2 Principles of ecological forestry

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Ecologically sound stewardship has long been a cornerstone of the forestry profession. But just what does 'ecologically sound' mean in practice? Historically, foresters were often taught that forest ecosystems could be engineered at will for human benefit. Ensuring ecological integrity meant not violating 'constraints' associated with soil, water quality, and wildlife (implicitly defined as well-known birds and mammals). Recently, the definition of ecological integrity has expanded; clearly, a primary focus is now on maintaining, and even restoring, native biological diversity. At the same time, a growing worldwide demand for forest products has encouraged foresters to expand traditional high-yield practices, amidst growing evidence that such systems often conflict with biodiversity.

While not discounting the difficulty of these conflicts, we believe there is a vision of ecological forestry that offers hope. To set the stage for the rest of this book, we define ecosystems, stands, and landscapes. Next, we review various incarnations of forestry, with emphasis on North American practice and the strong influence of the U.S. Forest Service. Hopefully, this will help readers to place the current discussion of ecological forestry into an historical, scientific, and professional context. Important principles of ecological forestry are defined and discussed, and related to traditional timber production forestry. Finally, a balanced forestry paradigm, which blends elements of traditional and ecological forestry, is described.

Ecosystems, stands, and landscapes

Asked to define *ecosystem*, a politician who was espousing the importance of protecting ecosystems hesitated for a long time then finally said, 'Well... they're kind of like an aquarium... they have plants and animals... and other stuff.' In fairness to the politician, ecosystems can be rather hard to define. Ecologists readily construct definitions such as 'a community of interacting species plus the physical environment that they

occupy', but, as we saw in Chapter 1, it is not always easy to move from a conceptual definition to defining ecosystems in the real world. Separating a lake and a forest is easy but where do you draw the boundary between a spruce forest ecosystem and a spruce swamp ecosystem? Is a spruce–fir forest that is 80% dominated by spruce (*Picea* spp.) a different type of ecosystem from one that is 80% dominated by fir (*Abies* spp.)?

One of the things that makes defining ecosystems particularly difficult is the fact that they can occur at any spatial scale. The examples used above (a forest, a lake, a forested wetland) imply a spatial scale that is commonly used: patches of vegetation that one can easily see from a small plane – patches one would usually measure in hectares, rather than square kilometers or square meters. However, ecosystems can be much smaller or larger. Aquariums are indeed small, artificial ecosystems. One could even argue that all the invertebrates and microorganisms that occupy a single fallen acorn constitute a tiny ecosystem (Winston 1956). On the other hand, we could argue that because all the organisms on earth interact with one another and their physical environment (through global carbon and oxygen cycles for example) that the whole earth is one ecosystem (a concept close to the Gaia hypothesis of James Lovelock, 1979). In recent years there has been a growing tendency, especially among natural resource managers, to define ecosystems at quite large scales, as in the 'Greater Yellowstone Ecosystem' (Mattson and Reid 1991). This tendency can probably be traced to the increasing emphasis on *ecosystem management*, a key principle of which is thinking at larger spatial scales.

Because *ecosystem* is a scaleless term we will avoid using it in this book except where the emphasis is on the general concept of ecosystems and not on any particular scale. For patches of forest vegetation that are reasonably homogeneous in terms of species composition, age, and density, we will use the traditional forestry term, *stand*. Stands are usually defined at scales that make them roughly equivalent to *communities* (although in fact, community is really a scaleless term like ecosystems) and we will use this as a generic term for forests and non-forests. For the arrays of forest stands, grasslands, wetlands, and so on that form heterogeneous mosaics across the land we will use the term *landscape* (Forman 1995). In recent years landscape ecology has emerged as an important subdiscipline of ecology that focuses on the ecological patterns and processes that emerge at spatial scales where vegetation is seen as a heterogeneous mosaic (Figure 2.1). The distinction between forest stands and forest landscapes is the basis for delineating two major parts of the book: Part II: The macro approach,

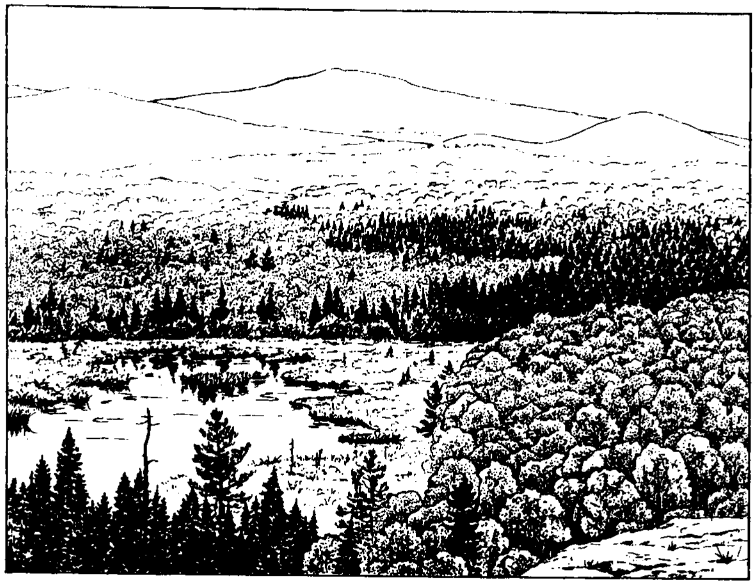

Fig. 2.1. Ecosystems can be defined at many scales; they are often recognized in terms of distinct patches of vegetation such as forest stands, wetlands, fields, etc.

managing forest landscapes; and Part III: The micro approach, managing forest stands.

Different models of forestry

Forestry in the broadest sense involves the science, art, and business of managing forests for human benefit. Forestry began at different times and different places throughout the world as societies' demand for wood outgrew the volumes obtainable by exploiting wild forests (Fernow 1913, Sedjo 1996). The earliest forms of forestry could be characterized as *custodial* (focusing on protecting the forest from overexploitation and fire), usually followed by *sustained yield timber production* (focusing on assuring a continuous supply of timber). More recently, explicit efforts to manage forests for a broad array of resources led to *multiple-use* forestry on many lands, while in other forests intensive efforts to maximize timber production following an agricultural paradigm led to *production* forestry. Some forests continue to be managed *extensively*, with little investment other

than protection. Many believe we have entered an era of *ecological* forestry, in which maintenance of ecological integrity will be paramount.

CUSTODIAL FORESTRY

The earliest roots of custodial forestry are obscure but it was no doubt well established in Europe during the middle ages. It took the form of increasingly restrictive laws that limited widespread unsustainable forest exploitation and conversion to agriculture (Fernow 1913, Plochmann 1992). Custodial forestry reached the United States in the late nineteenth century. In the western United States, forests were set aside from the remaining unsettled, unlogged public domain and later these became national forests. In the eastern and Midwestern United States, where lands had largely been logged over, the U.S. Forest Service began to buy them back in strategically chosen watershed protection zones. Custodial management emphasized fire protection, with low harvest levels; silviculture focused on natural regeneration. Planting was limited mainly to restoring trees to severely understocked lands. Owing to limited markets and a strong professional aversion to clearcutting, harvests tended to focus on large trees of valuable species.

The effect of custodial forestry on ecological integrity of public forests was probably mixed. On heavily exploited lands, the strong emphasis on protection and restoration has been undeniably positive. For example, 50–80 years later, many hardwood forests throughout the eastern United States are again beginning to resemble their earlier composition (minus the American chestnut, *Castanea dentata*, that was extirpated by an introduced fungus). Where custodial forestry was applied to old-growth, virgin stands, the focus on removing only large trees may have simplified stand structures, but the overall effect was fairly benign due to low harvest levels.

SUSTAINED-YIELD TIMBER-PRODUCTION FORESTRY

As timber demands increased and unexploited wild forests became scarce, it became clear that management needed to become more sophisticated in order to ensure that harvests could be sustained. Custodial forestry was thus supplanted by sustained-yield timber-production forestry. Key concepts of sustained-yield forestry are rotation (harvest) ages set at the point where average annual yield is maximum (the 'culmination of mean annual increment') and the regulated or 'normal' forest structure with equal areas of age classes up to the rotation age (Sedjo

1996). Although foresters promoting sustained-yield forestry clearly had a strong stewardship ethic, this model tended to predate, at least in Europe, the recognition of ecology as a science (Toumey 1928), and thus was generally devoid of any ecosystem perspectives.

MULTIPLE-USE FORESTRY

It has long been recognized that forests provide more than timber, and should be managed as such. To some extent the origins of managing forests for other resources are very old indeed; certainly European land managers have long been sensitive to the needs of various game species. However, it is only relatively recently that this perspective has been codified in law and pursued scientifically. For example it was in 1960 that U.S. National Forests were mandated to link multiple use with the traditional sustained-yield paradigm in the seminal Multiple Use-Sustained Yield Act. Opinions vary widely on the success and legacy of National Forest management under the Multiple Use-Sustained Yield paradigm; however, most agree that timber tended to remain the dominant output. Other values tended to be viewed as constraints, not equally important objectives (SAF 1993), despite an apparent legislative mandate to maximize social value of both market and non-market values (Krutilla and Haigh 1978, as cited in Sedjo 1996). Generations of foresters were educated under a philosophy that equated forest management with efficient timber production. Indeed, the index of Davis and Johnson's (1986) widely used text, 'Forest Management', refers to 'multiple use' only once (a cryptic paragraph in the introduction about laws governing forestry in the United States).

