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Helicopter-based natural forest management for New Zealand's rimu (*Dacrydium cupressinum*, Podocarpaceae) forests

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Abstract

The management of indigenous conifer forests in Westland, South Island, New Zealand has focussed largely on harvest of the Podocarpaceae tree rimu (*Dacrydium cupressinum*). Today, key management objectives are to maintain the pre-harvested state of the forest in terms of biomass, tree size ranges (especially old trees), natural spatial patterns, relative proportions of the major tree species, and forest timber quality. This approach to natural forest management is based on utilizing heavy-lift helicopters, pre-empting mortality by harvest of a proportion of single unthrifty rimu trees, and an auditable GPS-based forest record system. A transition matrix model is used to determine the expected natural mortality rate and the sustainable harvest is based on pre-empting a proportion of the mortality over a 15-year felling cycle. Trees with significant wildlife features (e.g. cavities and large epiphyte loads) are not harvested. Even natural forest management can impact ecological processes through loss of senile trees, pre-empting forest tree windfall, altering spatial patterns of forest structure and facilitating pest and weed spread. However, because of the dynamic nature of these forest environments, the impact of forestry should be insignificant over major natural disturbance events. The best way to ensure that key management objectives are realized is to adopt an adaptive management approach that involves regularly monitoring of key forest attributes and frequent recalculation of the sustainable harvest model. © 2002 Elsevier Science B.V. All rights reserved.

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1. Introduction

Rimu (*Dacrydium cupressinum* Lamb.), New Zealand's most common conifer, once dominated lowland indigenous forest throughout the country except for the driest eastern areas (Norton et al., 1988). Because of its availability and favorable timber properties, including attractive color, strength and easy processing, rimu has been widely exploited. However, rimu timber sales have declined steadily since the 1950s as plantation grown exotic species, especially *Pinus radiata* D. Don,

have become New Zealand's dominant source of timber (Roche, 1990; Memon and Wilson, 1993). Rimu timber is still harvested mainly for craft and furniture uses where its hardness and decorative properties are prized more than radiata pine.

Beginning in the 1970s the management of New Zealand's rimu forests became caught in a divisive political controversy that eventually led to the demise of the New Zealand Forest Service in 1987 (Roche, 1990; Memon and Wilson, 1993) and the reservation of 78% of the West Coast land area for nature conservation purposes. Responsibility for the management of the remaining indigenous production forest was contracted to a new government-owned company, Timberlands West Coast Ltd.

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In this paper we describe a new approach for management of New Zealand's rimu forests on public lands. It is a natural management system that depends on the use of heavy-lift helicopters and has several other innovative management solutions to set and control yield in a manner that meets broad ecological sustainability goals.

Initially we review the ecological context and the history of forest management on the West Coast. We then describe the present management philosophy and how it is implemented, including the forest growth model used, and review some limitations associated with this management. Finally we discuss the implications of this approach to forest management on the future of indigenous forestry in New Zealand and elsewhere.

2. Study area

Rimu forests of the South Island's West Coast ($42^{\circ}30'$) occur in an area some 250 km long by 10–20 km wide, and occupy an altitudinal band from sea level to ca. 500 m a.s.l. The eastern edge of this area is defined by the alpine fault, the boundary between the Indo-Australian and Pacific continental plates. The climate is equable with cool summers, mild winters and high rainfall (2500–5000 mm; Hessell, 1982).

Much rimu forest is found on fluvial-glacial landforms (outwash surfaces and moraines) (Warren, 1967) with poorly drained and generally infertile soils (Mew and Palmer, 1989). Here the forest is dominated by rimu, but a number of other podocarp and angiosperm species are present in the subcanopy (Wardle, 1977). Miro (*Prumnopitys ferruginea* (D. Don) de Laub.) and kamahi (*Weinmannia racemosa* L.f.) are common in the subcanopy, and silver pine (*Manoao colensoi* (Hook.) Molloy), Halls totara (*Podocarpus hallii* Kirk), Westland quintinia (*Quintinia acutifolia* Kirk) and southern rata (*Metrosideros umbellata* Cav.) are less common. Many varied shrubs, climbers, ferns and abundant bryophytes dominate the shrub stratum and ground vegetation.

