The influence of some forest operations on the sustainable management of forest soils—a review

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Summary

This review paper describes the nature and scale of changes to forest soils brought about by forestry operations. A relatively non-technical approach is adopted with the aim of stimulating debate within as wide an audience as possible. The paper does not aim to be exhaustive but rather a position statement. Areas where further study is required are highlighted.

The concept of sustainability is explored in relation to forest soils, and the condition highlighted is that impacts of forest management operations should not, in the long term, exceed the capacity of soil to recover by natural processes (e.g. erosion losses should not exceed soil formation rates, nutrient removals should not exceed nutrient inputs etc.). Soil erosion, nutrient removal, compaction, and changes in organic matter content and soil water status are identified as the most important processes involved in the impacts of management.

The impacts of some of the more intensive forest management regimes on soil compaction, nutrient removal and erosion rates appear to be of similar magnitude to the recovery capacity of soils. Where the most intensive forms of forest operation are used on susceptible sites some degree of long-term soil degradation appears to be likely, and it can be regarded as valid to describe such management practices as unsustainable. However, the scale of occurrence of such management is probably relatively modest, and decreasing. On less susceptible sites, and where less intensive forms of management are employed, impacts on soils are low enough for management to be regarded as sustainable, and are often less than under pre-existing land uses. Compaction caused by heavy harvesting and extraction machinery, nutrient depletion resulting from whole tree harvesting on infertile sites where rotations are short, and erosion following cultivation and harvesting on erodible soils are the greatest causes of concern. Compliance with recent Forestry Commission guidelines should lead to lower impacts than those recorded during recent decades. However, rotation-length audits of the impacts of different forest management regimes on a range of site types are needed before definitive statements about the sustainability of management operations can be made.
Introduction

Soils are clearly of such fundamental importance to all land uses, that a first step in determining what constitutes sustainable land management is to consider the role of soils. This paper collates information on the long-term effects of current forest management practices on soils, with the aim of contributing to the process of defining sustainable management in the context of forests and forestry.

Forest management regimes consist of a series of individual operations many of which potentially affect soils, either adversely or beneficially. The overall aim is to identify the processes by which soils are changed by forest operations and to present and interpret information on the scale and nature of these changes. A relatively non-technical approach is adopted in the paper with the aim of engaging as wide an audience as possible. It should be noted that this paper is confined to considering the direct effects of operations on soils, rather than the wider influence of tree cover per se.

The paper begins by discussing the concept of sustainability in relation to forest soils and forestry operations. In the subsequent section the main processes by which soils are influenced by management practices are identified and, where possible, information is presented on the scale of management-induced changes. The question of whether forest soils have been changed by forest practices to such an extent that tree growth has been affected is then discussed on the basis of evidence from abroad. The concluding section highlights both the processes and operations which are most likely to cause long-term changes to soils and which may, therefore, have consequences for sustainable management.

Sustainability

The idea that it is a duty to conserve soil fertility has a long history; statements to this effect appear in the Bible (Num. 35: 34), the works of the ancient Greeks, as well as in the writings of more recent philosophers (Leopold, 1949; Passmore, 1974). History provides us with examples of soil fertility having been conserved for long spans of time but there are also many examples running right up to the present where soils have been seriously damaged. These latter serve to remind us that, despite there being a strong moral and practical case for conserving soil fertility, mankind, either through ignorance or greed, can achieve the opposite. The Brundtland report defines sustainable development in terms of inter-generational equity as 'ensuring that [humanity] meets the need of the present without compromising the ability of future generations to meet their own' (World Commission on Environment and Development, 1987).

Sustainability and the nature of environmental impacts

Among the different ways in which the term sustainability is used there are some common themes which relate to the environmental impact of management operations. An impact is the change in the status of a soil which occurs as the result of a management operation. Impacts frequently manifest themselves as changes in the rates of action of naturally occurring processes such as erosion and nutrient loss. Impacts are balanced by countervailing natural processes such as soil formation and nutrient input from weathering which act to renew or repair soils. A basic requirement for sustainable management is that the impacts of management operations should not exceed the natural capacity of sites to renew or repair themselves.

The following questions need to be asked in order to make a basic assessment of whether an operation constitutes sustainable management of soil:

1. What is the magnitude of the impact, is it desirable or undesirable, and what is the geographical scale (extent) and time period over which it manifests itself?
2. What is the significance of the impact; i.e. does it exceed the natural capacity of the site to repair itself, and if so by how much?
3. To what extent is the impact reversible?
4. What are the long-term economic benefits and costs for present and future generations associated with the operation?

In addition it is clear that management operations have considerable and complex off-site
impacts which need to be taken into account in a comprehensive assessment of sustainability (Clayton and Radcliffe, 1996). These include the indirect impacts of operations themselves (e.g. effects of sedimentation on aquatic fauna) and wider impacts resulting from the consumption of energy and raw materials needed to carry them out. This paper however concentrates solely on the long-term impacts of management operations on forest soils and their implications for sustainability.

**Magnitude of impacts** Operations giving rise to large negative impacts are more likely to exceed the natural capacity for renewal than those giving rise to small ones. The magnitude of the change is measured as the difference between the soil status under a new management regime as opposed to an old one. Thus impacts are always assessed relative to a pre-existing baseline state. Different types of baseline have been used to assess the magnitude of impacts, depending on the context of the study (see Figure 1). Three baselines have been used in Scotland, namely:

1. The status of the soil under pre-existing land uses such as moorland and hill grazing. This has been used as a baseline when assessing the impacts of afforestation practices (e.g. erosion rates; Soutar, 1989; Moffat, 1988).

2. The status of soils under first rotation forest plantations. This has been used when assessing the impact of harvesting and establishment (e.g. changes in nutrient status; Goulding and Stevens, 1988).

3. The status of the soils under (semi-) natural forest. This has been used to assess the

![Figure 1. Indices for assessing the magnitude and significance of environmental impacts of forest operations.](https://academic.oup.com/forestry/article-abstract/70/1/61/541168)

1. Comparison with pre-existing land use. May be positive or negative depending on the impact considered and the intensities of management. 2. Comparison with other land uses. Used to rank different land use according to impact (e.g. erosion rates, nutrient outputs). 3. Comparison with semi-natural woodland. Gives best assessment of ecological impacts. 4. Long-term net impact. Determined by magnitude of the impact and the capacity of the ecosystem for self-renewal. 5. Impact of deforestation of native woodland. Indices 4 and 3 are the most useful for assessing sustainability.
ecological impacts of forest management (e.g. species composition; Miles, 1985) or of deforestation (Malcolm, 1957; Dimbleby, 1976). This baseline is rarely used in Scotland because few areas of semi-natural forest remain. It is more frequently used in North America (e.g. Kimmins, 1990b).

The use of different baselines can cause difficulties in interpreting the significance of impacts, and their consequences for sustainability.

Desirability of impacts Impacts may be beneficial or adverse. Characterizing impacts in this way requires assessing whether the impact has changed soil conditions towards or away from an ideal state. While this may be fairly self-evident in the case of some impacts (e.g. increased erosion), for others it is not so clear (e.g. increasing podsolization). Although scientific data may exist to describe the magnitude of such a change, different individuals and interest groups may hold divergent views on what constitutes the ideal state. As a rule, impacts are considered as beneficial when:

- the degree of naturalness of an ecosystem or landscape is increased or artificiality is reduced; or
- valuable aspects of managed ecosystems are enhanced (e.g. productivity).

These two precepts can clearly come into conflict, and decisions about how to characterize impacts may become subject to value judgements.

Geographical extent of impacts Operations which impact over a wide geographical scale are more likely to be considered unsustainable than those that are of restricted scale. For example, road building typically affects only about 1 per cent of the soil surface of forests and therefore attracts less attention than harvesting which may affect 5 per cent to 69 per cent of clearfelled areas (Bockheim et al., 1975).

Time The question of time is crucial and complex, because ultimately it is the cumulative effects of environmental impacts over an extended time period which determine the sustainability or otherwise of an operation. Impacts which continue over long periods are more likely to exceed the capacity of the site for self-renewal than short lived ones—all other things being equal. Current measurements of the magnitude of impacts are almost exclusively short term; relating to the effects of single operations over short periods. Impacts need to be aggregated over at least one rotation length before meaningful statements regarding the sustainability of management practices can be made (Figure 1).

