5.4   Buffer zones

Minimum intervention reserves will always be prey to unwanted influences from the surroundings, but these can be reduced by buffer zones. These combine two functions: (i) shielding of the reserve from species and conditions in the matrix habitats, and (ii) enlarging some aspects of the minimum intervention reserve.

Lateral influences include windborne soil and nutrient drift, insecticide and herbicide drift, modification of incoming water flows and quality, and the movement of plant and animal populations in matrix habitats into the woodland (section 3.1.4). In addition, there are many kinds of edge effect, which span anything from enhanced light intensity and decreased humidity within the wood edge to increased rates of nest predation by farmland species on woodland species. If the minimum intervention reserve is surrounded by mature woodland, albeit managed, these influences will be minimised. Furthermore, buffer woodland also enlarges the area of mature woodland and the populations of species within the minimum intervention reserve.

The ideal management for buffer zones would probably be continuous cover systems of site-native species. Alternatively, small-scale clear-cuts within a normal age-class distribution run on a rotation of about 100 years would simulate some aspects of natural woodland within the buffer zone.

The need for a buffer zone plays a part in site selection (section 4.2.2). For this and other reasons, there is a case for establishing minimum intervention reserves within large woodlands. If this is not practicable, a large wood should be assigned to minimum intervention, save for a marginal zone where more intensive operations are acceptable, eg in order to control invading exotics.

5.5   Responses to damaging changes

Minimum intervention reserves may be subjected to several kinds of unwelcome event, so it is prudent to decide in advance how to respond.

5.5.1   Unwanted tree and shrub invaders

These will generally take the form of chronic invasions by naturalised species, such as sycamore, rhododendron, Norway spruce, western hemlock, from nearby woodland that is not within the control of the reserve managers. The objectives of the response should have been resolved at the outset when a decision was made about whether to seek future-natural woodland, or not. If this is *not* the intention, the invading species should be cut out. Inspection and remedial action on a regular basis of, say, 5-10-years should be sufficient. Ideally, such action would be confined within the buffer zone.

5.5.2   Grey squirrels

These rodents can be quite devastating to broadleaved woodland, stripping bark from beech, sycamore, sallows and oak, but usually not ash and lime. They concentrate their attacks on fast-growing poles, commonly killing the leader and reducing individuals to malformed shrubs.

They also debark the upper sides of crown branches of mature trees, introducing rot fungi and increasing the rate of loss of crown branches.

Grey squirrels can exert highly selective effects in any woodland. In woods where beech might naturally dominate, they have the potential to transform the outcome of natural processes. In original-natural and inherited-natural minimum intervention reserves they should be controlled, and there seems little objections to the use to standard techniques. In future-natural minimum intervention reserves, there is a case for allowing them free rein.

5.5.3 Semi-natural disturbances

Minimum intervention reserves are not immune to elm disease and they are unlikely to be free of diseases affecting alder, oak or any other tree species. If remedial actions are available, such as the fungicide injections against elm disease, there would be a case for using them in original- and inherited-natural minimum intervention reserves, but the cost is likely to be prohibitive. It is difficult to avoid the conclusion that we will just have to live with them, accepting that this is one reason why minimum intervention reserves cannot be truly natural.

5.5.4 Invading non-native ground vegetation

Some of the more robust colonists can dominate ground vegetation and control forest processes, notably the carpet- or clump-forming species, such as *Vinca minor* and Japanese Knotweed. Other species may become abundant, but probably exert no more control than the native species, eg *Impatiens glandulifera*.

Most species will have to be accepted. Well-established clumps will probably disqualify a wood from becoming minimum intervention reserve, but if they are present efforts should be made to remove them during set-up treatments (5.1.1). Species that start to colonise after a wood becomes an minimum intervention reserve should be arrested before they cross the buffer zone. There seems little objection to herbiciding the advance guard if it has not penetrated the core of the reserve.

5.6 Levels of minimum intervention

Interventions in minimum intervention reserves should be carefully controlled. Control should cover (i) whether the high forest or wood-pasture models should be adopted, (ii) which kind of naturalness should be targeted (3.4), (iii) the need for and extent of set-up treatments (5.1), and (iv) policies on fencing, paths, buffer zone management and responses to unwelcome changes.

