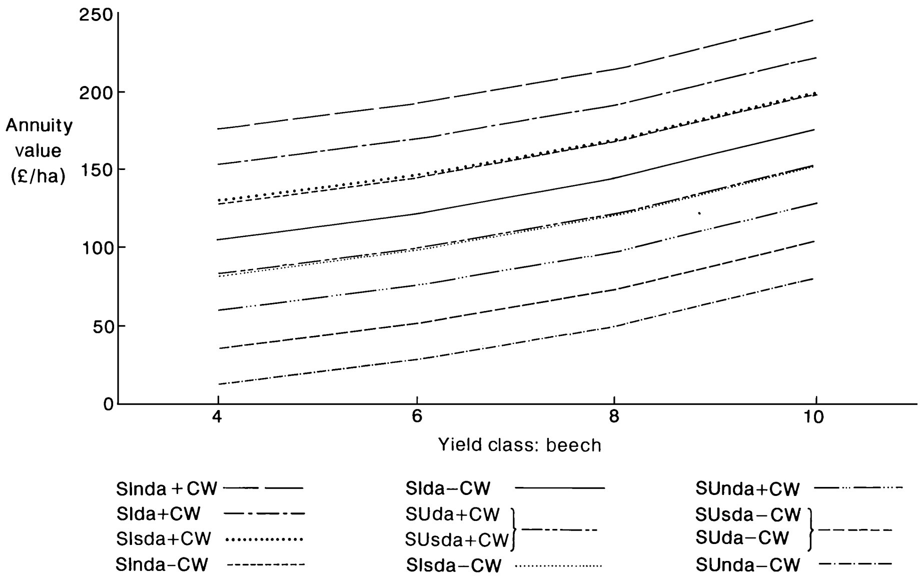

Figure 5.6. Farmers' private timber values for beech (annualised equivalents of a perpetual series of optimal rotations: $r = 3\%$). Various yield classes and subsidy types.

equivalents for highest output Sitka spruce (YC24) and beech (YC10) under one subsidy permutation are given for all the discount rates considered.

In subsequent chapters we examine how forest timber values compare with agricultural returns under a variety of scenarios. However, we now turn to consider the other, non-market, benefits of woodlands.

The social value of woodlands

In moving from the private to the social value of timber production a number of issues need to be addressed. The basic plantation costs and timber (thinnings and maincrop) benefits can defensibly be used in an unaltered form as, unlike the prices of agricultural produce, UK timber prices are not the subject of intervention or otherwise controlled. However, we do have to subtract all grants and subsidies, as these are simply transfer payments, to obtain our baseline social value for timber net benefits (see Bateman, 1996, for further details).

As discussed in our opening chapter, the social value of a woodland is more than just the value of timber therein. In earlier work (Bateman, 1992) we identified and discussed a detailed set of environmental and non-environmental non-market costs and benefits which may arise from afforestation. Here we summarise that discussion by briefly considering the major non-market items which may need to be examined when moving from a private to a social assessment of woodland.

Non-environmental non-market social costs and benefits

Here we discuss four major issues: national security; economic security; import substitution; and employment.

National security

While national security formed the impetus for the creation of the Forestry Commission just after the First World War, and was an important spur to planting after the Second World War, the prospect of the UK being blockaded from receiving timber supplies for any extended period seems rather unlikely. We therefore conclude that there are no significant national security benefits to be derived from the expansion of a domestic supply capability.

Economic security

While not of strategic importance, uninterrupted security of supply does bring avoided-cost benefits. In a study of this issue, Pearce (1991) states that 'an evaluation of the chances of embargoes and other supply interruptions suggests that a small increment in prices of 0.2 to 0.8 per cent to reflect the shadow value of economic security would be justified'. Accordingly, timber benefits were increased by 0.5 per cent in our social evaluation models.[71]

[71] Note that this is an across-the-board single increase, not a compounding of an annual real price increase. Consequently the net effect is very small.