PRODUCTION SILVICULTURE

The implementation of sustained-yield forestry has become increasingly sophisticated, paralleling scientific advances in forest biology and technology. In the 1960s, just as forest management at the landscape level was equated with timber production, silviculture was often equated with maximizing timber yields at the stand level. Nowhere was this attitude more apparent than in the second paragraph of the influential text, 'The Practice of Silviculture', where David Smith (1962) characterized silviculture as '... somewhat analogous to ... agronomy in agriculture, in that it is concerned with the technical details of crop production.'

'High-yield' silvicultural systems patterned after an agricultural model

are common throughout the world. Beginning early in the nineteenth century, German foresters began a widespread and ultimately very effective effort to replace degraded mixed-species forests with conifer plantations, a program which has long been regarded as a national model of forest rehabilitation (Plochmann 1992). In other regions such as the British Isles, Chile, New Zealand, and the southern United States, conifer plantations were established over vast areas of abandoned crop or pasture lands, often with major government subsidies. In the southern United States, 20% of the softwood growing stock and 15% of timberland are now in pine plantations, 55% of which is owned by the forest industry (Rosson 1995). Production silviculture uses intensive practices to achieve high yields of economically valuable, genetically improved species (see Chapter 12). Any non-crop plants are viewed as competition to be controlled, usually by herbicides (Walstad and Kuch 1987). High timber yields demand close control and simplification of naturally diverse plant communities, and thus conflict inherently with promoting stand-level biodiversity.

EXTENSIVE FORESTRY

Another form of forestry has persisted on some private and industrial lands which have not adopted the above models. Often called *extensive* forestry to contrast it with the intensive nature of production forestry, this type of management tends to be characterized by opportunistic timber harvesting driven mainly by product demands. Silviculture usually consists of crude, broad-brush harvesting treatments with little investment in non-commercial treatments. Harvest levels typically are low relative to potentials under intensive management, but can still be non-sustainable relative to the actual (i.e., low or non-existent) level of investment. Allowable cuts are rarely determined with any reliability, however, so its timber sustainability is difficult to assess.

Some environmentalists tolerate or even support extensive forestry because its 'low-budget' approach tends to eschew practices such as clearcutting and herbicide spraying that they find objectionable. They may not realize that protracted extensive forestry has tended to reduce age diversity (by discriminating against old trees and stands) and to favor aggressive, often early- successional species. Over time, forest regions dominated by multi-cohort stands of valuable late-successional forests may be gradually replaced by younger, single-cohort communities that are neither ecologically well adapted nor economically productive. This conversion can be somewhat insidious, because it happens over time horizons that span human generations and may thus be imperceptible. An example is the red spruce (*Picea rubens*) forests of the Acadian region of northeastern North America: the short-lived, aggressive balsam fir, red maple, and aspen have become increasingly common at the expense of the spruce and long-lived northern hardwoods such as yellow birch and sugar maple (Seymour 1992).

ECOLOGICAL FORESTRY

By the late 1980s widespread dissatisfaction within the forestry profession and scientific community with traditional sustained-yield multiple-use forestry in the United States was publicly manifested in three prominent critiques. In 1989, Jerry Franklin published an influential article in which he argued for a 'New Forestry' on U.S. national forests. A year later, a distinguished panel of forest scientists issued a visionary report calling for a new approach to studying and managing forest ecosystems, distinct from traditional commodity approaches (NRC 1990). Shortly thereafter, another task force of scientists and professionals convened by the Society of American Foresters issued a controversial indictment of traditional sustained-yield forestry, also calling for a revamped, ecologically based approach (SAF 1993).

Only history can judge whether we are witnessing an historic revolution to a profoundly new era of 'ecological forestry', or just an incremental evolution of the multiple-use doctrine. We do not intend to contribute to the lengthy debate about the exact meaning of 'ecological forestry', 'forest ecosystem management', 'new forestry', and similar terms (cf. Grumbine 1994, Irland 1994, Salwasser 1994). We accept their inherently fuzzy nature (More 1996), recognizing that broadly accepted definitions will only come after future implementation by practitioners. We will use the term *ecological forestry* as a collective heading for the concepts and practices which constitute this new brand of forestry.

We do not mean to suggest that sustained-yield forestry and production silviculture are devoid of ecological underpinnings. Clearly, these practices manipulate ecosystems, albeit simplified ones, and ecological processes provide the sideboards that bound silvicultural possibilities. Nor has traditional, timber-oriented silviculture ignored natural stand development processes. The seventh edition of Smith's (1962:6–7) silviculture text captures quite well the attitudes prevailing among American foresters until perhaps a decade ago. Under the heading 'Silviculture as an Imitation of Nature', Smith wrote:

> The fact that [the forester] must know the course of natural succession does not indicate that he should necessarily allow it to proceed. Economic factors ultimately decide the silvicultural policy on any given area; the objective is to operate so that the value of benefits derived . . . exceeds by the widest possible margin the value of efforts expended.

What distinguishes ecological forestry, as we define it here, is the emphasis placed on natural patterns and processes: understanding them, working in harmony with them, and maintaining their integrity, even when it becomes financially difficult or inconvenient to do so.

In this chapter, we limit our treatment to the *biological* concepts that characterize ecological forestry. We recognize that ecosystem management in the broad sense also involves many administrative and socio-political issues (Grumbine 1994, More 1996) which are dealt with in Part IV of this book. We also recognize, but do not discuss, the prominent ecological role of physiographic factors which govern below-ground processes (hydrology, geology, nutrient cycling) and create above-ground gradients in productivity.

Natural disturbance regimes

It is easy to endorse the premise of sustaining ecological integrity, but just what does this mean in practice? We believe that the central axiom of ecological forestry is that *manipulation of a forest ecosystem should work within the limits established by natural disturbance patterns prior to extensive human alteration of the landscape.* The key assumption here is that native species evolved under these circumstances, and thus that maintaining a full range of similar conditions under management offers the best assurance against losses of biodiversity. This is analogous to the 'coarse-filter' approach (i.e., conserving diverse ecosystems and landscapes), in that it should maintain habitats for the vast majority of species (Hunter *et al.* 1988). With an effective coarse-filter strategy in place, the more costly and information-intensive fine-filter management can be focused on the few species of special concern.

Ecological forestry that maintains an effective coarse filter differs markedly from the 'engineering' approach common under sustained-yield timber management. Under that model, foresters try to define precise objectives for specific ecosystem components (e.g., trees, water, habitat for a particular endangered species) and use sophisticated quantitative methods to determine optimal management strategies. Though it can be considered appropriate for certain narrowly defined problems, we believe that there is a certain arrogance to such an approach to managing forests for biodiversity. It assumes a near-perfect understanding of the ecosystems under management.

EMULATING DISTURBANCES WITH MANAGEMENT

The fact that all forests are profoundly shaped by natural disturbances is now so widely accepted by ecologists and foresters that it is hard to imagine a time when things were otherwise. Yet, only a few decades ago, disturbances were viewed as extraordinary events – unnatural deviations from the normal successional development of equilibrium communities (Chapter 4 in this book, Oliver 1981, Pickett and White 1985, Oliver and Larson 1996). As a consequence, our knowledge of disturbances – along with our ability to use them as a template for managed stands and landscapes – is limited to relatively recent research, unlike the classical scientific underpinnings of sustained-yield forestry, which often go back a century or more.

Ecologists define *disturbances* as 'any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability or the physical environment' (Pickett and White 1985). To describe a specific disturbance *regime* and its effect on plant communities, three important parameters must be quantified (Chapter 4 in this book, Pickett and White 1985):

- *return interval*: the average time between occurrences in a given stand. Sometimes this is expressed as the *frequency*, which is simply the inverse of the return interval. For example, a regime with a 50-year return interval means that 2% (the frequency) of the landscape will, on the average, be disturbed in a given year.
- *severity*: the amount of vegetation killed, and the type of growing space made available for new plants, relative to that present before disturbance. A closely related, complementary concept is the *biological legacy* (Hansen *et al.* 1991), or the biomass that survives the disturbance in various forms, ranging from unaffected trees to dead and down material.
- *spatial pattern*: distribution of disturbance effects at various scales, from within-stand to large landscapes.

Important disturbance *agents* include fire, wind, herbivore outbreaks (Attiwill 1994a), as well as floods, avalanches, ice storms, landslides, volcanic eruptions, and glaciers (Oliver and Larson 1996:99–126).

Disturbance parameters often have high variability about average values. When this variability is coupled with many different disturbance

Table 2.1. *Some important forestry decisions typically made with a timber production emphasis but which also have strong ecological underpinnings, with examples*

Stand-level decisions (= silviculture)	Examples
Age structure (of trees in stands)	*single-cohort (= even-aged); multiple-cohort (= uneven-aged).*
Harvest timing	*50-year rotation; 10-year cutting cycle*
Regeneration method	*shelterwood; seed-tree*
Landscape-level decisions (= forest management)	
Annual harvest	*volume control: 200k m³/year*
	area control: 300 ha/year
Age structure (of stands in landscapes)	*areas by 10-year age classes*
Area in production silviculture	*25% of land in strategic timber base*
Area in ecological reserves	*10% of land not managed*
Location of harvest operations	*concentrated or dispersed cutting blocks*

agents operating in the same forest, the potential array of disturbance regimes can seem bewildering, defying meaningful categorization. Nevertheless, even broad groupings can be useful. For example, British Columbia has classified their forests into five broad 'natural disturbance types' (B.C. Ministry of Forests 1995) based largely on the return interval.