One approach to control has been developed by Jim Latham of CCW. He recognises three classes of minimum intervention, which would form a structured basis for permitted operations. These are given in his words [with my comments]:

Non-intervention. Literally no intervention and acceptance of any outcome. [This would only be possible where the target is future-natural woodland].

Protective intervention. Control of grazing and non-native species, but no constraints on the abundance, composition and structure of native species. This is the widely understood

definition of minimum intervention. 'Protective' refers to protecting natural processes. [This broadly corresponds with type Ia objectives (3.4)].

Limited intervention. Limits are set on the change that would be accepted, eg, no less than 50% cover by beech, or no more than 10% cover by sycamore. Protective intervention applies within these limits, but direct intervention would be allowed if a limit were breached. [As stated this would be just outside the concept of a minimum intervention reserve, and closer to minimum intervention as a prescription (1.5).]

6. Minimum intervention inclusions in managed woodland

Managed woods are not minimum intervention reserves, but management can, and often does, generate mature habitats similar to those that usually develop in minimum intervention reserves. This section considers the character and value of mature habitats in managed woodland, and addresses the issue of small minimum intervention stands within otherwise managed woods.

6.1 Mature structures in managed woods

Traditionally managed native woods contain several kinds of large, old tree:

- Coppices contain standards which have often been allowed to develop into substantial, spreading trees during the 20th century. They also included pollard trees, usually on the margins and internal boundary banks. Coppice of ash, lime, oak, beech or elm developed large stools with rotten cores. The coppices in some districts contained high-cut stools and stubs, and in a few places the 'standard' oaks were treated as pollards.

- Wood-pastures traditionally comprised pollard and shred trees, that developed substantial trunks containing large decay columns, mixed irregularly with younger trees and shrubs. Latterly, these pollards have rarely been lopped, and their crowns have developed into substantial structures, often with large quantities of dead branchwood.

Forests managed as high forest may also contain large mature trees and complex vertical and horizontal structure. The possibilities include:

- Long rotation stands. Stands retained beyond commercial rotation, which eventually develop large trees and mature structures, even if they originated as even-aged plantations. Some stands achieve this state simply by being neglected. Elsewhere, they are deliberately allowed to develop where amenity considerations are prominent.

- Selection systems. Whilst individual trees may not be retained to a great age, felling and restocking can mimic the gap processes of natural woodland. Selection and group selection systems are still rare in Britain, but they are becoming commoner in beech-dominated stands as interest grows in the techniques of continuous cover forestry.

● Two-storied high forest. A silvicultural system that retains a shelterwood of mature trees, and leaves them to grow through a second rotation in a matrix of younger growth. This remains rare as a system in Britain, but the equivalent structure has been generated in (i) woods that have been partially felled; (ii) in wood-pastures where grazing intensity has been markedly reduced; and (iii) in neglected coppices.

Levels of dead wood in these circumstances rarely achieve natural levels (Kirby *et al* 1998). Some wood-pastures may have routinely contained 100 m³/ha, but coppices generally contained less than 10% of natural volumes. Neglected stands and long-rotation high forest may build up to 30-50% of natural levels, but rarely achieve more. Selection and 2-storied high forest systems can be operated with very little dead wood, but in practice they probably average around 20-30% of natural levels.

6.2   Discrete minimum intervention stands in managed woods

Some otherwise managed woods contain small stands treated as non-intervention. These are generally sited on ground that is too inconvenient or expensive to work, notably small, deeply incised valleys ('gills') and riparian strips. Narrow minimum intervention stands may also be retained around wood margins as a screen for forestry operations within. Most such stands have been left unmanaged for no more than 50 years or so, and it is possible that they will eventually be wholly or partly felled. While they remain, they increasingly develop large trees and accumulations of dead wood.

Small minimum intervention stands have also been deliberately retained within woods. Some are legacies from a time when owners were prepared to set aside a small patch 'for conservation' whilst all around the wood was converted to conifer plantations. In these circumstances the retained semi-natural patch was effectively treated as minimum intervention woodland. Other, more idiosyncratic motivations have been encountered: in one wood a group of trees was left entirely untouched as a memorial grove, where the owning family scattered the ashes of its pet dogs.