Forest composition and structure is strongly influenced by landform and disturbance history (Chavasse, 1971; Wardle, 1977; Duncan et al., 1990; Norton and Leathwick, 1990; Norton, 1994; Almond, 1996). The density and basal area of rimu is higher on poorly

drained outwash surfaces and lower on better drained moraine slopes. Several studies have commented that the rimu forest comprise distinct even-aged cohorts of trees (Cornere, 1992; Stewart et al., 1998). The spatially and temporally overlapping tree cohorts appear to have been determined by infrequent catastrophic disturbance, perhaps triggered by earthquakes associated with movements along the alpine fault (Wardle, 1980; Bull, 1996). More frequent smaller-scale disturbances from windfalls superimpose a smaller-scale patch structure over this (Cornere, 1992; Stewart et al., 1998).

2.1. History of forest management

New Zealand's indigenous Maori people used many plant and animal resources from forest but on the West Coast they had little impact on the main tree species, at least in terms of forest clearance (Peet, 1987). Significant exploitation of forest began in the 1860s with the arrival of European and Asian gold seekers and steadily increased as permanent settlements expanded. The construction of a railway line through the southern Alps to the eastern South Island accelerated the forest cut as the timber found a ready market in New Zealand's growing urban centers (Peet, 1987).

Rimu has typically been logged at unsustainable rates by exploitative harvesting systems. Despite concerns about the destructive impacts of harvesting, subsequent wildfires and the rate of forest loss (Foweraker and Hutchinson, 1934; Chavasse, 1954), the exploitation continued unabated until the 1950s. The first serious attempts at sustainable management by the New Zealand Forest Service were initially strip-felling (clear-cuts of alternate 80 m \times 500 m strips) (Holloway, 1954) followed by various selection systems using both high-lead cable harvesting and rubber-tired skidders. In the early 1980s, a coupe system was tried that involved small area clear-cuts (0.2–2 ha) that were replanted with nursery-raised rimu seedlings (James, 1987).

Despite a history of different silvicultural management and trials of less destructive harvesting methods (James, 1987), it has only been in the last decade that true sustainable management principles have been implemented in forest management (Richards, 1994).

2.2. Saltwater and Okarito forests

Two forests on the West Coast, Saltwater and Okarito (ca. 9500 ha), were legally gazetted in 1984 as reserves for the sustainable management of rimu and other indigenous timbers, and their current management forms the focus of this paper. A major river valley, predominantly used for agriculture, separates the two forests but elsewhere they adjoin high value conservation areas and therefore provide important regional landscape connectivity. Okarito forest has two small outliers and it contains an embedded ecological reserve (Fig. 1).

Considerable timber volumes had been extracted from these forests prior to their gazette as sustained yield forests, with ca. 80% of the area of Saltwater forest and 20% of the area of Okarito forest affected by previous selection harvesting (ca. 30% of the original forest volume was harvested). A good roading system is present in both forests. Some limited new roads have

been established while other existing roads have been decommissioned.

3. Methods and results

3.1. Forest plan

Given the past three decades of public controversy, it was obvious to Timberlands management that an entirely fresh approach to forestry was required if timber harvesting in indigenous forest was to have any future (Richards, 1994). Urban public opinion is generally unsympathetic to indigenous forestry in New Zealand and the reality for forestry is that harvesting operations must go largely unnoticed to the casual forest visitor.

The sustainable natural forest management plan written for Okarito and Saltwater forests (James, 1992, 1998a) attempts to balance timber production with ecological sustainability and the public perception of the forest's natural values.

In silvicultural terms, the forest plan aims to do the following:

- Maintain over time the biomass and structure of equivalent undisturbed forest. The biomass removed by harvesting should be recovered through growth before another harvest cycle.
- Harvest a proportion of trees across a range of diameter size-classes according to their predicted mortality rates rather than simply take trees from a particular size-class.
- Maintain the same composition of tree species over time.
- Select trees for harvest that are senile or unthrifty and expected to die within a particular management time-frame.
- Individually select single trees or small groups of trees (up to four trees maximum) and use GPS to ensure an even coverage of the area.