Assessing the significance of impacts Many authors have contented themselves with presenting evidence on the magnitude of impacts and (understandably) have neglected the more difficult task of assessing their significance. Three approaches have been used in assessing the significance of impacts of forest operations, namely:

1 Comparing them with impacts observed under agricultural land uses. For example, Moffat (1988) reported that erosion rates under forestry were of a similar magnitude to those observed in catchments of mixed agricultural land use, and Miller and Miller (1991) observed that nutrient removals due to forestry were lower than those under most forms of agriculture. Such comparisons can be used to rank land uses in respect of their impacts, but are only of limited assistance in assessing sustainability.

2 Comparing them with impacts observed in (semi-) natural forest processes (e.g. Kimmins, 1990b). Management operations such as felling and site preparation have impacts which are similar in certain respects to natural disturbance in forests; for example both processes cause erosion and changes in nutrient flux.

3 Comparing them with the countervailing processes which renew/repair forest soils. This provides information on whether a net change in soil status can be expected through time. For example, rates of soil erosion can be compared with rates of soil formation; nutrient removal in harvested timber can be compared with inputs from weathering and precipitation; and compaction can be assessed
against the capacity of soils to recover by shrinkage and swelling.

Of these approaches, the latter two are most useful, but the information necessary to make comparisons is only slowly becoming available.

Problems of changes in baselines through time

Problems in assessing both the magnitude and significance of impacts are encountered because the baselines against which impacts are assessed are rarely stable themselves, but alter through time as the result of natural processes. Over extended time periods forest soils evolve as a result of processes such as leaching, weathering and nutrient capture from the atmosphere by the canopy. The soils present today are only a snapshot in a continuing evolutionary sequence (White, 1979)—'a space-time continuum forming the upper part of the earth's crust' (Fitzpatrick, 1980). We should not therefore expect sustainably managed ecosystems to be absolutely changeless or stable. On the contrary we should expect and allow ecosystems to be able to respond to natural changes in the environment. One of the challenges of sustainable management is designing them so that they can respond.

On a relatively short time scale, forest ecosystems undergo cyclical changes through disturbance by wind, fire or insect attack, or as the result of immobilization of nutrients in trees, ground vegetation and organic horizons. Rates of erosion or soil nutrient levels, for example, will vary widely according to the life cycle stage of the forest. It is important to bear these changes in mind when assessing the significance of the impacts of forest operations. Ultimately this leads to the need to aggregate impacts of forest operations over at least one rotation.

The effects of man

In addition to natural changes, most, if not all forest soils in Scotland have been affected by the activities of man throughout the last 6000 years. Deforestation has increased the amount of water reaching soils, leading to increased runoff, waterlogging, leaching, weathering and probably erosion (Pyatt and Craven, 1979). Colonization by moorland vegetation has accentuated podsolization and acidification on many sites, and has led to the formation of extensive areas of iron-podsol soils (Dimbleby, 1976). Cultivation and drainage for both agriculture and forestry have disturbed soil profiles over significant areas and have often, at least temporarily, increased erosion. The establishment of conifer plantations accelerates podsolization particularly on sites where natural processes would have led to the formation of brown forest soils under broadleaved tree cover (Grieve, 1978); though the presence of trees has also returned the water status of reforested soils towards a more natural state. Increased input of chemicals in precipitation has provided elevated inputs of some nutrients and accelerated acidification and loss of base cations on some sites.

It is against this background of extensive disturbance, and in many cases degradation, that the future impacts of forest operations needs to be set. In many instances the requirement that the capacity of soils should not be degraded is a minimum, and sustainable management should often be centred on restoring soils.

The degree to which forest soils can be restored following degradation depends on the reversibility of individual impacts (see Figure 2). Negative impacts that are irreversible in the foreseeable future clearly reduce the options available for future generations and therefore can be considered as unsustainable. The accelerated erosion that occurred in some Mediterranean countries following deforestation is a case in point. Many impacts are however reversible to a greater or lesser degree either by instituting less intensive management and letting natural recovery processes act over long periods, or by ameliorative treatments. It can be argued that management operations which cause impacts can nevertheless be regarded as sustainable provided that provision is made for ecosystems to repair themselves. This raises the possibility of systems similar to 'rotational cropping' being used in forestry.

Assessing sustainability

There are three main aspects to the difficult concept of determining whether an operation is unsustainable or otherwise, namely:

1 Which components/outputs of a forest do we wish to sustain?
<table>
<thead>
<tr>
<th>Time profile</th>
<th>Degree of reversibility</th>
<th>Examples</th>
</tr>
</thead>
<tbody>
<tr>
<td>Progressive deterioration</td>
<td>Low</td>
<td>Accelerated erosion as a consequence of deforestation</td>
</tr>
<tr>
<td></td>
<td>High</td>
<td>Loss of nutrients by harvesting combined with windrowing or litter-raking exceeding nutrient inputs</td>
</tr>
<tr>
<td>Deterioration reaches new steady state</td>
<td>Low</td>
<td>Severely compacted soils</td>
</tr>
<tr>
<td></td>
<td>High</td>
<td>Increased wetness of soils following deforestation (in Scotland)</td>
</tr>
<tr>
<td>Deterioration is short-lived and repaired by natural processes</td>
<td></td>
<td>Increased leaching loss of nutrients from carefully harvested sites, or sites disturbed by natural agencies</td>
</tr>
</tbody>
</table>

Figure 2. A classification of nature of environmental impacts with respect to their development through time and reversibility.

2 How serious (significant) are the impacts?
3 How much account do we need to take of any benefits which may accrue from the operation?

Forest soils perform a number of functions important to mankind. These include:
• medium for tree growth;
• medium for growth of forest plants and fungi;
• habitat for invertebrates, part of habitat for some vertebrates;
• medium for cycling or organic matter and nutrients;
• regulation of hydrology and chemistry of watercourses;
• provision of sink/buffer for atmospheric deposits of chemicals.

In this paper it is assumed that all the outputs derived from forests and forest soils are valued by society and potentially require to be sustained to varying degrees. The concept of sustainability does not however help us to determine the relative importance of, for example, timber production as opposed to biodiversity. It does, however, impose limits such that changes should not put at risk the natural processes that support production. By doing so it suggests a change in paradigm away from treating forests as crops and towards forms of management that treat forests as natural ecosystems.

All management operations have an impact on ecosystems. As Lanly (1992) states: ‘it must be accepted from the start that utilization of a given forest ecosystem implies some change in its structure and composition, and that sustainability cannot mean the identical reproduction of the ecosystem in its original state.’ Impacts which exceed the long-term capacity of sites to renew themselves will lead to progressive site degradation. After a shorter or longer period a new steady state will probably be reached at a
lower level of site productivity (Figures 2 and 3). Sustainability can be conceived of in a narrow sense in which all such reductions in productivity (i.e. reductions in natural capital) are regarded as evidence of unsustainable management. Alternatively a wider definition can be used in which losses of natural capital are tolerated provided they lead to commensurate gains in man-made capital (see Pearce et al., 1989).

Balancing impacts and benefits The assessment of sustainability therefore frequently needs to include consideration of whether the benefits derived from the operations are sufficient to compensate present and future generations for any negative impacts. Even large environmental impacts may be tolerated by people if the benefits they receive are sufficiently attractive. Resolving such questions is fundamentally an exercise in environmental economics. Trying to value costs and benefits on behalf of future generations poses enormous problems. However it is clear that to be regarded as sustainable, forests operations must, at the very least:

- not impair the capability of forest soils to produce a wide range of goods and benefits in the foreseeable future; and,
- maintain the options to alter the mix of benefits since those required by future generations may be different from the ones preferred by our own.

Further problems are posed by the need to determine the weighting of interests of current as against future generations. While acknowledging this wider aspect of sustainability, this paper is restricted to reviewing information on the nature (magnitude, scale, duration and time profile and reversibility) of environmental impacts. Beyond this, decisions pass out of the hands of soil scientists, conservationists and foresters, into the domain of economists and politicians.