Foresters have found it useful to separate major, or *stand-replacing* disturbances, which kill virtually all of the overstory, from minor or *partial* disturbances which leave much of the stand alive (Oliver and Larson 1996: 95). Silviculturists have further subdivided lethal stand-replacing disturbances into *releasing* disturbances that kill the overstory only (releasing understory vegetation) or *severe* disturbances that have progressively more lethal effects on the understory vegetation and forest floor (Smith *et al.* 1997:163–4). Hurricanes are an example of the former, whereas most fires fall into the latter category. Such classifications have long been used by silviculturists to choose appropriate age structures and regeneration methods.

To provide an initial framework for discussion, we have listed some of the most important forestry decisions in terms typically understood by production foresters (Table 2.1). The remainder of this chapter will explain how to use an understanding of natural disturbance regimes when making these decisions. Forestry decisions are typically made at either the stand or landscape scale. In general, stand-level issues involve details of the silvicultural systems employed, whereas landscape-level decisions involve allowable cuts, harvest schedules, and protection strategies.

The next two sections will describe ecological forestry in the context of stand and landscape-level age structure, with their associated silvicultural systems and harvest levels.

Stand-level decisions

CHOOSING THE APPROPRIATE AGE STRUCTURE

The most defining feature of a silvicultural system is the age structure of the stand. Until quite recently, only two alternatives were recognized in North America: even-aged or *single- cohort* stands in which the trees are more or less the same age, and uneven-aged or *multi-cohort* stands which contain at least three distinct age classes (Smith 1986). Until *c.* 1960, uneven-aged or *selection* silviculture was the prevailing doctrine in North America, in part because clearcutting was still closely associated with exploitative logging, and because planting was limited to old-field reforestation. Then, an abrupt shift to even-aged management occurred, in recognition that many early attempts at selection silviculture had been failures (Seymour *et al.* 1986) and to implement high-yield silvicultural systems. Much highly polarized debate, both within and outside the forestry profession, has since focused on the merits of two extreme (and uncommon) endpoints of this silvicultural continuum – clearcutting with intensive site preparation and planting of monocultures, versus balanced single-tree selection cutting – ignoring the fact that the most logical silvicultural solutions to forest management problems often lay between (Smith 1972).

Happily, modern American silviculture is diverging from its earlier self-imposed rigidity surrounding the four classical systems of clearcutting, shelterwood, seed tree, and selection. Increasingly, silvicultural systems are viewed as a means of producing a virtually infinite array of stand structures to address an equally varied set of societal objectives (O'Hara *et al.* 1994). One example is the much wider use of *two-aged* silvicultural systems – essentially variants of even- aged systems where *reserve* trees or *standards* are left after the regeneration period. Although two-aged structures have very old lineage in Europe, actually predating more uniform systems (Troup 1955:121–8), only recently were they 'legitimized' in American terminology (Helms *et al.* 1994). These systems are treated extensively in two

recently published American silviculture texts (Smith *et al.* 1997, Nyland 1996), and Franklin *et al.* (1997) feature such *variable-retention* systems prominently in their discussion of ecologically based harvesting.

An important development with respect to strengthening silviculture's ecological basis has been to use the concept of *cohorts* to describe silvicultural systems (Oliver and Larson 1996, Smith *et al.* 1997:22–3), rather than mensurationally based *age classes*. In silviculture, cohorts are populations of trees that originate after some type of disturbance (natural or silvicultural) that makes growing space available, regardless of whether they differ in age by more than 20% of the rotation. For example, foresters can now speak of growing irregular 'three-cohort' stands without worrying about the fact that such a system might not fit some standard cookbook.

Even-aged silviculture is popular mainly because it is easy and economical to understand and implement, and, in the case of high-yield production forestry, because the specific stand establishment practices require it. It is tempting for overworked foresters to extend these administrative advantages to the treatment of natural stands. Whether this is justified ecologically depends critically upon the interaction of disturbance severity and return interval, relative to the life span of the tree species under management.

Where stand-replacing disturbances have a high probability of occurring within the typical life span of the dominant trees, single- or two-cohort structures best emulate natural patterns. Here, it is extremely important to distinguish disturbances that cause complete overstory mortality – creating true single-cohort stands – from events where *a few* members of the predisturbance cohort (large *legacy trees*) survive to form two-cohort structures. Some examples will clarify this distinction.

Single-cohort stands

Prominent examples of truly complete mortality events come from forests of fire-adapted species such as *Eucalyptus regnans* in southeastern Australia (Attiwill 1994b), boreal forests of spruces and pines (Cogbill 1985), and serotinous-coned conifers such as lodgepole pine (*Pinus contorta*). In these forests, individual trees usually do not survive the severe crown fires that naturally regenerate these forests, so single-cohort stands accurately emulate natural disturbances, as long as spatial patterns and the percentage of the landscape affected also resemble those of natural disturbances (Hunter 1993).

Other, extremely severe disturbances such as very hot fires in high-fuel situations which consume much of the forest floor (e.g., after an extensive blowdown or insect outbreak), and severe erosional events such as landslides and volcanic eruptions, also create single-cohort stands, although the stand initiation stage may well extend for many decades. Such events are arguably so rare and so destructive that one would not consciously perpetuate them. In other words, natural forces will produce enough of these without human assistance. However, important silvicultural analogues to these disturbances do exist, in the form of conifer monocultures that pioneer on abandoned agricultural lands. The prominence of this successional pattern in New England a century ago (Whitney 1994), where the only natural analogue would have been the post-glacial environment, testifies to the resiliency of these forest ecosystems. Indeed, these unnatural old-field monocultures have proven to be so productive that high-yield silvicultural systems essentially emulate this disturbance regime (Smith *et al.* 1997:163). Aspen forests in the Lake States, which originated after severe, repeated fires following logging of the old-growth pine, are now valuable enough that foresters consciously perpetuate a severe disturbance regime that was quite uncommon before human exploitation (Johnson 1995).

Two-cohort stands

Often, severe disturbances do not completely eliminate the mature forest. Virtually all the disturbed area regenerates to a new cohort, but

Fig. 2.2. (a) Fire origin lodgepole pine–spruce forest in central Alberta, Canada, showing clearcut blocks with 'islands' of unharvested trees reserved to simulate the natural fire pattern. (Robert Seymour photo)

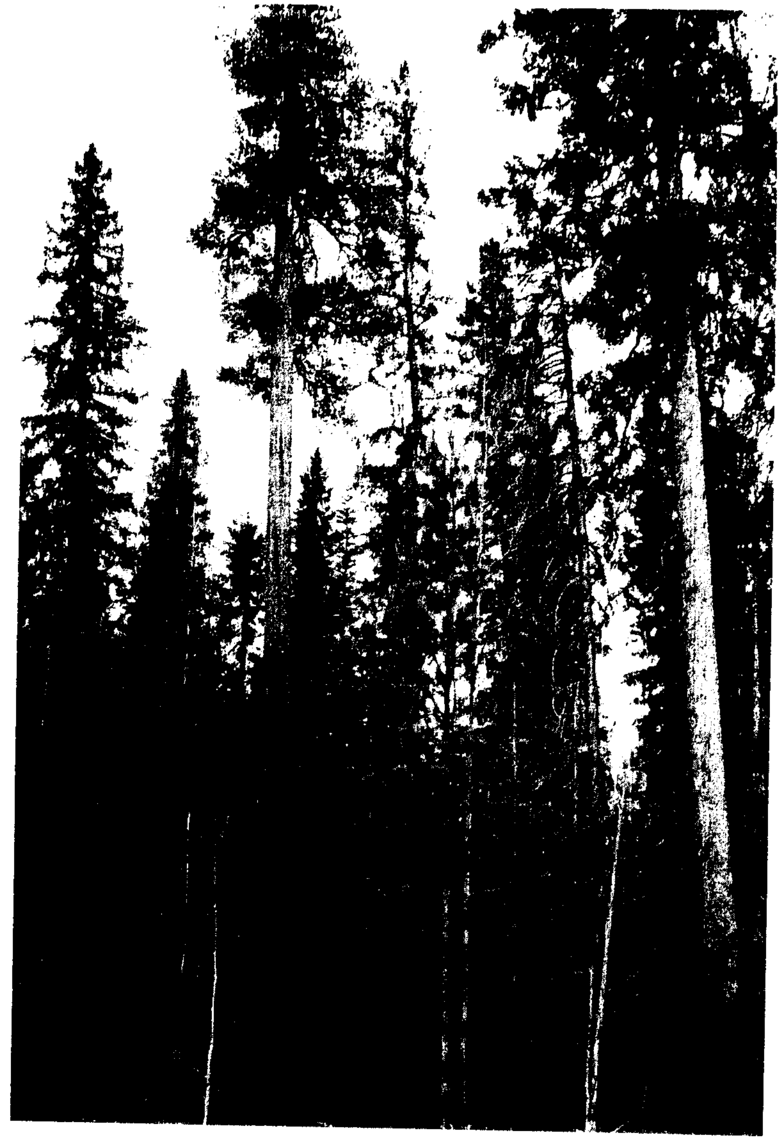

Fig. 2.2. (b) Reserved patches would eventually develop an old-growth structure like that shown, unlike the matrix which either burns or is harvested on a 80- to 100-year rotation. (Robert Seymour photo)

scattered veterans of the older, predisturbance cohorts remain. Two cases can be distinguished: survival as small patches under a few hectares in size (often too small to be considered separate stands), and survival as scattered, large individual stems. Such living *legacies* can be very important in such obvious ways as seed sources and refugia for recolonization, and very likely serve a myriad other ecosystem functions not fully understood (Chapter 11).