Import substitution

As mentioned earlier in this chapter, timber is one of the UK's major import items. In 1999 the UK imported over 21 million m^3 of wood and panels and over 25 million m^3 of pulp and paper (FC, 2001c). Despite this dependence on imports, the basic theory of comparative advantage has long shown that this does not necessarily imply that the UK should strive to change this situation (see, for example, Söderstern, 1980). This theory shows that a given amount of resources should only be invested into reducing timber imports if those same resources cannot be invested more productively in some other manner. Repeated studies of precisely this issue have consistently shown that this is not the case (H.M. Treasury, 1972; Bowers, 1985; National Audit Office (NAO), 1986; Pearce, 1991) and so the import-saving argument is rejected.

Employment

It has been argued that creating jobs in forestry is a good way to stem the ongoing trend of rural depopulation and combat the psychological and other economic costs of rural unemployment. However, numerous studies have suggested that forestry is a relatively expensive and inefficient method of providing rural employment, particularly when compared to agriculture (H.M. Treasury, 1972; Laxton and Whitby, 1986; NAO, 1986; Evans, 1987; Johnson and Price, 1987). Forestry expansion could therefore be seen as creating shadow costs.

Such conclusions were tentatively disputed by studies in the early 1990s which argued that, as Forestry Commission employment was falling and productivity rising, an economic benefit of rural employment might occur over the course of the decade (Thompson, 1990; FICGB, 1992). However, in the event, the steady increase in private woodlands meant that employment in British forestry (excluding primary processing) rose from about 17,000 in the mid 1980s to over 27,000 a decade later (FC, 1985b, 1997). The low-employment/high-productivity argument may be due for a revival in coming years as employment levels have recently fallen to just over 18,000 (FC, 2001c) and it is interesting to note that rural employment has again moved to centre stage as a policy argument for increased forestry (FC, 1998). Our view is that a more likely promoter of the economic case for forestry employment benefits is the parlous state of UK agriculture (discussed in Chapter 8). However, in the absence of a specific and contemporary study these are mere speculations and a cautious approach is to assume that the case for the employment benefits of forestry is unproven.

In conclusion the only clearly valid non-environmental, non-market social benefit we can isolate is a small benefit due to increased economic security of supply and we adjust social values marginally (as indicated above) to reflect this.

Environmental non-market social costs and benefits

Woodlands create a number of social benefits and costs of which we discuss the following major issues: recreation; carbon storage; acidification impacts; landscape amenity; biodiversity effects; and other non-use (bequest and existence) values.

Recreation use and option value

Recreation use value is the major focus of our valuation research as discussed in Chapters 2–4. Because of the potentially significant problems of declining marginal utility,[72] we have decided not to incorporate such benefits within the plantation value models presented in this chapter. Instead these models deal primarily with timber values to which recreation benefits are added in subsequent chapters.[73] One potential deficiency in our research is that travel cost estimates of recreation value ignore option values. These are in theory addressed through our contingent valuation studies; however, we recognise that option value is not a principal aim of these studies.

Carbon sequestration

As with recreation, we deal with carbon sequestration separately (in chapter 7). This is not because of diminishing marginal utility, for (as explained in later chapters) the likely levels of sequestration will not have a significant impact upon the global CO_2 budget, but rather because of the complexities of this issue which we feel deserve separate attention.

Acidification

Forests are cited as both the victims and perpetrators of acidification damage. Although research into the impact of acidic deposition upon trees is abundant (European Commission and the United Nations Economic Commission for Europe, 1994; Marques *et al*., 2001; Takahashi *et al*., 2001), relatively little ongoing work concerns the contribution, if any, which trees make to acidic impacts upon soil and watercourses (Hornung and Adamson, 1991). Indeed, some dispute what they term ‘the myth of soil acidification’, asserting that ‘Within its lifetime, a spruce cannot significantly acidify the soil below it’ (Baldwin, 1996: p. 1). The Forestry Commission suggests that forests tend to act as a catalytic fixing medium for industrially emitted atmospheric acid (Innes, 1987); others argue that this is only part of the story and that conifers, in particular, directly contribute to a lowering

[72] As the area of woodland expands we would expect the increase in recreation opportunities to result in an observable decline in per hectare recreation values. Given supply and demand conditions we would not expect this to be a problem for timber production.