Figure 3. Possible scenarios for the development of long-term environmental impacts. Most forms of progressive deterioration would eventually attain a new steady state.
Impacts of management operations on forest soils

Since the inception of modern forestry practices in Europe last century, rapid advances have been made in management practices aimed at increasing timber production. Several sources of concern about the sustainability of such practices have arisen. The earliest of these centred on the possibility of alteration to forest soils in Germany leading to reduced timber production (Grigor, 1868; Wiedemann, 1923, 1935). During recent decades off-site impacts, particularly on watercourses, have become the focus of attention (e.g. Maitland et al., 1990). Concern has also been expressed over potential long-term impacts on the capacity of forest soils to provide benefits such as biodiversity (e.g. The Helsinki Guidelines).

Changes to forest soils as a result of forestry can be subdivided into those which result directly from forest management operations (e.g. erosion following cultivation or harvesting) and those due to the influence of the tree crop (e.g. podsolisation as a result of replacement of broadleaves with conifers). This paper concentrates on the impacts of management operations on the physical and chemical characteristics of forest soils and considers the processes causing and repairing such impacts. It then goes on to review the effects these may have on tree growth and soil biota. Only on-site impacts are considered in detail; the important off-site effects having been reviewed elsewhere (Ormerod et al., 1987; Maitland et al., 1990; Nisbet, 1990).

The main impacts of forest management operations on forest soils are:

1. The erosion of soil exposed by cultivation and harvesting.
2. Changes to the nutrient status of soils resulting from harvesting, including both the export of nutrients from the site and harvest-induced leaching losses.
3. Compaction of soil mainly as a result of the use of machines.
4. Changes to the quantity and quality of organic matter.
5. Changes in water status and aeration.
6. Physical disturbance to soil profiles during cultivation and drainage.

Erosion

Recent literature on erosion in British forests provides a good example of approaches towards, and difficulties in, interpreting the long-term environmental impacts of forestry practices.

It is now accepted that ground preparation, harvesting and road building on upland sites frequently cause rates of erosion and consequent sedimentation of watercourses to increase above levels recorded under pre-existing land uses (see Table 1 and reviews by Moffat (1988) and (1991) and Soutar (1989)). Sediment yields on afforested catchments are reported as being between 200 and 1300 kg ha$^{-1}$ a$^{-1}$ and are generally between 1.2 and 4 times higher than those reported on catchments that had not been ploughed and drained (Moffat, 1988; Soutar, 1989). Values of erosion measured directly on ploughed land (rather than as sediment in watercourses) are reported to be significantly higher than this; for example Carling et al. (1993) recorded values of 1.4-6.4 tonnes per 200 m length of plough furrow immediately after ploughing. Erosion rates usually diminish rapidly after ground preparation has taken place, but may persist longer into the rotation where drains continue to erode. Erosion limited to short periods can be significant: for example, during a 5-year period following pre-afforestation drainage in the Coalburn catchment, sediment yields were equivalent to nearly half a century's loading at pre-drainage rates (Leeks and Roberts, 1987), though by canopy closure they had diminished to close to pre-afforestation levels (Robinson, personal communication).

Assessing the significance of erosion  The significance of erosion has been assessed in the past by comparing published rates of erosion on forested land with:

1. Rates of erosion on unafforested catchments. On this basis Soutar (1989) reported three-
Table 1: Sediment yields associated with forestry in upland Britain (after Soutar, 1989)

<table>
<thead>
<tr>
<th>Catchment</th>
<th>Afforested or unafforested</th>
<th>Sediment yield as bedload or settled sediments (kg ha(^{-1}) a(^{-1}))</th>
<th>Sediment yield as suspended sediments (kg ha(^{-1}) a(^{-1}))</th>
<th>Total sediment yield if measured (kg ha(^{-1}) a(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>Llanbrynmair A</td>
<td>Unafforested</td>
<td>Not measured</td>
<td>37</td>
<td>Not measured</td>
</tr>
<tr>
<td></td>
<td>After ploughing</td>
<td></td>
<td>90</td>
<td>Not measured</td>
</tr>
<tr>
<td>Llanbrynmair B</td>
<td>Unafforested</td>
<td>Not measured</td>
<td>7</td>
<td>Not measured</td>
</tr>
<tr>
<td></td>
<td>After ploughing</td>
<td></td>
<td>31</td>
<td>Not measured</td>
</tr>
<tr>
<td>Hore</td>
<td>Mature plantation</td>
<td>118</td>
<td>244</td>
<td>362</td>
</tr>
<tr>
<td></td>
<td>After felling</td>
<td></td>
<td>571</td>
<td>Not measured</td>
</tr>
<tr>
<td>Hafren</td>
<td>Mature plantation</td>
<td>Not measured</td>
<td>353</td>
<td>Not measured</td>
</tr>
<tr>
<td>Tanyllywyth</td>
<td>Mature plantation</td>
<td>384</td>
<td>121</td>
<td>505</td>
</tr>
<tr>
<td>Cyff</td>
<td>Unafforested</td>
<td>64</td>
<td>61</td>
<td>125</td>
</tr>
<tr>
<td>Coalburn</td>
<td>Unafforested</td>
<td>Not measured</td>
<td>30</td>
<td>Not measured</td>
</tr>
<tr>
<td></td>
<td>Average yield</td>
<td></td>
<td>240</td>
<td>Not measured</td>
</tr>
<tr>
<td></td>
<td>for first five years</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>after ploughing</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Balquhidder</td>
<td>Unafforested</td>
<td>1</td>
<td>380</td>
<td>381</td>
</tr>
<tr>
<td>Monachyle</td>
<td>moorland</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Baquhidder</td>
<td>Afforested</td>
<td>21</td>
<td>1310</td>
<td>1331</td>
</tr>
<tr>
<td>Kirkton</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Holemstyles</td>
<td>Unafforested</td>
<td>Not measured</td>
<td>32</td>
<td>Not measured</td>
</tr>
<tr>
<td></td>
<td>1st year after ploughing</td>
<td></td>
<td>513</td>
<td>Not measured</td>
</tr>
</tbody>
</table>

Four-fold increases in long-term sedimentation rates, which he implied were sufficiently high to be unacceptable.

1. Rates of soil formation. This provides information on whether or not net long-term loss of soil material could be anticipated and therefore whether current management practices could be sustained.

2. Rates of erosion under natural forest. This could highlight long-term differences in soil volume under plantations compared with that under natural forest cover and therefore provide additional information on the long-term ecological impacts of current practices.

Information on both 1 and 2 above is very scant and general.

It also needs to be borne in mind that erosion is generally localized, so that even if rates of erosion remained below soil formation rates, localized damage could still occur (Pyatt, personal communication).
Rates of soil formation Rates of soil formation from non-calcareous rocks have been estimated to be between 20 and 1900 kg ha\(^{-1}\) a\(^{-1}\) (Alexander, 1988); a range of values similar to the observed rates of erosion (sedimentation) in some mature British forests in the uplands (200–1300 kg ha\(^{-1}\) a\(^{-1}\)). Alexander reports two estimates of soil formation rates from Wales of 230 and 310 kg ha\(^{-1}\) a\(^{-1}\). These are rather lower than the two available estimates of total sediment yield under mature plantation in Wales of 362 and 505 kg ha\(^{-1}\) a\(^{-1}\). These latter values relate to the Tanllwyth and Hore catchments which were afforested using particularly poor silvicultural techniques, and Soutar’s data on suspended sediment yields suggest that total sediment yield in more recently afforested catchments, such as Llanbrynmair, is below 200–300 kg ha\(^{-1}\) a\(^{-1}\) (see Table 1). The highest value for soil formation rate quoted by Alexander (1900 kg ha\(^{-1}\) a\(^{-1}\)) was reported from a moorland catchment in Scotland, and in this case is higher than the estimate of erosion under mature plantation of 1310 kg ha\(^{-1}\) a\(^{-1}\). This limited information leaves us in the tantalizing position of knowing that the balance between erosion and formation of soils may be fairly close on susceptible sites where ploughing is used.

It should be noted that current Forestry Commission Guidelines (Forestry Commission, 1993) discourage the use of ploughing, recommending instead the use of lower impact cultivation techniques. This should significantly reduce erosion on most sites. More site specific information is required before firm conclusions can be drawn.

In respect of harvesting, Powers et al. (1990) reported that erosion resulting from logging in the US mostly falls within the limits of the rate of soil formation, though McColl and Grigal (1979) estimated soil losses from erosion for the first 5–10 years after logging and roading in the Pacific Northwest US ranged between 1000 and 5000 kg ha\(^{-1}\) a\(^{-1}\).