Both fire and stand-replacing windstorms tend to produce patterns in which trees survive in small patches owing to natural fuel breaks, microsite or topographic variation, chance occurrences, and other phenomena. Where such patches are large enough to be considered separate stands, then the disturbed area can be considered a separate single-cohort stand. Smaller patch sizes, however, should be treated as the older of two cohorts and the silvicultural system designed to perpetuate this structure. Here, studies of natural disturbance patterns are extremely valuable as landscape templates. For example, in northern Alberta Eberhart and Woodard (1987) found that fires of 41–200 ha (a typical harvest block) had 0.4 unburned islands per 100 ha, averaging 2.3 ha each. (Islands under 1 ha were not recorded.) Emulating this with a two-cohort 'clearcutting with patchy reserves' silvicultural system would be quite straightforward (Figure 2.2).

Cases where legacy trees tend to occur as individuals include: (a) wildfires in old-growth stands where older trees or certain species have thick bark and thus are more likely to survive the severe fires than others, and (b) insect outbreaks in mixed stands, in which tree species differ in susceptibility. Species exhibiting the first pattern include Scots pine (*Pinus sylvestris*) and coastal Douglas-fir (*Pseudotsuga menziesii*). A well-studied example of the second case is the spruce budworm (*Choristoneura fumiferana*) in the fir–spruce forests of the subboreal region of eastern Canada, in which the budworm periodically defoliates and eventually kills the susceptible mature balsam fir (*Abies balsamea*). Scattered black or white spruces (*Picea mariana*, *P. glauca*) tend to survive, however, thereby perpetuating a two-cohort structure dominated in numbers by the regenerating fir (Baskerville 1975; Figure 2.3). The exact choice of silvicultural systems here depends on the appropriate silvicultural *regeneration method* (see below) and should recognize the possibility of multiple successional pathways that maintain landscape species diversity (Bergeron and Harvey 1997). For example, one would likely use a 'seed tree with reserves' system for the fire-resistant species that establish after the fire, and a 'shelterwood with reserves' system for the fir–spruce forest that depends on advance regeneration.

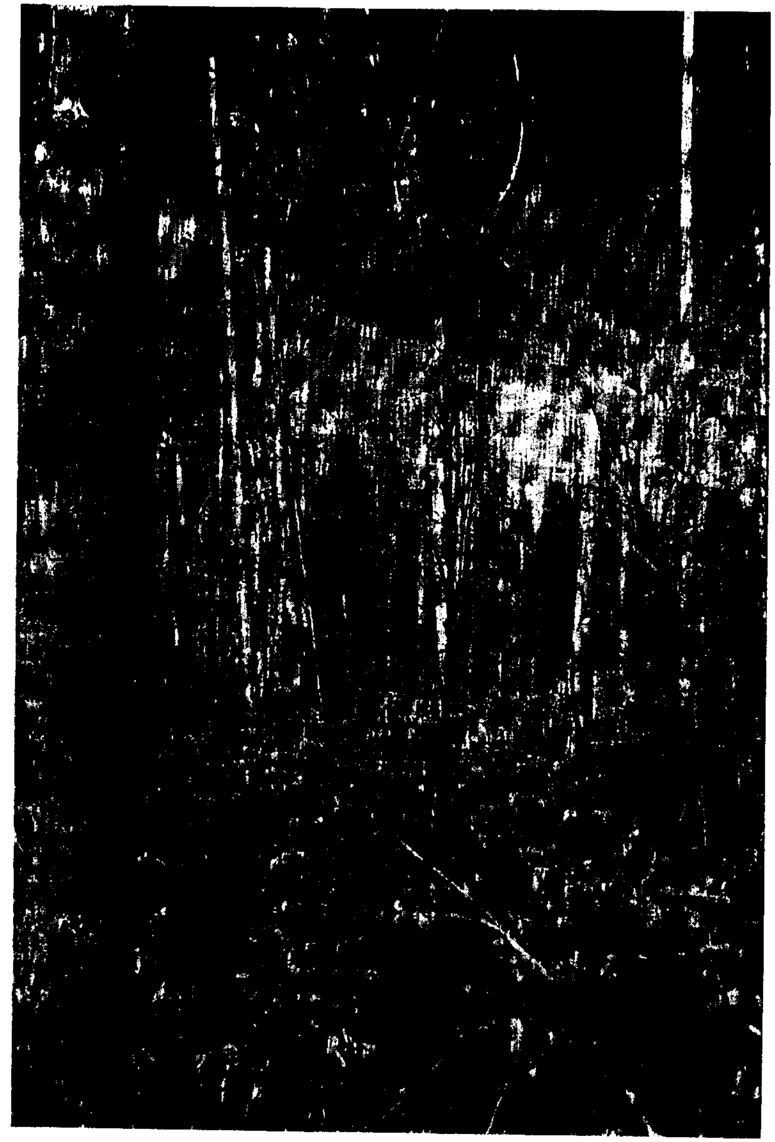

Fig. 2.3. Surviving individual white spruce legacy tree in a stand formerly dominated by balsam fir that was completely killed by spruce budworm defoliation during the 1970s outbreak in northern Maine, USA. Though not apparent in the photograph, the stand is well regenerated with advance conifer seedlings. (Robert Seymour photo)

Fig. 2.4. Old-growth ponderosa pine stand in eastern Oregon, USA; different tree sizes represent different cohorts. The preponderance of small trees is an unnatural condition induced by decades of fire suppression. (Robert Seymour photo)

Multi-cohort stands

Where stand-replacing disturbances are very infrequent (several times the life span of the late-successional tree species), but partial, gap-creating disturbances are dominant, then creating multi-cohort structures will most closely emulate this pattern. Multi-cohort structures are also most appropriate in dry conifer forests such as ponderosa pine (*Pinus ponderosa*; Figure 2.4) and inland Douglas-fir, where frequent patchy ground fires continually recruit new cohorts but prevent fuel buildup that would allow crown fires to kill the dominant trees (White 1985, Covington and Moore 1994, Agee 1991).

Some forest types exhibit more complex patterns that tend to grade into two-cohort structures. For example the Acadian forest characterized by the long-lived red spruce tends to develop multi-cohort structures on deep soils, sheltered locations, or in mixture with hardwoods (Figure 2.5). On poorly drained 'flats', however, or where the shorter-lived balsam fir dominates stand composition, chronic windthrow and spruce budworm outbreaks appear to prevent the buildup of more than two or three distinct cohorts (Seymour 1992, Cogbill 1996). Single-cohort structures evidently were quite rare in the presettlement forest, however, as return intervals for

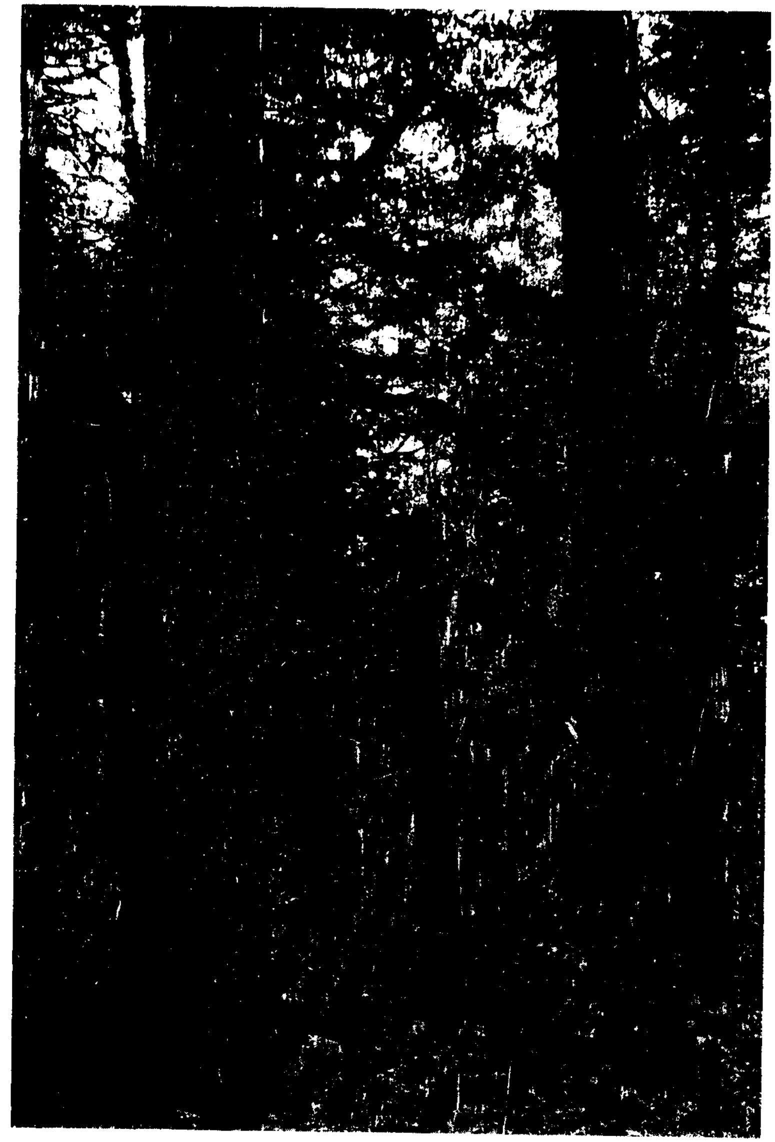

Fig. 2.5. Old-growth multi-cohort stand of red spruce in western Maine, USA. (Robert Seymour photo)

stand-replacing fires and windstorms ranged from 1000 to nearly 2000 years in northern Maine (Lorimer 1977).