Any changes in these attributes should be from natural causes or processes rather than harvesting. Professional foresters choose individual trees above 30 cm diameter within canopy and/or stem condition classes similar to those described by Seydack (1995). Senile or unthrifty trees are clearly recognizable in

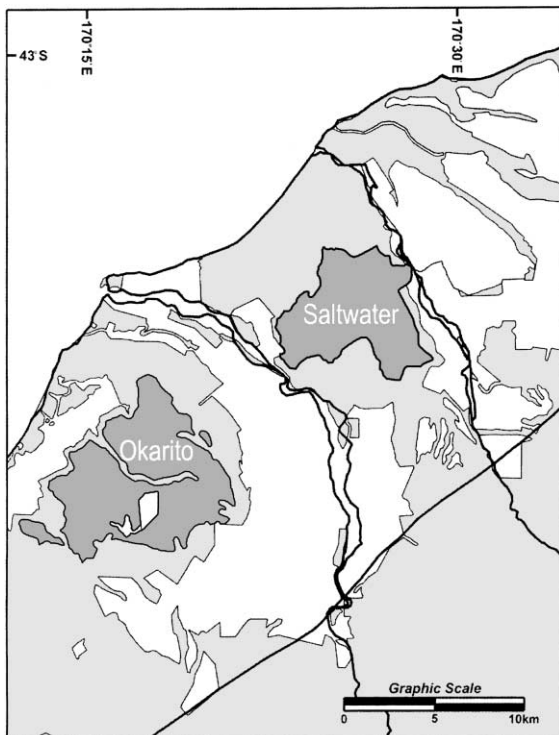


Fig. 1. Location of Okarito and Saltwater forests (dark shading), adjacent conservation lands (light shading) and private land (no shading).

rimu forest. An effort is made to avoid trees with a heavy epiphyte load or obvious potential nest holes. The overall aim is to maintain a healthy forest ecosystem in terms of stand structure and species composition that includes dead and dying trees, and to continue to sustain the full range of biodiversity that occurs within the forest.

3.2. Forest growth model

Population models that deal in changes in numbers of trees are used to determine the sustained yield for natural forest management rather than the conventional forestry practice of calculating and expressing increment in terms of timber volumes. Use of tree numbers rather than volumes allows a precise account of the yield to be made during the tree selection and harvesting operations, makes it easier to maintain forest composition and structure, and allows for every stump and log to be tagged and a GPS record taken to provide an auditable record on GIS of the size and location of all trees felled within the forest. Finally, non-forestry personnel and the general public find a yield expressed in tree numbers easier to understand than a yield based on timber volumes.

Permissible harvests are determined from a spreadsheet transition matrix model based on that developed by Usher (1966). This approach provides for more accurate forecasts than simple increment-based forecasts because each diameter class is modeled

separately (Vanclay, 1994). It also provides the forest manager with practical harvest rates for each diameter class.

The model requires data on tree size ranges, recruitment rates, growth rates and mortality rates. Tree diameter size distributions for rimu are derived from extensive inventory data from these forests (Table 1). All trees >100 cm diameter are grouped into a single maximum size-class and the combined data for Saltwater and Okarito forests are subdivided into five main forest-landform types (Almond, 1996).

The growth and survival of natural rimu seedlings had been measured over 15 years in an adjacent forest (James, 1998b). Cumulative survival rate was 28% after 15 years and 21% after 25 years. Mean height growth rate was 4 cm per year. It was predicted that 15% of natural seedlings would survive 35 years to reach an average height of 1.4 m (the smallest tree diameter class 0–10 cm). These data, combined with seedling numbers per hectare, form the basis to predict the recruitment rate for the growth model (Tables 2–4).