6.3   Value of small minimum intervention stands in managed woods

The issue here is whether there is any value in setting aside a compartment or sub-compartment in a managed wood as an minimum intervention reserve. Typically, these might be compact stands about 2ha in extent, surrounded by woodland managed on a commercial rotation, ie with no special provision for mature structures.

In terms of the three general objectives of minimum intervention stands:

Science. Such stands would be far too small to maintain a full cycle of conditions, and could only include one or two gaps. Edge effects would be felt throughout the stand. They would be subjected to the grey squirrel population in the wood as a whole, and, unless they were fenced, they would likewise share the grazing environment of the surroundings. They might just be large enough to act as reference points for experimental research in the surrounding managed stands. For example, research on compaction and other soil changes resulting from extraction by heavy machinery would benefit from access to a sample of ground that had been free of heavy machinery. However, for most subjects the value of very small minimum intervention stands for research is minimal.

Nature Conservation. Small minimum intervention stands diversify stand structures in the wood as a whole, and probably provide a greater density of dead wood than other parts of the wood. The ground vegetation is unlikely to differ significantly from the rest of the wood, but large trees might harbour epiphytes that do not occur elsewhere. The number of species in a minimum intervention sub-compartment may well be less than the number in some managed compartments, but on a whole-wood scale small minimum intervention stands add a distinctive habitat and specialised species. This is probably the most important role of small minimum intervention stands in managed woods.

Culture. A few mature trees in an otherwise managed wood provide some visual diversity and a sense of scale. They may include large trees with individual associations, which become the focus or target of a walk.

6.4 Design requirements

Is there a minimum useful size? For many benefits there seems no justification for a particular minimum. Even a single large tree can afford the visual benefits of a group, and even semi-isolated trees in wood-pasture may harbour exacting saproxylic species. However, one can envisage that some birds may need a minimum size, and certainly the Forestry Commission leave substantial groups untouched around the nest site of a rare bird of prey. A degree of permanent shelter may help, say, some fungi. Furthermore, very small stands may be vulnerable to wind if all adjacent compartments are in a much younger stage of growth.

If we accept that edge effects generally extend 50m into a stand, then a stand would have to be 100m across its smallest width to include even a vestige of interior conditions, ie a minimum of 0.8ha. Since this minimum refers to a circular stand, which is not a common shape in forestry, the effective minimum would be a 100m square, ie 1 ha.

Does location matter? The premium may be on developing 'interior conditions', ie shelter and minimal side light. This points to a location within a wood, where the stand will be buffered by the rest of the wood, and in hollows or valley bottoms, where wind-speeds, light and humidity will be most like a large forest interior. Unmanaged marginal broadleaved belts around woods otherwise treated as plantations tend to be dominated by edge effects.

6.5 General

In the interests of habitat diversity and thus biodiversity, individual woods should always contain some mature habitats, even if they are mainly treated for commercial timber production. This can be achieved by retaining some stands beyond ordinary rotation age (before eventually harvesting them), diversifying the silvicultural system, or retaining a scatter of individual trees amongst the plantations or on ride margins. If there are small areas that are difficult to manage, or which cannot be felled for amenity or safety reasons, then they can usefully be regarded as small minimum intervention reserves, contributing some diversity to the wood as a whole.

Small minimum intervention reserves on ground that could easily be used to grow and harvest timber are hangovers from a period when provision for conservation often meant leaving a small patch of native woodland untouched while the rest of the wood was felled, treated with herbicides and planted with conifers. Today, when Forestry Commission guidance (Forestry

Authority 1994) and incentives enable woods to be retained as native woodland with a wide range of age-classes, it is preferable to adopt long rotations and retain large trees and dead wood in the managed woodland.

7. Research and monitoring

Research, survey, monitoring, demonstration and casual observation are parts of a spectrum of activity, which aims to understand and to transmit that understanding. As far as nature conservation and minimum intervention woodland reserves are concerned, it may be useful to partition this spectrum for planning purposes, but we should not forget that, for example, that survey is a form of research, that some forms of research are tantamount to monitoring, and that casual observations can contribute to monitoring and generate research.

This section aims to demonstrate that minimum intervention woodland reserves, and near-natural woods that probably ought to be reserves, have been used for ecological and conservation research. It reviews the range of work available from old-growth and virgin stands in North America and continental Europe, and equivalent research from relatively natural woods in GB. The special character of research on minimum intervention reserves and their equivalents is considered, and recommendations are made for a minimum baseline record in GB minimum intervention reserves.