Silvicultural literature about the treatment of multi-cohort (formerly uneven-aged) stands historically has emphasized the idealized balanced stand structure, in which a reverse-J-shaped diameter distribution is maintained indefinitely through carefully controlled *selection cuttings* (Nyland 1996, O'Hara 1996). Although common in Swiss mixed conifer forests, successful applications in North America over long time frames are rare and limited mainly to northern hardwood forests (Seymour 1995, Lorimer 1989) and the loblolly–shortleaf pine (*Pinus taeda–P. echinata*) forests on the Crossett Research Forest in Arkansas (Baker *et al.* 1996; Figure 2.6). Many silviculturists (e.g., Smith *et al.* 1997) have long considered the goal of maintaining balanced size or age distributions to be an unduly constraining feature of traditional selection silviculture. More importantly from the standpoint of disturbance regimes, such a finely balanced age structure rarely has a natural analogue. Rather, it serves mainly as a timber management construct aimed at sustaining frequent, equal harvests from individual stands.

Given the popularity of selection cutting among the public and many environmentalists, it is worthwhile to recount why this system became discredited within American forestry circles in about 1960, so that foresters do not reinvent a square wheel in well-meaning attempts to practise ecologically based forestry. Typical misapplications of multi-cohort silviculture are harvests that: (a) remove just large trees; and (b) reduce density uniformly throughout the stand to a level that regenerates a new cohort virtually everywhere instead of in discrete gaps. These practices usually result from financial pressures to cut too many large trees; few natural analogues for such a disturbance pattern exist (Lorimer 1989). Such cuttings actually are a crude form of two-cohort silviculture; they differ from the previous examples by virtue of the fact that the older cohort is represented by numerous medium-size trees rather than a few larger ones. Often short-sighted management causes such 'selective' cuttings (*sensu* Nyland 1996:502–8) to be repeated more frequently than the natural disturbance intervals, each time discriminating heavily against the oldest or largest trees. The unfortunate result is typically a reduction in age, size, and species diversity, with the cohort structures becoming more uniform over time and economically valuable species being lost.

Importance of legacy trees

The distinctions among different naturally occurring age structures highlight the point that vegetation (living and dead) that survives the dis-

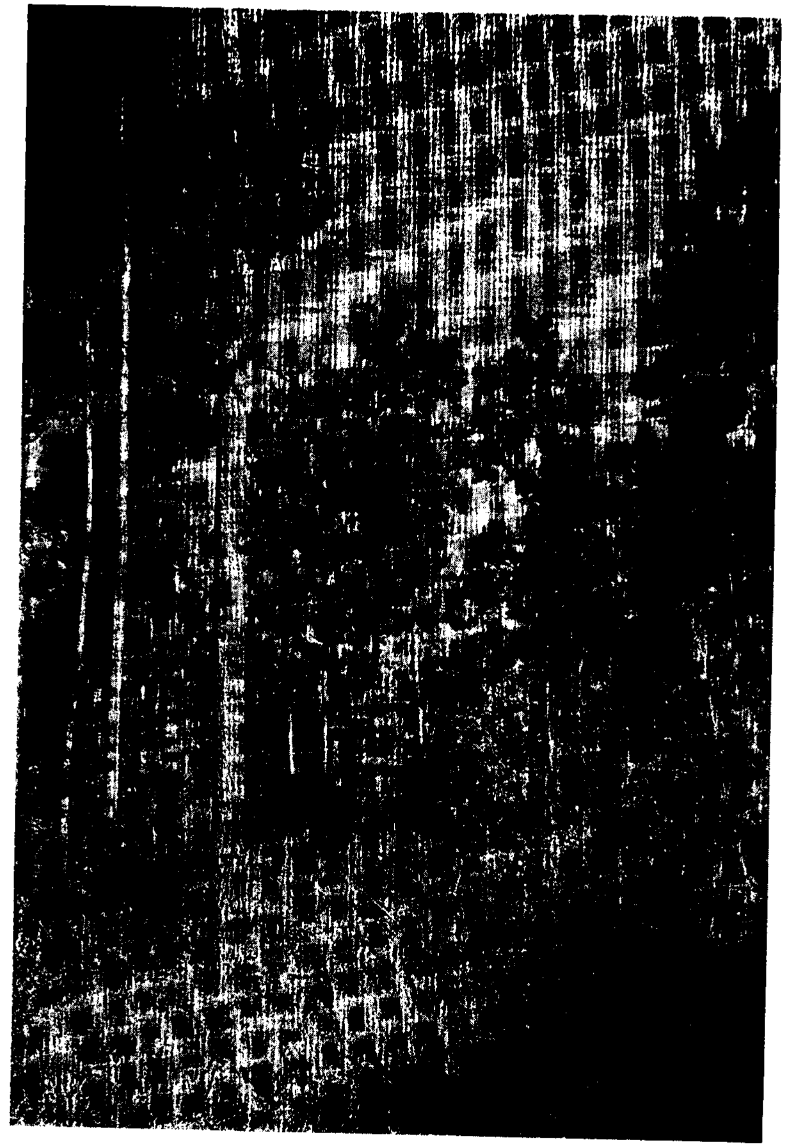

Fig. 2.6. Multi-cohort stand of loblolly and shortleaf pines that has been under selection management since the 1930s, Crossett Experimental Forest, Arkansas, USA. (Robert Seymour photo)

turbance is the critical issue in creating the appropriate silvicultural analogue. To be ecologically justified, disturbance mimicry must be more than superficial; it cannot be applied selectively just where it happens to suit some timber-management purpose but ignored where it is inconvenient or costly. For example, too often in the past, foresters have endorsed complete clearcutting as emulating all kinds of severe natural disturbances, even though this argument is valid only in certain specific cases of truly lethal mortality events. In other cases where cuttings were not complete, reserve trees were chosen not because they resembled those that would have survived natural disturbances, but simply because they were too small. Because the large Douglas-firs and pines that survive disturbances invariably are the most valuable trees in the stand, serious conflicts can arise between financial demands and ecological integrity.

Dead trees are just as important to retain as part of the biological legacy as are living ones (Chapter 10). In fact, a very important reason for reserving mature, *living* individuals is to ensure that during the next generation of the young cohort, there will be a continual supply of large *dead* trees for cavities and other habitat (Woodley and Forbes 1997). As such, reserve trees may have merit even in otherwise very artificial systems such as high-yield plantations.

An excellent illustration of the dangers of overly superficial mimicry comes from Hutto's (1995) study of bird communities in 34 post-fire sites in western Montana and Wyoming. Hutto noted that recent U.S. Forest Service 'green retention' practices, which leave scattered *living* lodgepole pines after clearcutting but destroy most standing dead stems, do not emulate natural patterns as well as complete clearcuts that leave all standing *dead* snags. To emulate natural fire regimes in this forest, Hutto advocated leaving some living trees then killing them after harvest with a prescribed fire, in order to ensure adequate nesting and foraging habitat for certain birds that depend on this particular post-fire structure.

HARVEST TIMING

Rotations for single and two-cohort stands

Where stand-replacing disturbances dominate and single- or two-cohort stands are the prevailing structure, foresters must decide how long the stands will be allowed to develop between regeneration harvests – the *rotation*. Under sustained-yield timber management, rotations are set using biophysical or economic criteria that maximize commodity outputs or present net worth (Davis and Johnson 1986). It is rare to encounter a

forest where this is actually done routinely, however, due to unbalanced age structures, mill demands, and other overriding factors. If the forest composition is simple and a single disturbance agent predominates, then ecologically based rotations clearly depend on the disturbance return interval. Because this is largely an issue of setting an appropriate age structure at the landscape scale, it will be discussed later under that heading.

In situations where multiple disturbance agents affect forests of mixed species with very different natural life spans, it is a gross oversimplification to view the rotation as a single number. A variety of species may initiate during the same year after a severe stand-replacing disturbance, but these same species will reach their average life spans at different times, perhaps centuries apart. Community composition will thus naturally vary, until all pioneering individuals die off and autogenic succession takes over, or another stand-replacing disturbance occurs. This initial floristics view of plant succession (Egler 1954) has been widely adopted by silviculturists as a model of natural development for stratified, mixed-species stands (Oliver and Larson 1996, Smith *et al.* 1997:164).

If species mature at very different rates in the same stand, a single rotation – i.e., removing most or all of the stand in a single regeneration cutting – is clearly inappropriate. If the rotation is based on the shorter-lived members of the community, later-successional individuals may never reach maturity, and the next generation community composition will be simplified in favor of the early-successional species. North America has numerous examples of forests that have been profoundly altered by unusually severe stand-replacing disturbances at unnaturally short intervals. The original pine forests of the Lake States that are now largely dominated by aspen (*Populus tremuloides*), and the northern hardwood–hemlock (*Tsuga*) forests of the Allegheny plateau in Pennsylvania that are now cherry–maple (*Prunus–Acer*) forests, are two prominent examples. Conversely, setting the rotation at the life span of the longest-lived species would emulate disturbance well, but would sacrifice substantial economic production from the shorter-lived species. A reasonable compromise in such stands is to use multiple, species-specific rotations when incomplete removal cuttings are made. Such silvicultural systems can be effective in maintaining species diversity while also restoring single-cohort forests to their naturally diverse, multi-cohort structures. Smith *et al.* (1997:391–419) and the chapters in Kelty *et al.* (1992) outline many creative approaches to this common problem.