Rimu typically live for 400–600 years, with diameter increments of less than 2.5 mm per annum typical in these forests (Norton et al., 1988). The estimation of mean annual diameter growth within size-classes for each species has been determined from repeat measurements over four decades from historic inventories and large experimental plots in several lowland rimu forests in the general area (James, 1998a). These data cover a mix of both

Table 1

Average number of rimu trees (stems ha⁻¹) within diameter size-classes in the main forest-landform types in Saltwater and Okarito forests based on data from unpublished forest inventory plots^a

Diameter size-class (cm)	Terrace		Moraine		
	T4	T5 and T6	Ablation	Terminal	Lateral
0–9.9	521.9	77.0	454.6	186.3	23.3
10–19.9	82.9	21.0	46.3	14.8	6.4
20–29.9	54.0	22.8	22.3	16.2	5.8
30–39.9	52.2	24.4	17.3	16.8	7.7
40–49.9	43.1	24.2	17.0	13.9	6.1
50–59.9	27.2	20.2	13.6	11.9	6.1
60–69.9	13.6	15.7	15.2	9.6	6.7
70–79.9	7.0	8.5	12.0	9.6	6.4
80–89.9	3.0	4.0	6.9	7.3	5.3
90–99.9	1.1	2.3	4.3	3.5	4.4
>100	0.6	1.3	4.8	4.2	3.5

^a James (unpublished data).

Table 2

Average numbers of rimu seedlings (0.1–1.4 m height) and estimated annual recruitment rates into the smallest tree size-class (0–10 cm diameter size) based on data from unpublished forest inventory plots^a

Landform	Seedlings (ha ⁻¹) ± 95% CL	Recruitment (stems ha ⁻¹ per year)
Ablation	724 ± 124	3.10
Terminal	769 ± 138	3.30
Lateral	212 ± 78	0.91
T4	1082 ± 124	4.64
T5 and T6	1401 ± 411	6.00

^a James (unpublished data).

disturbed and undisturbed forest and are broadly representative of the current conditions of the two sustained yield forests. Altogether the growth data from these plots are for 9000 trees each measured for up to 36 years.

While growth is regular, mortality varies widely in scale and frequency. It seems likely for these lowland rimu forests that there are long periods of relative stability with isolated tree losses, punctuated by infrequent catastrophic disturbances that might result in the loss of 25% or more of trees within a forest (Cornere, 1992; Stewart et al., 1998). The infrequent

Table 3

Annual diameter growth rates (mm per year) for rimu trees within the main forest-landform types based on data from long-term forest research plots^a

Diameter size-class (cm)	Terrace		Moraine		
	T4	T5 and T6	Ablation	Terminal	Lateral
0–9.9	0.87	0.40	0.40	1.49	0.40
10–19.9	0.90	0.83	0.83	1.38	0.83
20–29.9	0.75	1.10	1.10	1.87	0.82
30–39.9	0.90	1.33	1.33	1.57	1.92
40–49.9	0.94	1.44	1.44	1.30	1.82
50–59.9	0.95	1.46	1.46	1.24	1.80
60–69.9	1.10	1.47	1.47	1.04	2.07
70–79.9	0.99	1.25	1.25	1.15	2.04
80–89.9	1.37	1.53	1.53	1.09	2.56
90–99.9	1.37	1.53	1.53	1.58	2.56
>100	1.37	1.53	1.53	1.95	2.56

^a James (unpublished data).

Table 4

Permissible harvest of rimu by landform classes (trees per year) for Saltwater and Okarito forests under the natural forest silvicultural system over one 15-year management period

Diameter size-class (cm)	Terrace		Moraine			Total
	T4	T5 and T6	Ablation	Terminal	Lateral	
30–39.9	24	5	5	4	1	39
40–49.9	50	15	9	22	1	96
50–59.9	60	19	16	40	1	137
60–69.9	89	45	29	51	3	216
70–79.9	71	84	33	33	4	226
80–89.9	26	84	28	27	6	171
90–99.9	40	42	20	16	15	133
>100	20	60	70	69	60	279
Total	380	354	210	262	92	1298

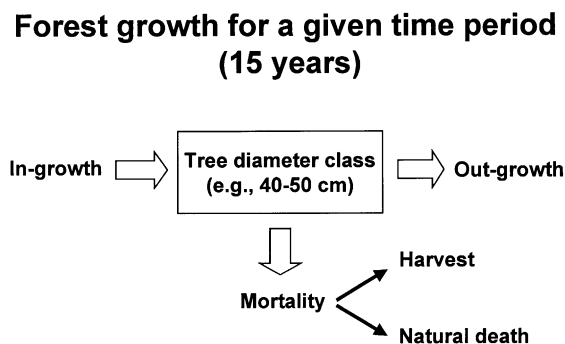


Fig. 2. Potential fate of rimu trees over a management time period of 15 years for one 10 cm diameter size-class group.

nature of mortality makes it difficult to measure the true rate without fieldwork over several centuries.