Most of the material deals with 'research' in the sense of investigations which have been published in the scientific literature. This restricted view is necessary for the review of published investigations (7.1, 7.2), but the broader view is taken in sections 7.3 and 7.4, where research is just one end of a spectrum which stretches through monitoring to casual observation.

7.1 Research in virgin and old-growth forests in Continental Europe and North America

7.1.1 Forest dynamics and structure

The value for research of minimum intervention reserves is difficult to demonstrate in GB, principally because few, if any, woods are regarded as near-natural. In North America and continental Europe, on the other hand, many old-growth and virgin stands are regarded as near-natural, and these have been used to study natural forest structure and processes, and to understand the changes that have been brought about by forest destruction and management. In fact, many hundreds of papers and books are published annually. Peterken (1996, chapters 3-12) gives a general review and describes many individual studies. The following summary is intended to indicate the variety of published research in natural temperate and boreal forests.

Perhaps the most comprehensive treatment of a single forest is Falinski's (1986) book about Bialowieza Forest. Together with associated publications in the house-journal *Phytocoenosis*, this describes the history of the forest, its composition, structure and patterns, and a description and inventory of its fauna and flora. The forest did not escape human influences in the past, but it is still the most natural mixed deciduous forest left in the north-temperate lowlands.

Although many publications on natural forests deal with particular aspects, it is often difficult to separate these from the forest as a whole. This is never more true of forest structure and disturbances regimes, which are really two sides of one coin. Comprehensive studies of structure, disturbances and composition were made in North America, for example, by Hough and Forbes (1943) in the Tionesta part of the Alleghenies, Heinselmann (1973) in the fire-dominated conifer forests of the Boundary Waters Canoe Area, Lorimer (1980) in the cove forests of the southern Appalachians, Romme (1982) in the sub-montane conifers forests of Yellowstone National Park, and Foster (1988) in the hardwood-conifer transition forests of central New England, all of which afford a profound understanding of how particular types of natural forest work. Historical records of natural forests that have now largely been destroyed have been used to reconstruct disturbance and composition patterns on a regional scale by Grimm (1984) in Minnesota, and Canham and Loucks (1984) in Wisconsin. Lately, landscape-scale dynamics have been modelled on the basis of disturbance regimes observed in surviving old-growth reserves (eg Frelich and Lorimer 1991).

For many years, equivalent studies in Europe have been more classificatory. Stand structures have been classified and mapped, generating diagrams of the patchwork of regenerating, optimal and degenerating phases of growth, eg, the map of Corkova Uvala in the Plitvice Lakes National Park, Croatia by Mayer et al (1980). Lately, however, and particularly in the fire-dominated Boreal forests of the Nordic countries, North America-style studies of disturbance patters and structure have become available, such as Zackrisson (1977) and Engelmark (1984). Emborg et al (1996) recently completed a study of structure and composition of a near-natural beech-ash-elm stand in Denmark, where gaps are generated mainly by wind and disease.

Numerous studies have revealed age-distributions at the stand scale to contain strong even-aged elements (Jones 1945), but a few have found the intimate mixture of all ages that is commonly supposed to be characteristic of natural forests. Early examples from freshly logged mixed broadleaved-conifer forests came from Manchuria (Iwaschkewitsch (1929), Ukraine (Mauve 1931), and Wisconsin (Maissurow 1943), and these demonstrated that even such stands had been disturbed in the remote past. Less destructively, detailed age distributions have come from Appalachian cove forests (Lorimer 1980) and oak-lime stands in Bialowieza Forest (Koop 1989).

Specialised studies of particular types of disturbance have been completed. Perhaps the most widely cited is Sprugel's (1976) account of wave-regenerated balsam fir forests, which undergo a regular short cycle of regeneration and degeneration in the face of annual ice damage. Fire and its immediate impacts have received numerous studies. The pattern of wind damage was studied by Bouchon et al (1973) in the beech-oak near-natural Stand at Fontainebleau. In the southern Appalachians, Clinton et al (1993) studied drought impacts.