Erosion levels in natural forest Erosion levels in natural forest are frequently described as being low (e.g. Pritchett, 1979) but, hard data against which to compare erosion levels in managed forests, are apparently absent. Boardmann (1987) states that erosion in woodlands is ‘much lower’ than the average value for mixed land-use of 500 kg ha\(^{-1}\) a\(^{-1}\). Soil erosion rates under plantations could be expected to exceed those recorded under natural forests due to the need for ground preparation. The difference will depend on the intensity of management, particularly:

- the rotation length, relative to the natural disturbance cycle;
- the amount of soil eroded during site preparation for afforestation and subsequent cycles of harvesting and restocking, relative to that eroding as a result of natural disturbance;
- the extent of roading.

Impacts on forest soils will clearly be minimized where management operations are designed to mimic natural processes; for example, the use of mounding to mimic ground disturbance following windblow.

*The effects of forest operations* Operations that cause the exposure of soil and provide obvious drainage channels will contribute most to erosion. Worrell (1996) ranked cultivation techniques in Scotland according to the proportion of ground area disturbed by excavation or spoil as follows: ploughing 44–60 per cent, dolloping 29–31 per cent, mounding 26–35 per cent, disc trench scarifying 20–32 per cent, patch scarifying 14 per cent, hand turfing 4–7 per cent and hand screefing 1–2 per cent. Tine ploughing is thought to be potentially more susceptible to erosion than ploughing without a tine, and mounding can lead to erosion on the same scale as ploughing on some sites (Pyatt personal communication). Erosion resulting from drainage is largely determined by the frequency, slope and length of drains. Current guidelines are designed to reduce erosion from drains to a minimum (Pyatt, 1990).

Felling operations will cause least ground disturbance when based on motor manual methods rather than on processors or harvesting machines which can cause rutting. Extraction operations can be ranked in order of decreasing impact: whole tree skidding, tree length skidding/shortwood forwarding, tree length cable crane, shortwood cable crane. When using wheeled or belted machines, the degree of erosion will depend on how well brash mats are used and on the topography and rainfall.
There are currently no published studies in the UK which allow quantification of the erosion rates attributable to different operations. The degree to which recently introduced water quality guidelines will reduce erosion rates from upland forests is also unknown. Further research is required from which rotation-length audits of erosion rates under different types of forest management could be developed. Data on erosion rates under undisturbed and semi-natural forests needs to be collated, together with estimates of soil formation rates on a range of sites.

**Nutrient removal**

Forest operations affect the nutrient status of soils in forests in several ways, the most important of which are:

1. The removal of nutrients in harvested timber (and branches and leaves if whole tree harvesting is employed).
2. Harvesting and cultivation leading to disturbance of the organic horizons, with subsequent increases in mineralization and leaching losses of some nutrients.
3. Fertilizing which increases uptake of nutrients into tree crops, and, in the case of P, may sometimes lead to increased soil nutrient status.

Concern over the nutrient status of soils in managed forests has centred on whether harvesting losses can be replenished by natural processes or cause progressive nutrient depletion leaving forest managers increasingly dependent on fertilizing. Assessing this requires estimating the different inputs and outputs in the form of a nutrient budget (Figure 4).

**Nutrient budgets**

The nutrient status of forest soils will depend largely on:

1. The rate at which nutrients are made available from weathering of soil minerals and from precipitation/deposition.
2. The rate of loss in water draining from the site.
3. Removal of nutrients in forest produce.
4. How efficiently nutrients are retained and cycled within the forest ecosystem.
5. Inputs from fertilizers.

Examples of approaches to nutrient budget studies are shown in Tables 2–4. Table 2 shows a comparison of inputs from precipitation relative to estimated losses caused by conventional harvesting (CH) and whole tree harvesting (WTH) in Britain. Because inputs generally exceed outputs Miller and Miller (1991) conclude that there are no grounds for concern with CH, and WTH might only cause nutrient depletion on the most infertile sites. It should be noted that the balance for phosphate (P) is close under CH regimes and is negative under WTH. Similar studies from Canada (e.g. Boyle and Ek, 1972) and Scandinavia (e.g. Nykvist, 1974) also concluded that nutrient losses were outweighed by inputs from precipitation.

![Figure 4. Factors affecting the nutrient status of forest soils (major influences in capitals).](https://academic.oup.com/forestry/article-abstract/70/1/61/541168)
Table 2: Comparison of nutrient input in bulk precipitation with removal as a result of harvesting (after Miller and Miller, 1991)

<table>
<thead>
<tr>
<th></th>
<th>N</th>
<th>P</th>
<th>K</th>
</tr>
</thead>
<tbody>
<tr>
<td>Input in precipitation (50 years)</td>
<td>150-1250</td>
<td>1-25</td>
<td>50-1000</td>
</tr>
<tr>
<td>Removal from site—conventional harvesting</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sitka spruce at 50 years</td>
<td>149</td>
<td>17.3</td>
<td>92</td>
</tr>
<tr>
<td>Norway spruce</td>
<td>141</td>
<td>18.3</td>
<td>100</td>
</tr>
<tr>
<td>Corsican pine</td>
<td>120</td>
<td>12.0</td>
<td>66</td>
</tr>
<tr>
<td>Scots pine</td>
<td>72</td>
<td>9.6</td>
<td>55</td>
</tr>
<tr>
<td>Removal from site—whole tree harvesting</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sitka spruce at 50 years</td>
<td>498</td>
<td>47.1</td>
<td>222</td>
</tr>
<tr>
<td>Norway spruce</td>
<td>498</td>
<td>46.7</td>
<td>200</td>
</tr>
<tr>
<td>Corsican pine</td>
<td>290</td>
<td>32.0</td>
<td>173</td>
</tr>
<tr>
<td>Scots pine</td>
<td>248</td>
<td>25.0</td>
<td>118</td>
</tr>
</tbody>
</table>

Table 3: Nutrient balance sheet for a 50-year Sitka spruce crop rotation at Beddgelert Forest. Phosphate concentrations in the stream were below detection limit. n.d. = not determined. Additional losses from site in stream at harvest are assumed to be the same for both felling practices. Units kg ha⁻¹ (after Stevens et al., 1988)

<table>
<thead>
<tr>
<th></th>
<th>N</th>
<th>P</th>
<th>K</th>
<th>Ca</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nutrient input in bulk precipitation</td>
<td>520</td>
<td>7.5</td>
<td>223</td>
<td>375</td>
</tr>
<tr>
<td>Nutrient output in stream</td>
<td>678</td>
<td>0.0</td>
<td>215</td>
<td>1 035</td>
</tr>
<tr>
<td>Nutrient accumulation in crop—all above-ground material</td>
<td>428</td>
<td>43.5</td>
<td>144</td>
<td>279</td>
</tr>
<tr>
<td>Nutrient accumulation in crop—stem only</td>
<td>128</td>
<td>12.3</td>
<td>38</td>
<td>151</td>
</tr>
<tr>
<td>Additional loss for site in stream at harvest</td>
<td>126</td>
<td>0.0</td>
<td>19</td>
<td>0</td>
</tr>
<tr>
<td>Overall loss from site—conventional felling</td>
<td>412</td>
<td>36.0</td>
<td>155</td>
<td>939</td>
</tr>
<tr>
<td>Overall loss from site—whole-tree harvest</td>
<td>712</td>
<td>4.8</td>
<td>49</td>
<td>811</td>
</tr>
<tr>
<td>Extractable nutrients in soil to rooting depth</td>
<td>n.d.</td>
<td>24.6</td>
<td>94</td>
<td>112</td>
</tr>
<tr>
<td>Total nutrients in soil to rooting depth</td>
<td>9 400</td>
<td>2 940</td>
<td>59 700</td>
<td>1 390</td>
</tr>
</tbody>
</table>

Table 3 shows a nutrient balance sheet for a Sitka spruce plantation in Wales. Net losses of nutrients were recorded which were significant compared with soil reserves. Inputs from precipitation (in this case excluding direct deposition/dust) were apparently balanced by losses in drainage water. On this basis Stevens et al. (1988) concluded that the potential for nutrient depletion exists, both for CH and WTH.