An alternative strategy is to use the forest growth model to derive mortality rates that in theory must occur if the forest is to maintain its existing biomass and structure. For any size-class range of trees within the forest (e.g. trees 40–50 cm diameter) over a particular management time period (15 years is presently being used), four things happen (Fig. 2). Some trees grow into the size-class from smaller size-classes (ingrowth), some trees increase in size but remain within the size-class, some trees grow out of the size-class and into larger size-classes (outgrowth), and some trees die (mortality).

Beginning with the smallest size-class, the mortality rate for each size-class was set by iteration so that the same number of trees remained in that size-class at the end of a set period (usually a single 15-year felling cycle). This results in the shape of the diameter distribution and forest biomass (basal area or volume) that remains constant through time. The ecological logic for this process is based on the assumption that the shape of tree diameter size distributions from large area samples reflects the response of the forest to the environment and site factors of each landform. To aim to perpetuate that natural situation, at least over a limited time, is a fundamental goal.

3.3. Permissible yield

The permissible yield was determined by assigning a proportion of the expected mortality to a harvest. The proportions assigned to the harvest vary from 20% for small trees to 60% for large trees (Fig. 3). This

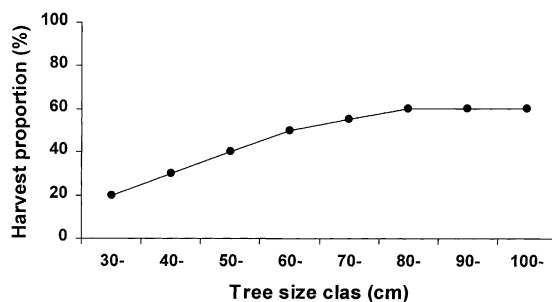


Fig. 3. Percent harvest rate for rimu in 10 cm diameter size-classes over 15 years.

range reflects our experience that because large trees are fewer in number and more conspicuous, their health status is easier to predict during the tree selection process than small trees.

The permissible harvest, expressed in terms of numbers of trees within 10 cm diameter size-classes per year over the felling cycle, is determined individually for each forest-landform type based on resource data specific to that type (Table 4). Any natural windfalls harvested, or trees harvested along roadsides, are included within this yield.

4. Discussion

4.1. Limitations of model

There are several areas of uncertainty associated with the forest growth model. The first is in how accurately we can determine which trees are likely to die within the next 15 years and therefore correctly select them for harvest. Long-term trials to assess the accuracy of mortality prediction classes as was done by Seydack et al. (1990) are needed. In the meantime, there is no simple solution to this problem, except to monitor the forest and re-calculate the forest growth model regularly.

A second but related problem arises in how to determine the proportion of expected mortality that will be harvested (presently this is set between 20 and 60%). The percentage harvested is critical for biodiversity conservation as dying and dead trees are essential for many species within the forest and too high a harvest level could have adverse effects on

biodiversity. At present this has been set on a precautionary basis and further research is required to better estimate the harvest level.

Third, the forest growth model is essentially an equilibrium model in that it aims to maintain the current biomass and structure of the forest. Clearly these forests are non-equilibrium systems that are structured by a mixture of frequent and infrequent disturbance events. The potential of the model to force the forest into a static condition can be reduced by recalculation of the growth model at regular intervals (e.g. every 10 years) so that it is sensitive to disturbance-induced changes in forest structure.

4.2. Harvest technology

One of the biggest problems with earlier forestry operations in these lowland rimu forests resulted from the high impact of ground-based harvesting operations on forest soils and soil water tables (James, 1987). A major change in harvesting technology proved financially viable after the fall of communism when ex-military Russian heavy-lift helicopters became available. The most commonly used helicopter, the Mil 8, has a 5 t lift capacity and when fitted with a log grapple can harvest upwards of 100 t h⁻¹.