Cutting cycles for multi-cohort stands

Unlike single- and two-cohort systems, multi-cohort stands are never affected by severe disturbances and thus have no rotation *per se*. Yet, the basic principle is the same: periodic partial harvest cuttings must regenerate cohorts at approximately the same rate as the relevant partial disturbance regime. This is quite a different approach from the typical way multi-cohort stands are managed, which focuses on sizes and volumes of trees harvested. For example, Runkle (1985) noted that disturbance frequencies for a wide variety of agents and severities ranged only from 0.5% to 2% per year throughout temperate forests. This equates to average return intervals of 50–200 years, which in turn represents the average time an individual tree would be expected to reside in the canopy – essentially equivalent to the *rotation* of a single-cohort stand.

To emulate this regime with multi-cohort silviculture, one sets a *cutting cycle* (time between silvicultural disturbances for the stand *as a whole*), and multiplies by an appropriate annual disturbance frequency to obtain the total area to be created in canopy gaps at each stand entry. The inverse of this number equals the number of cohorts to be maintained (Nyland 1996:200). For example, to emulate a 1% frequency (= 100 year return interval) on a constant 20-year cutting cycle, one would need to limit gaps to 20% of the stand at each entry, with the aim of maintaining a 5-cohort stand. Additionally, one would need to ensure that, on the average, trees are harvested after approximately 100 years canopy residence time. Holding the cutting cycle and disturbance rate constant over time, as in this example, produces a perfectly balanced age structure within the stand. Of course, nature is almost never so perfect and the cutting cycle and disturbance frequency could be varied (in complementary ways) to produce a more irregular structure with fewer cohorts that occupied different amounts of area. Legacy trees of long-lived species should also be reserved well beyond the 100-year limit to replenish large cavities and woody debris.

SILVICULTURAL REGENERATION METHODS

Well before the discipline of ecology even existed, foresters were devising regeneration methods by close observation of vegetation response to disturbances. Natural patterns and their silvicultural analogues can be usefully grouped into two distinctly different categories: releasing disturbances such as wind or insect outbreaks that kill from the 'top down,' and those (mainly fire) that kill trees from the 'bottom up' (Smith *et al.* 1997:162–4).

The shelterwood method

Releasing disturbances, even those that kill the entire overstory, tend to favor mid- to late-successional, shade-tolerant species that persist as advance regeneration or perennial rootstocks in the understory. The clear silvicultural analogue here is the *shelterwood* method, the defining feature of which is the establishment of seedlings under protective overstory cover *before* the overstory is removed. By varying the timing of establishment and removal cuttings, as well as the overstory density during the regeneration stage, conditions can be created to favor virtually all but the most disturbance-dependent species. By varying spatial pattern of the cuttings and leaving permanent reserve trees, vertical and horizontal diversity can be greatly enhanced compared with more uniform treatments.

The uniform shelterwood method traditionally has been associated with single-cohort structures, because the regeneration period occurs during a relatively short period near the end of the rotation of the older cohort. Smith *et al.* (1997:347–63) now refer to shelterwood cuttings as an example of a 'double-cohort' system, presumably because the old and new cohorts overlap, however briefly, during the regeneration phase. This is unconventional usage of the two-cohort terminology; here and elsewhere (Helms *et al.* 1994), two-cohort stands are those in which the older cohort is more or less permanently represented in the form of *reserve trees* left after the rest of the overstory is removed. Irregular shelterwood systems with more protracted regeneration periods fall in a gray area between the uniform and distinctly two-aged cases. An excellent example is the German Femelschlag (irregular group shelterwood) method (Spurr 1956), in which the regeneration period may extend up to half the rotation.

Seed tree and clearcutting methods

Fires which kill from the ground up tend to favor two categories of species: shade-intolerant pioneers that establish best in open environments with exposed mineral soil; and species which reproduce vegetatively as stump sprouts or root suckers. The appropriate silvicultural analogue depends on the specific source of propagules. Where severe fires leave only scattered large trees that are important sources of seed for a new cohort – such as the Douglas-fir and Scots pine examples discussed above – the *seed tree* method is clearly most appropriate. Smith *et al.* (1997:347) treat seed tree and shelterwood cuttings similarly; the main distinction is that seed tree cuttings do not provide shade and protection to the new seedlings.

Where fires or other severe disturbances kill virtually all vegetation, the appropriate silvicultural analogue is the clearcutting method. Unlike the terms 'seed tree' and 'shelterwood' which accurately convey the ecological intent of the cuttings, clearcutting is a timber harvesting term that has a wide variety of meanings within forestry. In a strict silvicultural sense, clearcutting applies only to cases where seedlings develop *after* the complete harvest. Seeds can come from surrounding stands, the crowns of trees harvested, or the seed bank in the forest floor. Where new plants arise from vegetative sources (e.g., stump sprouts), this is known as the coppice method. Great semantic confusion arises because clearcutting as a *harvesting* term is used by both foresters and the public to describe a wide variety of operations where most or all the merchantable timber is removed in a single entry. These include removal cuttings in the shelterwood method, seed tree cuttings, and heavy selective cuttings ('commercial clearcuttings') with no silvicultural intent. Silviculturists have thus found it necessary to use the modifier *true* or *silvicultural* clearcut when speaking of a regeneration harvest (Smith *et al.* 1997:327–8). When the intent is to release advance regeneration, the correct term is *overstory removal* cutting, even if there were no prior harvests and the advance seedlings are of purely natural origin.

Selection silviculture

The *selection* regeneration method is yet another source of confusion. Technically, this method applies to any type of harvest designed to create regeneration under a multi-cohort silvicultural system. From an ecological standpoint, most selection cuttings resemble the shelterwood method; the only difference is that with selection, only a small portion of the stand is regenerated in a single entry in order to perpetuate the multi-cohort structure.

Artificial regeneration

If one were beginning to implement ecological forestry in a forest unaffected by past human exploitation, then there would be no need for any 'artificial' practices. Unfortunately, centuries of human use has often reduced tree species diversity on scales that make natural reintroduction unlikely. Here, planting and direct seeding can play a useful role in augmenting natural methods; these practices in the context of restoration forestry are covered in Chapter 15.

A COMMENT ON SILVICULTURAL TERMINOLOGY

The preceding section highlights the fact that silviculture is in a state of evolution. Important new terms are replacing old, because silviculturists have become frustrated with the *prescriptive* implications of the old language and its inability to *describe* creative, ecologically based practices and systems. Foresters should not worry about whether a particular practice has a convenient traditional pigeonhole somewhere in the silviculture textbooks. The key is to create the appropriate overstory and microenvironmental conditions for tree growth, seedling establishment and habitat for other species, within the confines of the chosen cohort structure. If traditional silvicultural terminology cannot readily describe novel or unconventional combinations, this is a weakness of the terminology, not of the person prescribing the treatment or the treatment itself.

Landscape-level decisions

HARVEST LEVELS AND AGE STRUCTURES

The volume of wood harvested annually is probably the single most important decision affecting a forest property or region. Forest managers have typically used one of two forest regulation methods: *volume control*, typically associated with forests under uneven-aged management; or *area control* used in forests of single- or two-cohort stands. Area-based approaches have one outcome in mind: a perfectly rectangular age distribution, with equal areas in each age class up to the rotation, and none older – the so-called 'normal' or perfectly regulated forest. As computer technology has evolved, complex harvest scheduling models have emerged which combine both methods. Forest regulation under sustained-yield timber management has always attempted to maximize wood volumes harvested over time, subject to long-run sustainability constraints that may or may not include ecological parameters.

The best way to determine an ecologically sustainable harvest level is with an area-based approach that attempts to maintain (or recreate) a natural landscape age structure. Under this modification of area regulation, the harvest is the sum of all timber volumes derived by applying the appropriate silvicultural systems to the appropriate areas at a sustainable pace. It is *not* calculated as function of actual forest growth or growing stock volumes, as is done with various volume control methods. First, we will consider the simpler case of stand-replacing disturbances in forests composed of single-cohort stands, then extend this reasoning to partial disturbances in forests of predominantly multi-cohort stands.

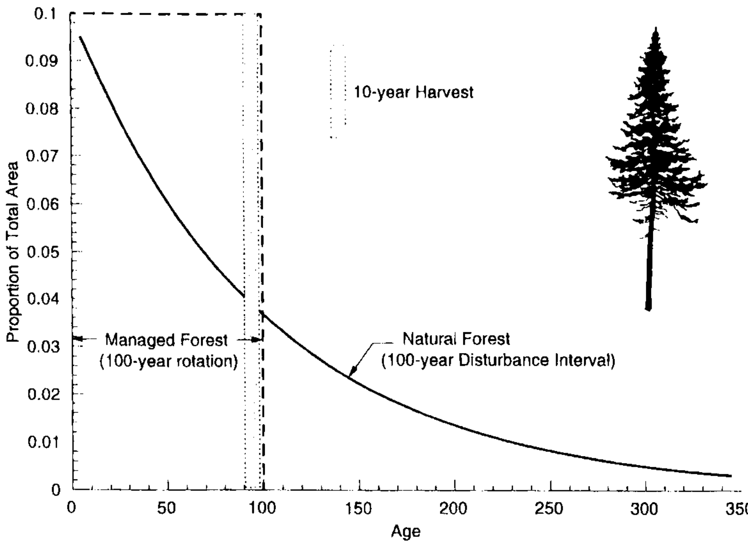

Fig. 2.7. Comparison of age structures resulting from a single rotation of 100 years vs. the natural distribution produced by random disturbances (after Van Wagner 1978).