[73] As discussed elsewhere, this implicitly assumes that the monetary evaluations of woodland recreation are surpluses to the amenity value of the present agricultural landscape.

of pH levels (see Harriman and Morrison, 1982; Batterbee, 1984; Edwards *et al*., 1990;[74] Nisbet, 1990; Grieve, 2001). We take the position that whether or not forests actually generate the acids concerned, they are significantly linked to increased acidification of soils and aquifers in non-buffered areas and therefore do generate costs. Our research in this area has not progressed beyond the stage of a literature survey, although this has shown that the acidification problem is eminently amenable to GIS analysis, which we intend to conduct in future research.[75]

Landscape amenity

The remit of this study excludes the landscape amenity values of forestry. Although the contingent valuation method has been applied to general landscape valuation (see, for example, Willis and Garrod, 1993), these studies have not looked specifically at the impact of woodland. However, a number of hedonic pricing studies have demonstrated that forests do generate significant amenity values, as reflected in property prices (Garrod and Willis, 1992a,b,c; Powe *et al*., 1997; Peterson and Boyle, forthcoming). Taken together, these studies indicate that while broadleaves generate landscape amenity benefits, certain conifers, including Sitka spruce, can result in amenity losses. Such results therefore constitute a caveat to our own findings. However, our recent research shows that GIS techniques are particularly appropriate to the estimation of landscape values via the hedonic pricing method (Lake *et al*., 1998, 2000a,b; Bateman *et al*., 2001a). The GIS allows the definition of 'viewsheds' quantifying what can and cannot be seen from any given location. Derived variables have been shown to be powerful predictors of amenity values (*ibid*.) and we intend to apply such an approach to valuing woodland landscape in future research.

Biodiversity impacts

Work, in collaboration with ecological scientists, is currently ongoing in an attempt to incorporate biodiversity impacts into our model of woodland values. Early findings indicated that afforestation of agricultural areas by conifers such as Sitka spruce is liable to change the balance of bird species to the detriment of some of the most endangered birds in Wales (Bateman *et al*., 1997c). More recently we combined GIS techniques with data provided by the British Trust for Ornithology to generate algorithms for selecting priority areas for conservation in Wales (Woodhouse *et al*., 2000).[76] In ongoing extensions to this work we are linking these findings with land use change data to model the relationships between agriculture, forestry

[74] This collection of papers focuses exclusively upon acidification in Wales.
[75] Such research would also allow consideration of related issues such as the impact of afforestation upon hydro-electric potential (see Barrow *et al*., 1986).
[76] See also the GIS-based approaches of Swetnam *et al*. (1998) and Cowling and Heijnis (2001).

and species diversity. This research should provide a mechanism for investigating the biodiversity consequences of policy decisions and consequent land use change in a manner which will link to the CBA assessments presented in this volume.

While there exist substantial (and possibly insurmountable) practical, economic and philosophical problems in the valuation of biodiversity impacts (Garrod and Willis, 1994; Kahn, 1995; Carson, 1998; Shogren *et al.*, 1999), it seems likely that afforestation with non-native species such as Sitka spruce would induce a loss of unknown and potentially substantial magnitude which should be set against the values reported in subsequent chapters. The biodiversity impacts of planting beech woodland are generally (although not exclusively) considered to be positive and should similarly be set against the other values presented subsequently.

Other non-use values

Biodiversity values may arguably provide a proxy for wider existence values (although this is debatable). However, other non-use issues such as bequest values do not feature explicitly in our study (although potentially they may influence our contingent valuation findings) and this provides a further caveat to the accuracy of our results.

Non-market social costs and benefits: summary

We are left with having to acknowledge a number of deficiencies in the extent of our study. While we feel that our analysis is relatively sophisticated and useful in a policy-making context, it remains far from perfect. Nevertheless, those items which we feel to be of major significance (recreation and carbon sequestration) are dealt with outside our rotation model in other chapters. Of the remaining social costs and benefits, economic security arguments seem to justify a minor upward revision of social benefit values which is quantified and incorporated within the rotation values model. Of the remaining issues, acidification, biodiversity and non-use values remain insufficiently addressed but the subject of ongoing research. If we accept that this must remain a partial analysis until that work is complete, we would defend the present study both as a significant improvement on existing CBA assessments of forestry values and, more importantly, as demonstrating an improved methodology for conducting such studies.