The advantage of helicopters is their ability to economically harvest widely scattered log resources with negligible impact on the forest ecosystem, especially soils. Their high productivity means that they are only present in the forest for a few days each year so that disturbance is reduced during the breeding seasons of wildlife species. Furthermore, helicopters can economically operate over a 2 km radius which reduces road densities to ca. 5 m ha⁻¹. At that level the negative impacts associated with roads are reduced (e.g. habitat fragmentation and weed invasion).

4.3. Other aspects of forest management

Forest management activities are regularly audited, with a formal 5-yearly independent audit (Burn-Murdoch and Norton, 1998), and annual operational audits undertaken by the New Zealand Ministry of Agriculture and Forestry. A number of monitoring programs are undertaken including a record of GPS location on all tree stumps, operation monitoring (quality control) and biodiversity monitoring including

vegetation (based on permanent sample plots) and “5 min” bird counts. In addition, researchers are encouraged to make use of these forests and hence can provide further information on the effects of forest management.

As well as timber production, management also involves control of introduced weed species and mammal predators. Possums (*Trichosurus vulpecula* Kerr) are a major herbivore in New Zealand forests feeding on a wide variety of food items including foliage, flowers, fruit and fungi (Nugent et al., 2000). Possums as well as rats (*Rattus* species) and mustelids (*Mustela* species) are also major carnivores in these forests feeding on birds, bats, reptiles, fish and invertebrates (Atkinson and Cameron, 1993). Active control of these introduced animals, and plant pests, is part of the forest management.

4.4. Ecological impacts of sustainable forest management

Managing an indigenous forest for timber production will have some impact on the indigenous biodiversity within the forest no matter how careful the extractive management is. While it is not possible to retain the natural values of a forest in an identical manner to that which would have been present without management (Norton, 1996), it is possible to minimise loss of value through appropriate ecologically-based forest management. The potential exists for sustainable forestry to maintain natural forest structure and composition to a level that provides habitat for the full range of indigenous biodiversity that would be expected to occur in a comparable unmanaged forest.

Five types of impact on ecological processes are likely to occur as a result of the sustainable management of indigenous forests (Norton, 1996). The following section briefly reviews each of these from a New Zealand perspective and suggests how the sustainable management proposed here relates to them.

4.4.1. Loss of senile trees

By removing some trees before they die, natural forest management will result on average in fewer senile trees in the forest. These trees play a key role in overall forest biodiversity. For example, in mixed New Zealand podocarp–angiosperm forests several forest birds feed primarily in the largest diameter trees

(O'Donnell and Dilks, 1994) including dead trees. Senile trees also provide nest sites (especially cavities) for many of these birds. The largest trees support the greatest abundance and biomass of epiphytic species, including ferns, lilies, orchids and climbing plants, which provide habitat for numerous invertebrate species while several beetles rely on the bark of mature trees for shelter (Didham, 1997).

Under the natural forest system proposed here, there will always be adequate numbers of trees that grow into the largest size-classes and hence there should always be habitat for indigenous biodiversity associated with these trees, and dead and dying trees, will continue to be present in these forests. Large trees with particularly heavy epiphytic loads or with obvious signs of bird nests are deliberately not selected for harvest.

Furthermore, predation by introduced mammalian species in most New Zealand forest appears to have a much greater influence on the abundance and distribution of indigenous animals, especially birds and bats, than does resource supply (food and breeding/roosting sites) (Atkinson and Cameron, 1993). Benefits to biodiversity that result from predator control are likely to exceed habitat losses that might occur as a result of harvests.

4.4.2. *Pre-empting forest tree windfall*

Removing senile trees before they die may reduce the incidence of tree windfall and hence soil turnover. Tree windfall disrupts the processes of soil development and maintains soils at a younger stage of development by raising less weathered material close to the surface and thus improves fertility and drainage (Campbell and Mew, 1986; Adams and Norton, 1991). This is a process not easily mimicked by silvicultural systems based on natural disturbance regimes. Secondly, tree windfall creates opportunities for regeneration through elevated light levels and elevated sites on root-plate mounds and fallen logs (June and Ogdén, 1975; Duncan, 1993). Gap creation is a process that forest managers can mimic.