Another approach has been to study gap formation, which may be caused by several different types of disturbance in a single site. The classic early study was by Sernander (1936) in a supposedly virgin spruce-pine stand in southern Sweden. Koop and Hilgen (1987) described gaps and gap enlargement in Fontainebleau. Perhaps the most comprehensive of modern studies was by Runkle (1981, 1982) in a range of mixed deciduous forests in the eastern USA, and a follow-up which demonstrated gap enlargement processes (Runkle and Yetter 1985).

In most of these studies, the long-term dynamics of natural forests have generally been reconstructed from current stand structure and direct short-term observations of change. Long-term changes in stand composition, ie over a few thousand years, have been made from sub-fossil remains, notably pollen profiles in small hollows within natural forests. Some have demonstrated very little change, eg in the Rothwald Urwald, Austria (Kral and Mayer 1968). Many have demonstrated substantial change, eg from mixed deciduous to beech and spruce at Siggaboda, southern Sweden (Björkman and Bradshaw 1996), and an eastern hemlock stand in Harvard Forest (Foster and Zebryk 1993). Several have shown a mix of continuity and change, eg in Draved (Iversen 1964) and Bialowieza (Mitchell and Cole 1998). Some have shown that supposedly virgin stands had actually been greatly disturbed by people in the not-too-distant past, eg a north Swedish swamp forest (Segerstrom et al 1994), and a spruce-pine forest in southern Sweden (Bradshaw and Hannon 1992).

The gap between long-term, pollen-based studies is now being bridged by permanent plot studies of change over several decades. Recent examples include studies of a boreal Norway spruce forest (Hofgaard 1993), a boreal pine-spruce forest (Linder 1998), and mixed deciduous stands in Bialowieza (Bernadzki et al 1998). Numerous studies have been published from virgin old-growth in the USA.

Tree responses to disturbances and the effects of disturbance on stand composition take many forms. The immediate responses to stand-destroying disturbances have been followed, eg Petersen et al (1990) following the blowdown in Tionesta Creek. The whole post-disturbance succession was described by means of a chronosequence of stands for the Sitka spruce - western hemlock forests of Alaska by Alaback (1982). Models of long-term succession were developed by Liebundgut (1959) and Koop (1989), based on observations of existing natural forests.

Other approaches to understanding natural forest composition in terms of succession and disturbances have been tried. For example, Hutnik (1952) seems to have been the first to look at the effects of tip-up mounds on regeneration. The micro-pattern of regeneration in small gaps mixed broadleaf forests of the in eastern USA was studied by Sipe and Bazzaz (1995) and Kupfer and Runkle (1996). Ward and Parker (1989) looked at the pattern of regeneration in mixed old-growth. The notion of species alternations was considered for American beech - sugar maple forests by Forcier (1975) and Poulson and Platt (1996). Transition frequencies of species change (ie the influence of canopy composition on regeneration at the individual tree scale) were calculated by Horn (1975) and Runkle (1981), then the long-term implications were computed. The demography of individual tree species seems to have been overlooked, but a few studies are available, eg the study of beech in relation to disturbance by hurricanes by Batista et al (1998).

The dimensions of natural forest emerge in some form from most studies. One of the first to study size-class distributions was Hough (1932), who found the negative exponential distribution which seems characteristic of less-disturbed stands. Jones (1945) included an early review of the tree sizes and stand dimensions, ie the sizes achievable by individual trees, and the volumes attained in old-growth. Quantities of dead wood in natural forests were reviewed by Harmon et al (1986), though this only revealed the paucity of information from natural broadleaved forests.

The dynamics of mortality and decay have been given more attention in the last 20 years than previously. Mortality rates tend to emerge as part of general studies of stand dynamics, eg Parker et al (1985), and were reviewed by Harmon et al (1986). Permanent plot studies have yielded some information on the turnover of snags and fallen wood, eg in the study of the old-growth beech stand of Donaldson's Wood (Lindsey and Schmeltz 1965, Macmillan 1980). The important interactions between fallen logs and woodland watercourses have been discovered (eg Lienkaemper and Swanson 1987).