Federer et al. (1989) summarized nutrient budget information for six sites in the eastern US (Table 4) and extrapolated the results over a 120-year period assuming zero harvesting, one harvest and three harvests. They concluded that outputs in drainage water and harvested material (under WTH) exceeded inputs from precipitation, except in the case of N when the rotation length was 120 years. Calcium was the nutrient showing the highest net losses and these were considerable even in the absence of harvesting. They also concluded (though tentatively) that dry deposition, weathering and root-zone deepening were insufficient to make up for the recorded shortfalls in calcium.

It is apparent that research into nutrient budgets is at a relatively early stage and is providing contradictory results. Many of the difficulties in interpreting such studies arise from differences in the components measured. However some general conclusions can be drawn:
1 An 'optimistic' school of thought can be identified which contends that because inputs from precipitation exceed harvesting output there is no danger of nutrient depletion. This can be contrasted with a 'pessimistic' school which concludes that nutrient depletion is a potential hazard because high leaching losses together with harvesting losses exceed inputs from bulk precipitation.

2 Losses due to harvesting of timber are often small compared with 'total' or 'available' soil nutrient reserves, suggesting that even if net losses of nutrients occurred, depletion would take many rotations to manifest itself. This, however, was not the case in some studies in Wales (Stevens et al., 1988; Goulding and Stevens, 1988), or in respect of Ca in north American studies (Federer et al., 1989). It

### Table 4: Nutrient budget for whole tree harvesting in six US forests (after Federer et al., 1989)

<table>
<thead>
<tr>
<th>Input/output (kg ha(^{-1}) yr(^{-1}))</th>
<th>No Pool change over 120 years (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Annual Harvest</td>
</tr>
<tr>
<td></td>
<td>Annual output</td>
</tr>
<tr>
<td>Nitrogen</td>
<td></td>
</tr>
<tr>
<td>ME</td>
<td>4</td>
</tr>
<tr>
<td>NH</td>
<td>6</td>
</tr>
<tr>
<td>CT</td>
<td>8</td>
</tr>
<tr>
<td>HB</td>
<td>6</td>
</tr>
<tr>
<td>LP</td>
<td>7</td>
</tr>
<tr>
<td>MO</td>
<td>11</td>
</tr>
<tr>
<td>Calcium</td>
<td></td>
</tr>
<tr>
<td>ME</td>
<td>1</td>
</tr>
<tr>
<td>NH</td>
<td>1</td>
</tr>
<tr>
<td>CT</td>
<td>2</td>
</tr>
<tr>
<td>HB</td>
<td>2</td>
</tr>
<tr>
<td>LP</td>
<td>6</td>
</tr>
<tr>
<td>MO</td>
<td>7</td>
</tr>
<tr>
<td>Potassium</td>
<td></td>
</tr>
<tr>
<td>ME</td>
<td>0</td>
</tr>
<tr>
<td>NH</td>
<td>1</td>
</tr>
<tr>
<td>CT</td>
<td>5</td>
</tr>
<tr>
<td>HB</td>
<td>1</td>
</tr>
<tr>
<td>LP</td>
<td>1</td>
</tr>
<tr>
<td>MO</td>
<td>2</td>
</tr>
<tr>
<td>Magnesium</td>
<td></td>
</tr>
<tr>
<td>ME</td>
<td>0</td>
</tr>
<tr>
<td>NH</td>
<td>0</td>
</tr>
<tr>
<td>CT</td>
<td>1</td>
</tr>
<tr>
<td>HB</td>
<td>1</td>
</tr>
<tr>
<td>LP</td>
<td>1</td>
</tr>
<tr>
<td>MO</td>
<td>1</td>
</tr>
<tr>
<td>Phosphorus</td>
<td></td>
</tr>
<tr>
<td>ME</td>
<td>0</td>
</tr>
<tr>
<td>NH</td>
<td>0</td>
</tr>
<tr>
<td>CT</td>
<td>0</td>
</tr>
<tr>
<td>HB</td>
<td>0</td>
</tr>
<tr>
<td>LP</td>
<td>0</td>
</tr>
<tr>
<td>MO</td>
<td>0</td>
</tr>
</tbody>
</table>

ME = spruce/fir in Maine; HB = hardwoods Hubbard Brooke, New Hampshire; MO = oakwood in Tennessee; CT = central hardwoods in Connecticut; LP = Loblolly pine in Tennessee; NH = northern hardwoods in New Hampshire.
should be noted that it is not the size of the 'available' nutrient pool which is crucial, so much as the rate of replenishment (Miller and Miller, 1991).

3 Some foreign studies suggest net losses of some nutrients may occur in the absence of harvesting (e.g. Federer et al., 1989). This is interpreted as showing that periods of apparent nutrient depletion may be a natural occurrence, or that they are brought about by acid precipitation. The suspicion must also remain that the budgets may not have quantified all the possible inputs and outputs correctly. It should be noted that a proportion of nutrient input is derived from pollution; so the balance in nutrient status is partly maintained by one anthropogenic influence (pollution) counteracting another (timber harvesting).

4 Nutrient depletion is potentially most important when WTH and whole tree chipping of thinnings is employed, because of the removal of nutrient-rich branch material. As a rule WTH removes two to three times as many nutrients as CH (Kimmins, 1977; Carey, 1980; Miller and Miller, 1991) and will be greatest when species have well developed crowns, large foliage biomass and high nutrient concentrations in foliage (e.g. spruce and firs) (Kimmins, 1977).

5 Problems of nutrient depletion are increased where rotations are short because a larger proportion of the harvested timber is nutrient-rich sapwood and bark (Kimmins, 1977; Raison and Crane, 1986).

6 The potential impact of increasing harvest intensity will be greatest where soil reserves of nutrients are low and where a large part of the nutrient capital is in the trees. Although studies of WTH provide conflicting results, the weight of evidence points towards the possibility of WTH potentially causing nutrient depletion on infertile sites where short rotations are used; for example on high windthrow hazard sites in Scotland (Nelson and Dutch, 1991).

7 Harvesting can lead to increased leaching loss, particularly of N and K, which usually lasts 2–5 years. At Kershope forest in Northern England, leaching losses of K increased five times, ammonium nine times and nitrate five times after felling, though changes in P were negligible (Adamson et al., 1987). Such losses are due to the cessation of nutrient uptake by trees, increased water percolation, warming of soil and increased inputs of organic matter (Nelson and Dutch, 1991). Such losses can be significant in terms of the nutrient status of a site (Stevens et al., 1988; Federer et al., 1989), though they rarely elevate nutrient levels in watercourses to values which might cause serious off-site impacts (Stevens et al., 1988). They are usually stabilized relatively quickly by revegetation of clearfelled sites. Such losses would be increased in short rotation crops due to more frequent disturbance. Leaching losses are greater after CH than WTH, because the presence of brash increases mineralization and delays revegetation (Carey, 1980).

8 Particular difficulties are encountered in defining and estimating total or available nutrient reserves, rates of weathering and in quantifying inputs from direct deposition (the latter may equal or exceed inputs in precipitation (Miller, 1979)).

The effects of cultivation

There is little useful information on impacts of cultivation on soil nutrient status. Robinson (1980) found leaching losses of calcium and to a lesser extent, magnesium, increased after ploughing of peaty soils at Coalburn. Increased levels of ammonium, nitrate, phosphorus, sulphate and aluminium have also been reported in drainage water following cultivation of mineral soils, but the changes are probably not significant in terms of site nutrient loss (Hornung and Newson, 1986; Leeks and Roberts, 1987).

The effects of fertilizing

Fertilizer is applied over about 50 000 ha of forest annually in Great Britain (Taylor, 1991), mainly to Sitka spruce plantations. Many upland soils are widely regarded as degraded as a result of previous land use practices, particularly where regular burning may have led to losses of nutrients by volatilization. Addition of fertilizer has been regarded as necessary to restore fertility, particularly in the case of phosphorus. In this respect it is worth noting that P deficiency is relatively rarely encountered in regions where forest cover has been maintained (Miller, personal commu-
Fertilizers have, however, also been used to boost the nutrient status of some soils to levels in excess of their natural levels.