Stand-replacing disturbance regimes

Simply setting a managed forest rotation equal to the disturbance interval does not accurately emulate a natural disturbance regime. To understand why, compare a forest with a 1% annual stand-replacing fire disturbance regime with a forest managed on a 100-year rotation (Figure 2.7). In nature, the quasi-random spatial pattern of disturbances results in some stands burning repeatedly on short cycles while others escape for long periods. Under certain conditions (see Van Wagner 1978), such a forest will approach a negative exponential (not rectangular) age distribution:

$A(x) = p \exp(-px)$
where $A(x)$ = area of age x; and
p = annual disturbance frequency = inverse of the return interval.

Under sustained-yield timber management, no stands 'escape' harvest and reach old age, nor are any young ones intentionally disturbed. Importantly, the area burned or harvested (and thus regenerated) annually is equal in both forests, but the natural forest has twice the mean age as the managed one. Over 37% of the natural forest is older than the 100-year

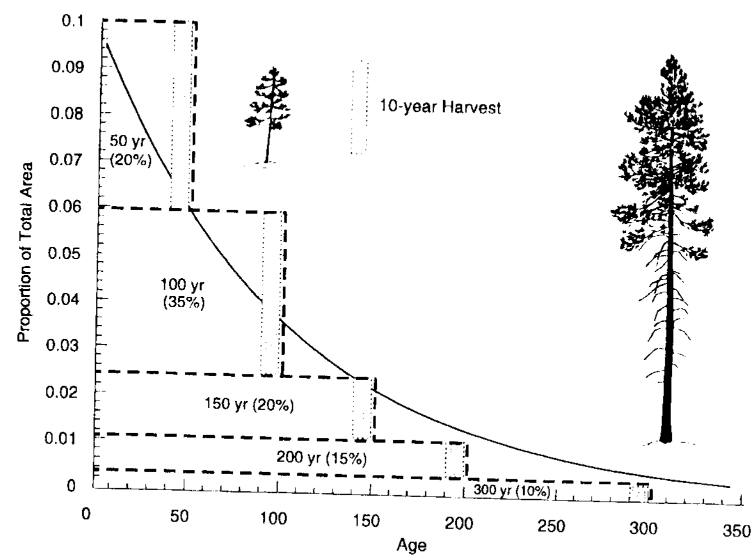

Fig. 2.8. Approximating the natural, negative exponential landscape age structure by managing different portions of the forest on varying rotations.

timber rotation. If the disturbance is 'shared' equally among natural causes and harvest (0.5% each), the resulting age structure will simply be a truncated exponential that resembles the timber-regulated forest much more closely than the natural structure, with no old-growth stands (Van Wagner 1983). Clearly, the problematical issue in mimicking natural patterns is the 'tail' (Chapter 4) of old growth that does not exist with harvesting.

The most straightforward way to emulate this pattern under management would be to allocate different portions of the forest to successively longer rotations, ranging from age 50 for short-lived species up to 300 years for late-successional habitat. This would appear as a series of rectangular distributions, each stacked on top of one another (Figure 2.8). For example, we could emulate the example above by harvesting 10% of the forest on a 300-year rotation, 15% at age 200, 20% at age 150, 35% at age 100, and 20% at age 50. Sixteen percent would be harvested at a younger age than the classic normal forest (Figure 2.7), whereas 45% would be managed on rotations longer than 100 years. About 10% of the forest would be harvested and regenerated each decade, distributed among age classes as shown, just as in the natural situation.

The foregoing example presumes that all natural disturbances can be preempted through management. In reality, some natural disturbances will occur, creating early successional forests, so the challenge of managers is to *complement*, not replace, the natural pattern. If substantial areas of older stands existed in ecological reserves where disturbance patterns were not altered by harvesting, then the age structure of the managed forest could be configured to complement that of the reserves (B.C. Ministry of Forests 1995). However, if reserves were small or isolated such that a single disturbance could eliminate the old forests completely, this approach could ultimately prove unsatisfactory.

Yet another option would be simply to let some disturbance occur and salvage mortality afterward (with due consideration for biological legacy issues), thereby letting natural events control the age structure. Although appealing ecologically, such an approach would be highly impractical in situations where the forest is fully utilized and a stable annual cut is needed, but the area annually affected by disturbance is highly variable.

Partial disturbance regimes

Where partial disturbances dominate, only a small portion of the landscape naturally occurs in single-cohort stands to which a single age can be readily assigned. Here, foresters should consider the forest not as a distribution of clearly defined age classes, but as a matrix of multi-cohort stands, each of which is continually regenerating in relatively small patches. Studies of disturbance regimes in such forests (e.g., Runkle 1991, Frelich and Lorimer 1991, Frelich and Graumlich 1994, and Dahir and Lorimer 1996 for northern hardwood temperate forests) provide excellent management templates to design multi-cohort systems. The key issues are: (a) average disturbance frequency, or the area regenerated annually within the stand, (b) the size distribution of gaps, and (c) how the gaps are configured spatially. When such parameters are known, incorporating them into multi-cohort silvicultural systems is conceptually straightforward. Just as with single-cohort systems, legacy issues must also be kept in mind by allowing some trees to reach their natural lifespans. For example, in a northern hardwood forest that averages a 1% partial disturbance frequency, one must allow some sugar maple trees to exceed 100 years of age, just as they would in nature.

SPATIAL PATTERNS OF HARVESTS

Matching the temporal patterns and intensity of harvests and natural disturbances may be the key issue in ecological forestry, but we

also need to consider spatial patterns. Specifically, we need to ask: How do the size, shape, and distribution of harvests compare with the spatial characteristics of natural disturbances? This question is relevant from a biodiversity perspective for at least three reasons. First, a large stand represents different habitat for some species than a small stand. Second, stands with irregular shapes have relatively more edge than regularly shaped stands and edges represent a different type or quality of habitat for some species. Finally, the spatial distribution of stands can affect the ability of organisms to move across the landscape (e.g., a carnivore patrolling its home range, or a plant propagule dispersing). All of these issues are particularly germane when managing landscapes that are a mosaic of single- and two-cohort stands. In landscapes covered by extensive multi-cohort stands, the question is probably less critical because it is unlikely that organisms are highly sensitive to the spatial configurations of small groups of trees.

The spatial patterns generated by natural disturbances can be complex. At a minimum they will be shaped by: (a) the unique attributes of a specific event (e.g., velocity of a particular hurricane or wind direction during a particular fire); (b) the topography of a given site (e.g., is there a hill to provide a wind break or a river to provide a fire break?), and (c) the vegetation itself (e.g., are the trees relatively vulnerable or invulnerable to being burned, blown over, killed by insects, etc.?). This variability might seem terribly daunting for foresters trying to emulate it with their harvest plans, but in a sense it also provides considerable latitude. The key is to understand the general pattern of past natural disturbance events and to use this as a template for laying out harvests. It is preferable if this information can be specific for a particular landscape. However, it is not possible to achieve perfection. In the big picture, one is always trying to hit a moving target; for example, global climate change will always be shifting the patterns of fires and wind storms (Clark 1988). Certainly, most forest operations have enormous scope for improvement when it comes to emulating the spatial patterns of natural events (Hunter 1993). Many landscapes managed for timber production look like someone has been at work with a square cookie cutter of about 10 hectares, punching a regular pattern of holes across the landscape (Figure 2.9). Spatial issues are covered in detail in Chapters 4, 5, 6, and 7.

Balanced forestry

In this book many prominent forest scientists have been asked to formulate a working hypothesis of how to maintain biodiversity in

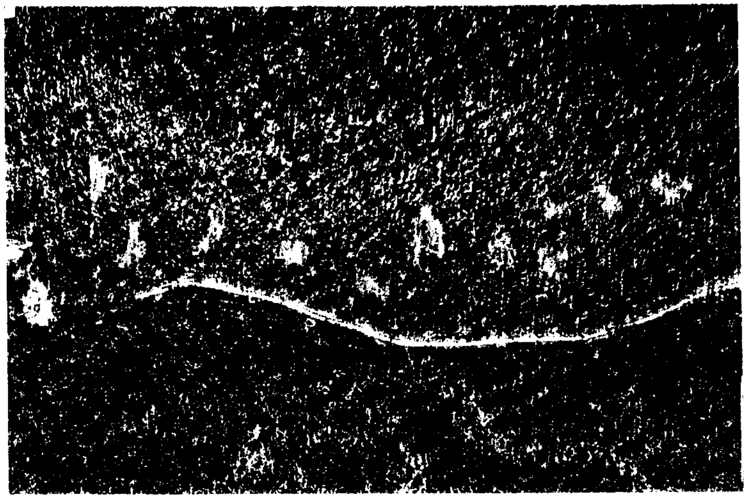

Fig. 2.9. Landscapes dominated by single- and two-cohort stands are often harvested in a series of patches that are quite uniform in size and shape and regularly distributed. This pattern bears little resemblance to the spatial pattern generated by natural disturbances. (The example shown here comes from the shortleaf pine forest type on the Ouachita National Forest, Arkansas, USA. Jim Guldin, USDA Forest Service photo.)

managed forests. The implicit assumption here is that forest ecosystems function to conserve biodiversity just fine on their own. To put this another way, a conservative person must assume that, until proven otherwise, any human manipulation represents a compromise between ecological integrity and society's demand for forest products. The disciplines of ecology and conservation biology help us to understand the biological consequences of human manipulation, but it falls upon the profession of forestry to balance these often-conflicting demands in practice. As human populations grow in numbers and affluence, this balancing act becomes more and more difficult, and it becomes clear that no single approach to forestry will meet all of society's needs.