Many other aspects of natural woodland have been studied. These include specialised forest types, such as floodplain forest (eg Hupp 1988, Putz and Shartiz 1991) and treelines (eg Arno 1984). Open spaces, which are far commoner in coniferous than in broadleaved forests, can be quantified from Vaisanen et al (1986) and several other larger-scale studies, such as Heinselmann (1973). Lutz (1940) described the effects on forest soils of uprooted trees. Beatty and Stone (1986) measured the small-scale variation in soil properties around tip-up mounds. Martin (1992) mentioned 'macropores', the tubes left by decayed major roots, which facilitate drainage in natural forests.

7.1.2 Biodiversity aspects

The biodiversity associated with natural forests has been reported in countless publications. These might be said to range from studies of the social behaviour of wide-ranging, large-territory animals, such as wolves and bears, to inventories of bryophytes or invertebrates. Again, the most comprehensive treatment of an individual forest appears to be the several hundred publications on Bialowieza Forest, of which Falinski (1986) is the main general account. A general review of the dead wood fauna was included in Harmon et al (1986). Heliovaara and Vaiasanen (1984) describe succession within fallen logs. The dynamics of the ground vegetation of American old-growth seems to have been rather neglected, but some accounts are available (eg Bratton 1976, Moore and Venkat 1980) and the particular flora of gaps was described by Thompson (1980). The breadth of the scope of studies can be illustrated by Tomioljc (1991), who described the bird communities of European old-growth and the specialised behaviours of some species within these forests. Carey (1989) reviewed the specialised vertebrates of old-growth forests in the Pacific Northwest, where the Northern Spotted Owl has become a factor in regional and national politics.

7.1.3 Impacts

An understanding of the impacts people have on natural forests emerges from comparisons between natural and managed forests. The 'natural' forests with which managed – usually modern – forests are compared range from those of several thousand years ago to surviving virgin old-growth.

Studies based on pollen and other sub-fossil deposits have revealed long-term change in immense detail, and many of these changes have been ascribed to people. Knowledge of the former states of the forests has enabled existing stands to be identified as relatively undisturbed relicts. In these instances the natural forests at some point in the past are not being directly compared with modern conditions: we do not know what modern natural forests would have been like if they had never been disturbed.

Modern natural forests have been used to study fire histories. These usually reveal that fire frequency and extent have been much reduced in the last century or so. Prior to fire suppression, the pattern of fires is thought to be more natural, though people undoubtedly influenced the incidence of fires

The impacts of logging on natural forests have been directly observed. In fact, they are still happening in Boreal regions and in parts of North America, and in Europe virgin forests were being felled throughout the 19th and 20th centuries. Comparisons are made with surviving natural stands and with records of the logged stands before they were logged. For example, Whitney (1987) used old records to quantify changes in the pattern of forests in Michigan since Europeans arrived in the mid 19th century.

The comparisons that are perhaps most readily appreciated by a modern audience are side-by-side comparisons of managed forest with relict virgin old-growth on the same site type. Many examples are available. Muller (1982) compared composition and size-class distributions in old-growth and 35-year-old second growth stands in mixed deciduous forests.
Robertson et al (1978) compared old-growth and 75-year-old second growth floodplain forests in Illinois. Buchert et al (1997) looked at the effect of logging on genetic diversity of eastern white pine in Canada. Several studies comparing the birds of old-growth and managed forests have been published, eg Whitcomb et al (1981), Helle (1986). A study of the effects of clear-cutting Appalachian old-growth (Duffy and Meier 1992) concluded that there were effectively permanent losses, but this was so controversial that the argument surfaced in the New York Times. These comparisons are only likely to be possible if the natural stands are protected in reserves.

7.2 Research in near-natural woods in Britain

Each region has a distinctive combination of forest types, disturbance regimes and wildlife species. Research in natural forests in one country may thus yield only broad generalisations about the natural forests in another. There remains a need to apply these generalisations to Britain, and to determine any special features. The difficulty is that understanding of the nature of natural woodland in Britain must be pieced together from incomplete, small-scale and generally short-term studies in woods that are no more than semi-natural, and by extrapolation from studies in Continental Europe. Hall et al (1999) recently reviewed the use of minimum intervention reserves for research in Britain.