Evidence from fertilizer experiments on sites restocked with Sitka spruce indicates that the nutrient status of second rotation crops is equally as good or better than first rotation crops during the establishment phase. However, the possibility exists that nutrient leaching from some sites may result in nutrient deficiencies in late thicket stage crops (Taylor, 1991). If fertilization remains restricted to the first rotation it could be concluded that fertilization and nutrient capture by the forest canopy has resulted in a relatively permanent improvement in site quality which is sustainable. If repeat fertilizer applications prove to be necessary to maintain forest productivity, this could be judged to be unsustainable, mainly on the basis of off-site impacts (energy use, mining, pollution).

Mitigating possible effects of nutrient depletion
Changes in the nutrient status of soils is a process which in theory at least is relatively reversible. Where there is cause for concern regarding nutrient depletion, and if fertilizing is considered unacceptable in the long term, the impacts can be mitigated by: using less productive species, lengthening rotations, and/or changing from whole-tree to conventional harvesting.

Compaction
Despite concern about the dramatically increased use of machinery during the last 20 years compaction has been the subject of relatively limited research. British forests may be at risk from compaction on account of the frequency of poorly drained soils, the use of heavy machinery in harvesting, multiple passage of machines during repeated thinning operations and the fact that harvesting takes place throughout the year (Moffat, 1991).

Compaction of soil leads to increased soil strength and density, decreased pore size, and concomitant reductions in soil aeration, water infiltration and plant root growth (Greacen and Sands, 1980). Forest operations that can cause compaction include: drainage, cultivation, harvesting and extraction, and the building and use of roads. The movement of machinery can churn and rut mineral soil, which together with reduced water infiltration, can increase the risk of localized erosion. Recreation pressures can also lead to local soil compaction.

Specific data on compaction and the loadings which cause them are scarce, and are available only from foreign studies. Greacen and Sands (1980) give data on the contact pressures for various forms of extraction (Table 5). They report increases in soil bulk density after logging operations in the order of 20–30 per cent. Sands et al. (1979) showed twofold increases in soil strength in the top 15 cm due to passage of a rubber tyred skidder and fourfold increases under logging roads. Compaction can be transmitted to considerable depths. Danfors (1974) recorded soil compaction under a 16 tonne load at 50–60 cm which persisted at least 3 years and also detected soil movement at 120 cm.

<table>
<thead>
<tr>
<th>Operation</th>
<th>Contact pressure (kPa)</th>
<th>No. of passes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ploughing</td>
<td>55</td>
<td>1</td>
</tr>
<tr>
<td>Thinning</td>
<td>0–125</td>
<td>6–300</td>
</tr>
<tr>
<td>Clearfelling</td>
<td>0–125</td>
<td>2–300</td>
</tr>
<tr>
<td>Extraction machinery</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cable crane, skyline</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Tracked skidder</td>
<td>30–40</td>
<td></td>
</tr>
<tr>
<td>Crawler tractor</td>
<td>50–60</td>
<td></td>
</tr>
<tr>
<td>Rubber tyred skidder</td>
<td>55–85</td>
<td></td>
</tr>
<tr>
<td>Forwarder with rear bogie</td>
<td>85–105</td>
<td></td>
</tr>
<tr>
<td>Forwarder with single rear axle</td>
<td>105–&gt;125</td>
<td></td>
</tr>
</tbody>
</table>

Table 5: Approximate contact pressures generated by the weights of unloaded forest machinery (after Greacen and Sands, 1980)
Impacts are frequently concentrated on part of the forest site and estimates of the proportion of stands affected by serious compaction range from 5-25 per cent following clearfelling (Froelich, 1973; Reisinger et al., 1988) to 0.5-3 per cent following thinning (Sidle and Laurent, 1986; Wingate-Hill and Jakobsen, 1982). In contrast to agriculture, compaction in forest soils is difficult and expensive to ameliorate because of the presence of stumps and roots.

The effects of compaction Compaction of natural or untilled soils usually, but not always, decreases yield. Greacen and Sands (1980) reviewed the results of 142 studies where soil compaction had been reported and recorded that this had led to reductions in tree growth in 82 per cent of cases. Compaction affects the penetration of soil by fungal mycelia, including mycorrhizal fungi associated with tree roots (Skinner and Bowen, 1974) and presumably also affects the growth of ground vegetation.

Compaction of indurated sandy loam gley on a restock site in Scotland led to reduced survival and reduction in height growth of Sitka spruce of 11-31 per cent (Pyatt and Anderson, 1986). Reductions in growth were well correlated with the degree of compaction and subsequent ploughing was not sufficient to remedy the damage. Similar reductions in early growth with increasing bulk density were reported by Tuttle et al. (1985) and Corms (1988). Froelich (personal communication quoted in Greacen and Sands, 1980) predicted a 15 per cent decrease in volume production as the result of tractor logging.

Retrospective studies of trees growing on compacted soils on skid trails show that tree heights may be reduced by 13->50 per cent and early volumes may be of the order of 10-20 per cent less (Helms et al., 1986; Helms and Hipkin, 1986). Froelich and McNabb (1984) reported that tree growth decreased 6 per cent for every 10 per cent increase in soil density; though Powers et al. (1990) doubt that such a linear relationship applies universally.

Compaction affects root growth in a complex fashion by influencing soil strength, aeration, and nutrient and water availability and limiting the presence of mycelia from mycorrhizal fungi. Research designed to test the effects of compaction on root and shoot growth is rarely sufficiently refined to allow the different effects to be distinguished (Greacen and Sands, 1980).

Rates of recovery Recovery depends on soil type and the degree of compaction with clay soils and those with a high organic matter content being more able to recover as a result of swelling and shrinking. Estimates of times to recovery taken from US, New Zealand and Australian studies range from 5-15 years after logging, 18 years for soil under landings, to 40 years or more under extraction tracks and forest roads (Moehring, 1970; Greacen and Sands, 1980; Wingate-Hill and Jakobsen, 1982).

It is likely that the current use of heavy machinery in forests in Scotland is causing compaction damage to susceptible soils and this will adversely affect tree growth during the second rotation. If total recovery takes place within the span of one rotation then long-term impacts will not be cumulative. If damage persists beyond one rotation length then long-term impacts may increase cumulatively, at least until a new 'stable state' is reached where soils are significantly more compacted than before the use of heavy machinery. This is most likely to occur where heavily mechanized harvesting takes place on wet soils and where rotation lengths are short and is likely to be restricted to extraction routes, temporary roads and landings.

Compaction in natural forests Compaction of forest soils can occur to a limited degree as a result of natural processes. Trees exert forces on the soil both on account of their weight when standing and fallen, and the forces they transmit to the forest floor from wind. In addition expansion of roots as they grow causes displacement and compaction of soil. For example Ryan and McGarity (1978) found a brown forest soil around the tap root of Eucalyptus grandis compacted over a radial distance of 2 m from the root. The lower horizons of some soils in Scotland are compacted at depth as a result of glaciation.

Soil under forests managed for timber production will almost certainly be more compacted than those under natural forests. Sands (1983) found that bulk soil densities beneath plantations of Pinus radiata were higher than on
comparable sites supporting native vegetation or pasture, and that soil densities were greater and extended deeper under second rotation plantations than first.

Effects of management operations All mechanized operations can cause compaction to some degree, but thinning and clearfelling operations would appear to pose the greatest threat as a result of the higher ground pressure and the likelihood of multiple passes (see Table 5). Mechanized cultivation techniques cause compaction where the tractor passes, but may reduce it in the surface layers of the cultivated soil. A high proportion of damage is caused by the first few passages of a machine, so cultivation should not be dismissed as a potential cause for concern merely on account of the fact that only one pass of a machine is involved.

Motor manual methods of harvesting will be associated with less compaction than harvesting based on harvesters or processors. Harvesting machines exert considerable ground pressures and traverse most of the site moving slowly, and hence could be expected to cause considerable damage to wet soils. Extraction by skidding and forwarding is liable to cause compaction despite the use of brash mats (Moffat, 1991); whereas cable cranes are associated with negligible damage.

Organic matter

Organic matter plays an important part in controlling chemical flux (including nutrient cycling), acting as a substrate for soil fauna and fungi, and influencing soil moisture, structure, strength and pedogenic processes. Forest soils in Britain span a wide range from littoral and heathland soils with relatively low organic matter contents, to deep peats composed entirely of organic matter. Many upland sites in Britain have accumulated large amounts of organic matter; a process which has often been accelerated by afforestation (Pyatt and Craven, 1979).