It was this recognition – that society's competing demands were on a collision course – that led us to propose a fundamental change in our home state of Maine, where there is little public forest land and where industrial landowners have managed over 3 million contiguous hectares under a mixture of custodial, extensive, and sustained-yield approaches for nearly a century. At first, the situation seemed hopeless. Conservation biologists had become concerned that only very small, non-representative areas were protected from harvesting, and thus there were few credible

benchmarks against which to judge ecological consequences of forest management activities. At the same time, forecasts of timber shortfalls suggested that any agenda for withdrawing forest land from harvest would meet stiff opposition on economic grounds.

While writing a review of ecological forestry in the Acadian spruce-fir forest (Seymour and Hunter 1992), we realized that simply replacing extensive forestry with ecological forestry – while clearly a very positive step from the standpoint of biodiversity – would not solve the larger problem. Setting aside adequate, representative areas for ecological reserves without reducing timber harvests would require a compensatory increase in production silviculture. Such a scenario is feasible in Maine because extensive forestry has produced such low timber yields; substituting production silviculture can raise per-hectare yields from threefold to fivefold (Seymour 1993). Consequently one could, in theory, set aside 3–5 ha of ecological reserves for every hectare shifted into production forestry, with no net loss in overall timber production. This rationale has also been advocated as a global forest conservation strategy (Gladstone and Ledig 1990, Sedjo and Botkin 1997), in grassland and aquatic ecosystems (Hunter and Calhoun 1996), and is supported by economists (Vincent and Binkley 1993) and some conservation geneticists (Libby 1993).

Where such increases in timber yields are possible, it is easy to see that timber lost from setting aside 10% of the landscape in ecological reserves could, in the long run, be replaced by timber from a small area of land dedicated to production silviculture. In this scenario, ecological forestry would supplant extensive forestry, and constitute the predominant matrix into which reserves and production forestry would be embedded. We call this vision a landscape *triad*, in order to highlight the three, fundamentally different, objectives to which forest land would be dedicated on the landscape. Designing and managing such a landscape that attempts to provide for all societal demands would be an example of *balanced forestry* (after Kimmins 1992), a term chosen to acknowledge explicitly that all uses have inherent worth and thus must be balanced against one another in practice.

The triad does not, as some have inferred, suggest an *equal* allocation; exact values in each sector must come from case-specific analyses. For example, consider the United States South where most of the landscape was once cleared for agriculture and land is inherently quite productive. Here, the forest industry is actively converting abandoned agricultural lands to high-production loblolly pine plantations, and it is probably inevitable that the landscape will be dominated by production forests. Nevertheless, ecologically viable blocks and corridors of natural forest, wetlands and riparian zones could be established to maintain critical biodiversity functions. Alternatively, rather than convert all degraded lands to commodity production, innovative silviculture could be used to restore more natural communities (e.g., longleaf pine) where necessary to ensure landscape connectivity and maintain critical habitats not provided by the production-oriented matrix. In contrast, consider New Zealand, where plantations of exotic species are extremely productive and native forests are difficult to manage on short rotations. Here, the approach has been to produce almost all timber from plantations and to set aside the vast majority of the remaining native forest.

IMPLEMENTING A LANDSCAPE TRIAD

The first and perhaps most critical step in practicing balanced forestry is to accept and support the premise that some of the landscape must be left alone. This is a difficult step for many foresters who have been inculcated with a 'manage everywhere' mentality that assumes virtually any forest can be improved through careful human intervention, and that good silviculture must be good for biodiversity. This may be true, but given our current state of ignorance about biodiversity, a conservative ethic dictates that we regard it as an hypothesis, not an established fact. Importantly, this hypothesis cannot be tested without adequate experimental controls from which to learn and adapt management accordingly; hence the need for a scientifically designed system of reserves (see Chapter 16).

Once a system of reserves is in place, one needs to assess the commodity-production potential of the unreserved landscape. Where timber demands are relatively low, intensive application of ecological forestry will likely sustain them. If, however, demands are relatively high, it is likely that some portion of the landscape will need to be managed under production silviculture to offset the reserves. Foresters would naturally seek to establish high-yield plantations on the most productive sites. Conservation biologists would advocate locating them where they would do the least damage; i.e., the most degraded communities (such as lands formerly converted to agriculture, repeatedly high-graded, etc.). Often these are the same lands, so decisions should be straightforward. If plantations do not disrupt key features such as landscape connectivity and riparian zones, are limited to a minority of the total forest, and are managed on sufficiently long rotations without whole-tree harvesting in order to maintain the integrity of nutrient cycles, then there should be little cause for alarm.

DO WE REALLY NEED PLANTATIONS?

Advocates of ecological forestry often question whether growing trees under an agricultural paradigm is really necessary in order to achieve high timber yields. They argue that by managing natural forests more intensively and sensitively, both timber yields and ecological values could both be increased. We accept this premise up to a point; it is certainly true where sophisticated ecological forestry replaces low-budget extensive forestry. However, the success of production silviculture, as in intensive agriculture, derives from the fact that canopy leaf areas are carefully controlled to maximize carbon fixation in merchantable stemwood of economically useful species, all of which is eventually harvested. Ecological forestry, in contrast, demands that a significant portion of the carbon fixed by photosynthesis be left on site in the form of various structural elements, and further, relies on manipulating canopy structure for purposes other than maximum leaf areas.

A related issue that has come under intense discussion in Europe and parts of North America is whether certain features of ecological forestry can be incorporated in production silvicultural systems. Clearly, practices such as leaving dead snags and conserving downed woody debris which do not result in competition to the crop trees, should be used wherever possible, regardless of the stand-level objective. Lengthening rotations may be another valuable option, especially if doing so actually results in higher production of more valuable products (Peterken 1996:425–63). Leaving living reserve trees, which may compete with the developing stand, is a more problematical issue. Is there some sort of hybrid silviculture that achieves both high timber output and high levels of diversity? We have few answers, though there is reason to be skeptical, for the more an ecosystem is simplified through production silvicultural practices, the more likely we are to lose some elements of biodiversity that depend on its natural complexity. However, this is not to discourage creative foresters from attempting innovative modifications to plantation silviculture, for we will undoubtedly learn much from this experience (Chapter 12).

Summary

Forestry has evolved many different models such as custodial forestry, sustained-yield timber production, multiple-use forestry, production forestry, and extensive forestry. One of the newest forms focuses on maintaining the ecological integrity of forest ecosystems; it is known by many terms – we call it 'ecological forestry'. The central axiom of ecological forestry is that any manipulation of a forest ecosystem should emulate the natural disturbance patterns of the region prior to extensive human alteration of the landscape. This axiom is based on the assumption that native species have evolved under these natural disturbance regimes and will be better able to cope with human-induced disturbances such as logging if these are designed to imitate the key characteristics of natural disturbances: the return interval between disturbances, disturbance severity, and the spatial pattern of disturbances.

Stand structures maintained under ecologically based silvicultural systems can be either single-cohort, two-cohort, or multi-cohort, depending on the disturbance agent, its severity, and return interval. Emulating disturbances at the stand level should pay close attention to providing biological legacies (typically large, old trees and dead snags) similar to those that survive natural disturbances. Silvicultural regeneration methods should also be patterned after natural disturbance processes; the critical difference is whether seedlings originate prior to, or after, the disturbance. At the landscape level, the crucial point is to regenerate areas of new cohorts at approximately the same rate as the natural disturbance cycles would have, and to ensure that the natural diversity in age and structure are conserved and maintained. To emulate large-scale stand-replacing disturbances, single- or two-cohort stands managed under several different rotations should be employed. Special attention should be given to ensuring that some forests reach at least twice the age of the average disturbance interval. In forests with patchy, partial disturbances dominated by gap processes, having multi-cohort stand structures managed under variable cutting cycles best emulates natural patterns.

Ultimately, human demand for timber is likely to make it impossible to practise ecological forestry in all forests, especially if we want to set aside larger areas of forests to serve as ecological reserves. This reality will likely dictate that we practise balanced forestry, represented by a triad of production forestry and ecological reserves embedded in a matrix of ecological forestry.

Further readings

Traditional silviculture texts such as Smith (1962), Smith *et al.* (1997), Nyland (1996) and Matthews (1989) are good starting points for understanding the context and evolution of ecological forestry. Peterken (1981) and Hunter (1990) are early syntheses of ecological forestry ideas; more recent treatments include Alverson *et al.* (1994) and Kohm and Franklin (1997). Recent forest ecology texts such as Perry (1994), Kimmins (1997), and Barnes *et al.* (1998) provide excellent coverage of biophysical process and how they interact with disturbances.

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