7.2.1 Natural states and processes

The broad configuration and post-glacial development of original-natural forests has been understood from pollen analysis and sub-fossil remains of trees and woodland organisms in peat. The pattern of forest types 5000 years ago was broadly similar to the present pattern, with mixed deciduous forests giving way in the Highlands to Boreal pine-birch-aspen forests. The balance between species has changed, principally through the reduction of lime and elm, and the increase of beech, ash and pioneer trees, such as birch. Disturbance regimes cannot be so readily determined, but fire was important in the Boreal forests, and wind had some impact on wetland woods.

Disturbance regimes in the inherited-natural woods of today have been studied from a mix of informal observation of events and careful studies of woods in minimum intervention reserves.

The storms of 1968 in Scotland and 1987 in south-east England brought home to ecologists the importance of wind, but there were few studies of their impacts. Detailed study of Lady Park Wood has revealed a mixed disturbance regime in which drought, windthrow, disease, snow and physical instability of steep slopes have all played a part (Peterken & Jones 1987, 1989; Peterken & Mountford 1996, 1998). The pattern of gap formation and the development of stand structure (eg size-class distributions of living trees; accumulation of dead wood) have been quantified, and long-term impacts of particular disturbances on tree populations have been described (drought on beech; disease on wych elm). Similar studies have been undertaken on other minimum intervention reserves (eg Denny, Clairinsh, Rannoch), which collectively help to build up a national pattern of natural forest dynamics.

The responses of species to natural developments and individual events have been studied in various ways. In recent years the broad pattern of genetic variation in oak has been described and interpreted in terms of broad post-glacial movements of forest types (Ferris *et al* 1993). Treelines have changed over the millennia (Pears 1967). Studies of the response of woodpeckers to storm-induced changes in woods (Smith 1994), and the importance of small gaps in the pattern of woodland birds (Fuller 1988), have indicated how wildlife reacts to structural changes in stands. Long-term studies of population changes in mature minimum intervention woodlands have shown how species maintain themselves in near-natural woodland (Perrins 1979). Interpretations of the distribution of saproxylic species in relation to habitat requirements and the history of individual woods have suggested the importance of deadwood and mature timber habitats in natural forests, which are corroborated by rare finds of beetles in sub-fossil wood.

7.2.2 Comparisons with managed land

The use of natural woodland as a reference point for managed woodland has been somewhat limited, perhaps because there are few woods where comparisons are available. Some attempt has been made at Lady Park Wood to understand the impact of thinning in an adjacent stand on grey squirrel damage to beech and the maintenance of diversity in the ground vegetation. At Denny Wood, the impact of inclosure on part of a wood-pasture has provided some information on the long-term effects of pasturage on the stand, but this has been obscured by continued grazing and browsing within the inclosed stand (Mountford *et al* 1999).

There has been a temptation to treat Bialowieza Forest as a substitute for the lost natural forests of Britain. The work of Falinski (1986) and many others is deemed to be relevant to Britain, eg Fuller's (1988) study of birds in gaps. No doubt this is reasonable enough, even though the climate is different and both spruce and pine are natural in Bialowieza, but the temptation highlights the need for natural reference points in Britain.

In Britain, ancient woods provide a surrogate for natural woodland. For example, Mackney (1961) used the soils in ancient woodland as the reference point for a study of podzolisation. However, as the review by Ball and Stevens (1981) indicated, ancient woods are equivocal substitutes for natural forests. Their soils may never have been ploughed and may always have supported native woodland, but the long absence of blowdowns and tip-up mounds has probably rendered the soils un-naturally under-disturbed.

Perhaps the best attitude is to accept that natural woodland cannot be totally natural, but that comparisons between relatively natural woodland and a managed state at least show the scale

and direction of the differences. In that spirit, ancient woodland is useful as a substitute for natural woodland, especially if it is currently allowed to function naturally and has not been managed for some time. Thus, monitoring schemes which include long-unmanaged ancient semi-natural woodland, such as the Common Bird Census (Greenwood *et al* 1995) and the Butterfly Monitoring Scheme (Pollard and Yates 1993), can yield a measure of the effects of management.

7.2.3 Environmental impacts

Natural woods could also provide locations for measuring widescale environmental impacts, such as diffuse pollution, climate change or land drainage. However, in this instance totally natural woodland is not necessary. The main requirement is that the reference site is semi-natural and not currently managed, ie we have a guarantee that there are no on-site sources of pollution or other local impacts, and no influence of silvicultural treatments to confound the comparisons.