Reports of problems arising due to loss of organic matter are commonplace in agriculture (Mann, 1986), but appear to be less severe in forestry, except in dry climates where organic matter is vital for preserving soil moisture (Farrell et al., 1986). The organic matter content of soils is influenced by forestry operations in the following ways:

1 Drainage and cultivation lead to drying and decomposition of peat and litter layers on wet soils (Pyatt and Craven, 1979), a process which continues under the tree canopy. On deep peats this can lead to irreversible cracking and drying of peats, a reduction in total soil volume and increased release of nutrients (Williams et al., 1978; Pyatt and Craven, 1979).

2 Cultivation, drainage and harvesting may lead to losses of organic matter by erosion. For example Lewis and Neustein (1971) recorded the loss of 136 kg ha$^{-1}$ a$^{-1}$ of organic matter following harvesting on 25-35° slopes in west Scotland.

3 Site disturbance resulting from harvesting may lead to increased rates of decomposition and release of nutrients (see preceding section).

4 Timber harvest leads to far lower inputs of woody debris than is the case in natural forests. This has particular implications for soil fauna and fungi.

5 Whole tree harvesting and post-harvesting practices such as windrowing (bulldozing felling debris and stumps into heaps or lines) lead to reduced inputs of organic matter. Several experiments (with fairly extreme treatments) have shown that removal of litter and subsequent reductions in organic matter can cause changes in soil nutrient status (Wiedemann, 1933; Nykvist and Rosen, 1985; Dyck and Skinner, 1990). Windrowing is not often used in the UK, but mechanized harvesting can lead to similar effects. For example Dyck and Skinner (1990) showed that WTH followed by litter-raking in Pinus radiata produced reductions of 30-34 per cent in exchangeable Ca and Mg in soil at 5-20 cm depth relative to control.

6 Burning of brash leads to considerable loss of organic matter (Powers et al., 1990), but is rarely practised in Scotland.

7 Trees replace the role of previous vegetation in providing litter (including woody debris). Establishment of woodland on arable land will lead to increases in total organic matter content of soils. On moorland, litter inputs
from native trees will probably be of the same order of magnitude as that of a Calluna sward (Alexander and Cresser, in press) while litter inputs from intensively managed conifer plantations will exceed those under extensive agriculture. The characteristics of organic horizons will depend primarily on the tree species present (Grieve, 1978; Miles, 1985).

Speculating about the long-term consequences of changes in organic matter content of forest soils is made difficult by the high degree of alteration wrought by the recent and historical activities of man. Where peat accumulations are artificially high due to previous land-use history, losses of organic matter which result from afforestation need not be a cause of concern, provided that organic matter content eventually reaches an acceptable new steady state under tree cover. One exception to this concerns afforestation of peats in areas of high nature conservation value which would not naturally have carried tree cover; here forestry operations cause irreversible changes to soils.

On soils low in organic matter the net effect of forestry operations where conventional harvesting is used will probably be to raise the organic matter content of soils, particularly under fast-growing tree species and where fertilizing is used. However, if whole tree harvesting were used on littoral soils the potential for long-term reductions in organic matter content exists. The lower inputs of woody debris to the soil under forests managed for timber production as compared with natural forests may have implications for forest soils, fungi and soil fauna.

**Physical disturbance**

Nature conservation requires that natural characteristics of forest soil profiles are retained, and this is a component of their sustainable management. Soil profiles have been widely disturbed in Scotland both by historical and recent cultivation, and as a result of changes in vegetation and land use. In some respects the remaining relatively undisturbed soils can be regarded as a non-renewable resource; and a case can be made that conserving a representative cross section of such soils represents the best interpretation of sustainable management.

Ploughing and draining causes considerable disturbance to soil profiles by inverting and mixing horizons and interfering with pedogenic processes. Worrell (1996) ranked cultivation techniques according to the volume of soil disturbed as follows: ploughing 350–850 m$^3$ ha$^{-1}$, dolloping 320–400 m$^3$ ha$^{-1}$, mounding 170–340 m$^3$ ha$^{-1}$, scarification using a disc trencher 110–280 m$^3$ ha$^{-1}$, hand turfing 40–60 m$^3$ ha$^{-1}$ and hand screefing 5–6 m$^3$ ha$^{-1}$.

**Soil disturbance in natural forests**

Natural disturbance in forests in oceanic climates is driven largely by windthrow (Peterken, 1977) which causes some disturbance to soil profiles. Long-term impacts of forest cultivation on soil profiles can be minimized by selecting ground preparation techniques which most closely mimic disturbance by wind and fire. This would favour the use of mounding and patch scarification.
Some effects of changes in soil status on forests

Tree growth

Considerable concern has been expressed over the potential for changes in soil status to be reflected in tree growth rates. Changes in timber production due to forest practices have been studied in Europe since the mid 1800s (e.g. Grigor, 1868) and this is practically the only index for which medium or long-term data exist. No British data of this type exist.

Substantiated reports of declines in productivity are relatively few in number, and it has been pointed out that they are outnumbered by reports of productivity having been maintained or increased as a result of management (Kimmins, 1990a). This should come as no surprise considering that forestry practice worldwide has been driven by the aim of increasing the rate of timber production by increasing the intensity of management. The important point is that some declines in productivity have been reported despite the scale of efforts designed to achieve the opposite.

Productivity declines are one of those subjects in which anecdotal evidence has a habit of becoming cast in tablets of stone, and separating myth from reality is difficult. The following section summaries evidence from the most frequently quoted reports of productivity decline from foreign studies.

Spruce sickness in central Europe

The earliest reports of reduced yields from one rotation to the next were based on largely anecdotal evidence concerning changes between the first and second rotations of Norway spruce monocultures growing on sites previously occupied by mixed or hardwood forest in Germany (Grigor, 1868; Wiedemann, 1923). Wiedemann concluded that growth had indeed declined in second and third rotation crops and that this was caused by adverse changes in soil properties; though in a later version of this paper he suggested that these changes had been overstated. Wiedemann's views were disputed by Holmsgaard et al. (1961) who concluded that there was little evidence of general decline between first and second rotations. He concluded that poor performance of spruce was limited to sites on heavy clay soils to which it was clearly not suited; this was partly due to progressive deterioration of drainage as old root channels left from previous hardwood crops disappeared.

As illustration of the difficulty of drawing conclusions from these retrospective studies is provided by the contrasting conclusions of two recent authoritative reviews. Powers et al. (1990) concluded that 'there is no evidence that spruce monocultures [in Europe] per se lead to soil deterioration'; whereas Pritchett (1979) states 'the problem is undoubtedly real with spruce in central Europe'.

Litter raking in East Germany

Wiedemann (1935) demonstrated considerably reduced growth of Scots pine where stands had been subject to removal of litter for many decades by local people for use as animal bedding. Volume increment, expressed as 'Site Class' on a scale of 1 (good) to 5 (poor), was 1–2 classes lower on plots that had been subject to litter raking than on undisturbed plots. This remains a benchmark study demonstrating that removal of significant amounts of organic matter can affect the tree growth rates on poor sandy soils.

Pinus radiata in New Zealand

Reduced growth in second rotations of Pinus radiata in New Zealand has been widely reported, particularly on poorer sites. Reduced growth as a result of nitrogen deficiency was first observed on gravel soils (Stone and Wills, 1965) which required that the second rotation was extended by 8 years in order to achieve the same final volume as the first rotation (Whyte, 1973). Ballard (1978) demonstrated an up to 40 per cent decline in volume growth in Pinus radiata as the result of windrowing persisting at least 17 years into the second rotation and thought to be mainly the result of poorer nutrition compounded by soil compaction.

In experiments, whole tree harvesting plus litter-raking produced 12 per cent reduction in growth of Pinus radiata at ages 16 and 26 (Ballard and Will, 1981; Dyck and Skinner, 1990) and these were linked to changes in soil nutrient status. In a trial combining litter removal, topsoil removal and soil compaction Skinner et al. (1989) reported growth at year 4 reduced to only
Table 6: The effect of litter removal, topsoil removal and soil compaction on the growth of 4-year-old Pinus radiata in New Zealand (after Skinner et al., 1989)

<table>
<thead>
<tr>
<th>Treatment</th>
<th>4-year volume compared with control (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Litter removal by hand</td>
<td>75</td>
</tr>
<tr>
<td>Litter removal by machine</td>
<td>35</td>
</tr>
<tr>
<td>Topsoil removal, low compaction</td>
<td>30</td>
</tr>
<tr>
<td>Topsoil removal, high compaction</td>
<td>20</td>
</tr>
</tbody>
</table>

20 per cent of control (see Table 6). Collectively this evidence suggests that growth reductions in fast-growing crops have occurred, and that they may continue where particularly intensive management techniques are used.