As much as 70% of tree mortality in these forests is due to windfall (James, unpublished data), although the spatial and temporal patterns of natural tree windfall are poorly understood. Tree windfall-induced soil turnover follows an exponential decay model (Norton, 1989). The time taken to turn over half the soil (soil

turnover half-life) has been estimated at 2960 years for the outwash surface rimu forest (Norton, 1989). The primary effect of harvests will be to extend this turnover, but at this stage it is uncertain as to how much this will be. However, infrequent catastrophic disturbances probably account for a large proportion of total windfalls and clearly reduce the effect of forest management on soil turnover.

4.4.3. *Altering spatial patterns of forest structure*

A range of disturbances from single tree windfall to catastrophic stand disturbance (e.g. after large-scale forest blowdown or earthquake) occur naturally in these forests and result in complex spatial distributions of tree sizes (Cornere, 1992; Stewart et al., 1998; Rogers, 1997). A key limitation to better understand these patterns is the longevity of trees and the long time-interval required to fully understand the processes of disturbance and tree regeneration.

Long-term forest management will find it difficult to mimic these complex natural patterns. There is therefore a potential for forest management to, intentionally or unintentionally, transform a forest into one with an all-aged structure, which has the potential to lead to long-term changes in forest biodiversity.

4.4.4. *Coarse woody debris and nutrient loss*

Coarse woody debris is important for regeneration of many plant species, especially trees (it provides elevated regeneration sites: June and Ogdén, 1975; Duncan, 1993), and as habitat for vertebrates and invertebrates (e.g. kiwi, *Apteryx* species).

Information on nutrient cycling in lowland rimu forest is also limited, although the distribution of nutrients within the forest is closely linked to the stage of stand development in New Zealand beech forest (Allen et al., 1997).

Extraction of trees will reduce the amount of coarse woody debris within the forest and could alter nutrient availability (Stewart and Allen, 1998). However, with the stump and root system left plus the upper branches and foliage, a reasonable amount of coarse woody debris remains within the forest.

4.4.5. *Pest and weed spread*

The use of helicopters dramatically reduces the amount of roading required for forestry. Roads can act as barriers for some organisms, facilitate unwanted

plant dispersal, alter hydrological patterns, increase windfall and provide access for unlawful activities such as the gathering of firewood and hunting of threatened species. However, good management such as proper construction and maintenance, and regular roadside weed control can mitigate many of these problems.

While roads can facilitate the movement of some predators (Spellerberg and Morrison, 1998) this can also be seen as an opportunity for predator control. Roads attract pest species, which combined with the easy access, facilitates control operations.

5. Conclusions

We believe that the natural forest management being practiced in Okarito and Saltwater forests is an example of ecologically-based sustainable management of a temperate rainforest. Management has now been in progress for 7 years during which time ca. 9000 trees have been harvested. This number is equivalent to a single tree per hectare over the 7 years, a low harvest rate by most standards.

The key to the success so far has been the use of heavy-lift helicopters, which have minimized the impact of harvesting, required fewer roads, and limited harvesting activity to a few days each year. Also important is the modeling of forest yield in terms of tree numbers rather than volume and basing that yield on the forest's expected tree mortality rates. The models provide clear guidelines and feed back for field staff in selecting senile trees or windfalls within a distribution of tree size-classes. The model also enables the development of predictive scenarios to test the effects of different management options on long-term forest structure.

There are a number of limitations or uncertainties in the natural forest management approach. The proportion of mortality that is pre-empted for a harvest is still an arbitrary figure that can only be validated over time. Similarly, the impacts of lowering the number of senile trees in the forest on coarse woody debris, nutrient cycling, and dependent wildlife need monitoring. However, because of the dynamic nature of these forest environments, the impact of forestry should be insignificant over major natural disturbance events. The best way a truly ecologically-based

management can be maintained and the risk from incorrect assumptions minimized is to monitor and frequently recalculate the basic inputs to the model.

Okarito and Saltwater forests are located within landscapes that include a number of high-value conservation systems. The sustainable management being undertaken within these forests has been developed with these values in mind and provides a good example of the integration of protective and productive land use at the landscape scale. It provides a useful working example for the natural management of other rainforest, both in New Zealand and elsewhere. Whether the system will survive politically is now unlikely, as the new government elected in late 1999 has decided to end all harvesting of publicly owned forest in New Zealand.

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