Several kinds of impact have been studied in this way. The impact of the arrival of muntjac on the ground vegetation of Monks Wood has been measured in unmanaged stands (Crampton *et al* 1996). Increases of nitrogen and decreases in pH in the soils of Wytham Wood were ascribed mainly to atmospheric pollution, though deer were present and some effect of stand maturation could not be ruled out (Farmer 1995). In the latter instance it is clear that a totally natural stand would have made a better reference point. Impacts of pollution on tree health have used mature, unmanaged semi-natural stands, which would help to detect any interactions with silvicultural treatments (Innes 1993). ITE's Environmental Change Network includes some minimum intervention woodland stands (eg Sykes & Lane 1996; Morecroft *et al* 1998).

7.2.4 Ecosystem recovery

Natural succession after clear-felling or from non-woodland to woodland is a form of ecosystem recovery. Such changes are interesting in themselves, and have absorbed academic ecologists for at least a century, but, if these natural processes are compared with succession after planting, site treatment, or any other management, we have a measure of the ecological impacts of the management. Natural successions running for a decade or less are easily obtained, but natural successions on the timescale of forest processes or a forestry rotation are unlikely to be available without special protection.

Several kinds of recovery have been studied in Britain. For example, soil changes over 100 years have been studied in Broadbalk and Geescroft Wildernesses, Rothampsted (Johnston *et al* 1986) which show the rate and direction of change starting from late 19th century agriculture (which may or may not be the same starting with 21st century agriculture). Peterken and Jones (1989) studied 40 years of stand recovery after felling in a semi-natural wood, and here too the process was partly determined by the particular starting point. Response to the removal of grazing has been studied at the Cairngorms treeline (French *et al* 1997) and in upland alder woodland (Latham and Blackstock 1998).

7.3 Some general features of research on minimum intervention woodland reserves

In GB minimum intervention woodland reserves may not yet be near-natural, so their current value for research into natural processes may be limited. Furthermore, their degree of naturalness may not be much greater than that of other ancient semi-natural woodland. As the years pass, however, the reserves will become more natural and thus more distinct.

Implications: Research in GB minimum intervention reserves may not be so concentrated on natural processes for the time being. Instead, we may still be studying the interaction between historical states and natural processes. We cannot use 'woodland archaeology' to give us clues about natural processes, as the Scandinavians can on fire histories, neither can we use old-growth to look at past disturbance patterns and gap processes. Note that there will be a difference in quality between the research that is possible in eastern USA and some parts of continental Europe. For the time being research that might be done on minimum intervention reserves in GB may just as well be done on other ancient semi-natural woodland, but as time passes minimum intervention reserves will become more valuable.

Since minimum intervention reserves cannot completely exclude the wider impacts, any research in minimum intervention reserves will probably find itself dealing with natural processes and human influences together. An example would be the impact of deer in minimum intervention reserves, which is interpreted as both natural change and change induced by excessively high deer populations. Another would be soil changes, which may be due to natural processes or widescale nitrogen rain. The study of the Black Wood of Rannoch by Peterken and Stace (1987) covers both ecosystem recovery after grazing and disturbance impacts, and natural mortality rates of mature pine.

Implications: We observe and interpret. But we may need experimental work elsewhere to separate natural from man-made factors. It may actually be impossible to study some natural processes if there are no places free from human influences, eg long-term soil maturation in an environment of acid rain, nitrate rain.

Minimum intervention reserves cannot be used for experimental research that leaves a significant mark on the wood, for the ecologist will become an ecological factor. Thus, for example, the studies by Henry and Swan (1974) at Pisgah Forest and Oliver and Stephens (1977) at Harvard Forest could not be allowed. Research on a small scale (eg soil pits), or taking only small samples (eg genetic characterisation) can be permitted, but otherwise research tends to be observational.

Implications: Some small-scale experimental research on, say, insect populations or bryophyte dynamics, might be allowed if it did not leave a significant mark. However, minimum intervention reserves would not be large enough to, say, set up experiments in artificial gap formation to an approved experimental design: this would have to be done in mature ancient semi-natural woodland elsewhere. Stand-scale research tends to be a matter of observing what happens, interpreting and working with imperfect experimental designs.