Pinus radiata in Australia Second rotation crops of Pinus radiata on some sites in southern Australia showed considerably reduced growth rates compared with those of the first rotation (Keeves, 1966). This has been linked to various changes in soil conditions: bulk density, organic matter content, reduced N content (Hamilton, 1965); nutrient content due to harvesting and soil moisture (Bednall, 1968; Flinn et al., 1980; Farrell et al., 1986). Subsequent research has demonstrated that if brash is retained after clearfelling, growth during the subsequent rotation equals or exceeds that of the foregoing one; and this is interpreted as the result of enhanced moisture retention (Farrell et al., 1986). Such effects are unlikely to be encountered in Scotland.

North America There are several reports of damage to soil resulting from management operations (Powers et al., 1990) but evidence of resultant declines in productivity are few. In British Columbia it is estimated that 20 per cent of the area of forest harvested between 1976 and 1986 was affected by soil degradation and that this could account for growth losses of over 1 million m³ a⁻¹ by the year 2000 (Baker and MacKinnon, 1990) though various links with climatic change and pollution have been suggested (Powers et al., 1990). A similar situation is reported for red spruce in the north-east US, though here natural ageing of the population may also be involved (Powers et al., 1990).

Methodological problems The reviews above illustrate the difficulties involved in providing conclusive evidence of productivity decline. A major difficulty is establishing a baseline or standard control against which to assess changes in productivity. This means collating data from two (or more) rotations on the same site (the historical approach) or from two identical sites with different treatments (paired plots). The historical approach requires detailed monitoring of growth over a long enough time period to even out the influence of exigencies such as changes in climate, incidence of disease, pollution damage etc.; and there are few instances of this having been achieved. In addition, the genotype and cultivation techniques of the original crop may be unknown making it difficult to draw conclusions from retrospective studies. The paired plot approach can never overcome nagging doubts about how similar the original sites are or were.

These problems have acted to dissuade researchers from embarking on investigations of this type, particularly in an era when research has been dominated by the need to achieve immediate increases in production by manipulation of the site.

Changes in biodiversity

Concern over whether soil conditions altered as a result of forest operations influencing flora and fauna has only recently come to the fore and the topic is relatively poorly researched. Management for timber production is known to cause reductions in the diversity of ground flora, fungi and invertebrates. However it is difficult to distinguish between the effects of soil disturbance on biodiversity and those of the composi-
tion (species choice) and structure of forest canopy; and most research to date has concentrated on the latter (e.g. Mitchell and Kirby, 1989). Studies in agriculture suggest that cultivation leads to reduced diversity of soil-living organisms, but that total biomass may remain relatively unaffected (Brady, 1974).

Possible effects of changes in soil status on flora and fauna include:

1. Erosion and compaction reducing the growth and biomass of ground vegetation, fungi, microflora and invertebrates.
2. Reduced input of woody debris resulting from harvesting which may affect diversity and density of soil fungi, microflora and saproxylic invertebrates. For example, densities of soil mites and microarthropods were reported to decrease in the order: unharvested forest, conventionally harvested, whole-tree harvested forest (Bird and Chatarpaul, 1986).
3. Soil disturbance increases the populations of species associated with disturbed conditions at the expense of those associated with stable conditions under the woodland canopy (Mitchell and Kirby, 1989). For example the populations of ground beetles was reported to increase after scarification in a pinewood in Scotland (Parry and Rodger, 1986).
4. Where cultivation and drainage during afforestation reduce the water content and improve the aeration of soils, the diversity and density of ground vegetation, fungi, microflora and invertebrates may increase.

This is a field in which greater research effort is required.

Discussion and conclusions

This review highlights soil erosion, nutrient status and compaction as the issues over which greatest concern has been expressed regarding the influence of some forest operations on the sustainable management of forest soils. The weight of evidence points towards there being a reasonably close balance between the impacts of some of the more intensive forest management regimes and the capacity of some susceptible sites to recover. The range in soil erosion rates under some deep ploughed upland plantations on erodible soils appears to be very similar to estimated rates of soil formation, at least in plantations established before current guidelines. The removal of several nutrients under whole tree harvesting regimes (and possibly under conventional harvesting on some sites) appears to be of the same order of magnitude as inputs on the most infertile sites and when rotations are short. Soil compaction is less well researched, but where soils have been disturbed by harvesting operations, reduced growth of subsequent crops has been observed, and compaction has been recorded as lasting close to rotation lengths.

These observations suggest that some degree of soil deterioration will probably occur if intensive forestry practices are used on susceptible sites. Some localized damage due to erosion and compaction is inevitable. The likelihood of more widespread degradation increases with the intensity of site preparation, the use of heavy machinery, and with shortening of the rotation. Susceptible sites include sandy and loamy soils in high rainfall areas (erosion), wet mineral soils particularly with a high clay content (compaction) and infertile soils such as littoral and the worst heathland soils (nutrient depletion). However, the scale of occurrence of these impacts is probably relatively modest. They are also decreasing, except for the impacts of mechanized harvesting.

On less susceptible sites and under less intensive management regimes it is unlikely that soil deterioration will occur; indeed in many instances the levels of human impact on soils under forestry may be lower than under preceding land uses. Compliance with recent Forestry Commission guidelines (Forestry Commission, 1993) should also lead to lower impacts than those recorded during recent decades, particularly with respect to erosion.

Differentiating sustainable from unsustainable practices or regimes is difficult given current information. Ploughing, the use of heavy harvesting and extraction machinery and whole-tree harvesting are the operations that are most likely to cause unacceptable long-term impacts. It should be emphasized that single operations cannot be considered in isolation but need to be assessed as part of a management regime the impacts of which need to be aggregated.
throughout an entire rotation. Collating data on and modelling these impacts to provide rotation-length assessments of the impacts of different management regimes on different sites should be a high research priority.

If it is correct that the balance between impacts and the capacity of sites for recovery is close, it also follows that only relatively modest changes in forest practices may be needed in order to secure sustainable management. Reducing the likelihood of soil degradation can be achieved by decreasing either the magnitude of impacts of modifying operations, or their frequency by increasing rotations. Current improvements in cultivation and harvesting practices are a welcome move in this direction (Paterson in press). Similar close scrutiny of the use of heavy harvesting and extraction machinery, particularly on wet soils, would appear to be the next priority.

Lengthening the rotation increases the period during which soils can recover from nutrient loss, compaction or soil loss. The term ‘ecological rotation’ has been used to describe the rotation length which allows soils to recover to their previous state after a damaging operation. The data collation and modelling exercise proposed above could be used to try to estimate the ecological rotation for forest crops on susceptible sites. At present, data on soil recovery processes is too limited to allow such estimates to be made with any precision.

Information on the impacts of operations on soils is almost invariably short-term and a variety of baselines have been used against which to assess the magnitude and significance of impacts. This review emphasizes the need to use appropriate baselines against which to assess the long-term significance of impacts. Comparing observed impacts with the rates at which soils are renewed by processes such as soil formation, nutrient input or recovery from compaction provides useful (if generalized) information on whether or not it is possible to continue to carry out present management practices into the foreseeable future.

This information can usefully be supplemented by comparing impacts observed in forests managed for timber production with those recorded in semi-natural woodland, thus providing information on the degree to which managed and ‘natural’ ecosystems are diverging. However research on the processes of soil renewal is still in its infancy and baseline measurements of soil erosion rates, nutrient inputs and compaction in natural woodlands are practically non-existent. This latter fact reflects an understandable preoccupation on the part of researchers worldwide with managed ecosystems. Priorities will need to change if information is to become available to allow informed decisions on sustainable management to be made.

In attempting to take account of the interests of future generations by instituting sustainable management practices, decisions will have to be made which are based on limited data; probably more so than for any previous major decisions in forestry. This presents a special challenge to scientists, policy-makers and practitioners, but it is one we cannot afford to ignore.

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