Annual equivalent social timber values

Given the above caveats, we can now calculate social net benefit timber values for our plantation models. These include timber values and the value of economic security of supply but exclude recreation and carbon sequestration values which

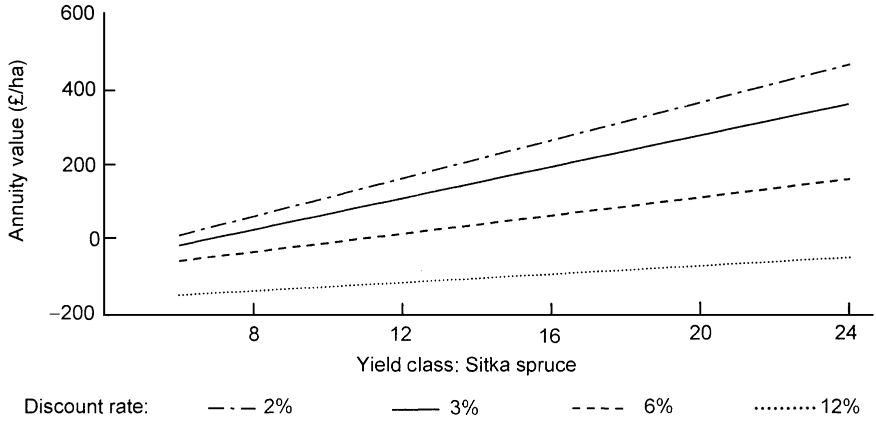

Figure 5.7. Social value for Sitka spruce (annualised equivalent of a perpetual series of optimal rotations). Various yield classes and discount rates.

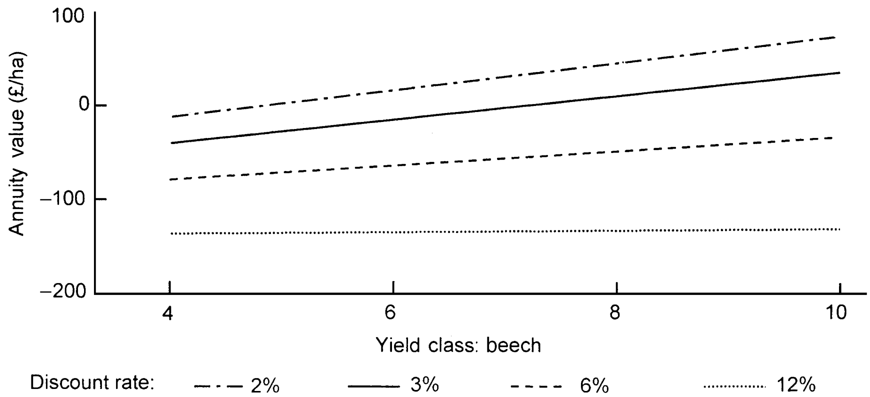

Figure 5.8. Social value for beech (annualised equivalent of a perpetual series of optimal rotations). Various yield classes and discount rates.

are dealt with subsequently. As there is no subsidy dimension to these calculations (remember that we have removed all transfer payments) we can illustrate results across all yield classes and discount rates on a single graph as shown in Figure 5.7 for conifers, and Figure 5.8 for broadleaves.

Comparison of Figures 5.7 and 5.8 shows relationships similar to those observed in the private sector evaluations. Again we see (on this restricted range of value types) that conifers are able to outperform broadleaves (interestingly values for broadleaves are negative for most low yield class/high discount rate combinations, illustrating the impact of discounting on the long rotation periods of such trees). Given that we have excluded recreation and non-user values, such a result is not unexpected.

Conclusions

We have constructed rotation models which, across the full range of yields for our two representative tree species, take into account plantation costs and benefits, real prices, grants and subsidies. We have also considered the difference between private and social perspectives both in terms of differential discount rates and with regard to the differing range of values which the two assessments should appraise. In subsequent chapters the private and social values derived from this analysis are aggregated with our estimates of recreation and carbon sequestration values to provide our overall assessment of the values generated by farm forestry. These values are then compared with those for existing agricultural activities so as to estimate likely conversion rates under a variety of scenarios.