# Spatial interactions and forest management: policy issues

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## Abstract

Biodiversity conservation policies in forest systems should taken into account the effect on ecosystem services of spatial interactions within the forest landscape. These interactions between landscape components may imply the existence of spatial externalities due to the interdependence between users/managers' decisions. Thus, management decisions of any one landowner may have consequences for the decisions of neighbouring landowners. In this paper, we review the way in which spatial interactions have been analysed in forest economics and management literature, and analyse the implication of spatial interactions in forest landscapes.

## 1 Introduction

In the decade since the signing of the Convention on Biological Diversity (CBD), global efforts to conserve biodiversity in forest ecosystems have focused on the species-rich humid tropical forests.[1] Conservation in these areas has consisted of both *in situ* conservation measures involving the designation of protected areas, and *ex situ* measures, including germ plasm preservation in zoos, arboreta, seed banks, tissue cultures and genomic libraries. The focus on species rich areas – the so-called biodiversity hotspots – is aimed at the protection of a global public good: the global gene pool (Heywood 1995, Millennium Assessment 2003).

Historically, however, biodiversity conservation in forest ecosystems has been addressed at the local level for strictly local reasons. In productive forests, for example, biodiversity conservation has implied protection of enough interspecific and intraspecific diversity to underwrite the productivity of the system. Productivity in such systems depends on a number of local ecosystem services such as flood control and water supply, waste assimilation, recycling of nutrients, conservation and regeneration of soils and so on. Forest biodiversity

supports this set of services over a range of environmental conditions. That is, forest biodiversity protects the resilience of forest systems (Perrings and Gadgil 2003).

Two aspects of the problem are especially important. First, biodiversity conservation at the local level is a local public good. Because it is a public good, users will typically ignore the social costs and benefits of their actions unless there are incentives to do to otherwise. The incentives in this case generally stem from the nature of the property rights or the regulatory and management regimes. However, because biodiversity conservation is a local public good there are generally some rights of exclusion. That is, the access regime tends to be regulated rather than open.

Second, the relative importance of genetic, species and ecosystem diversity tends to be rather different at the local level than at the global level. At the global level the primary concern is with the protection of the global gene pool, so biodiversity conservation is focused on preservation of genetic material either *in situ* or *ex situ*. At the local level, the primary concern is with the functional diversity of species and the interaction between species and ecosystem types in the provision of ecosystem services. Biodiversity conservation therefore tends to focus on maintenance of a patch structure or mosaic of land uses. Because of this, a key element in local conservation strategies is the regulation of spatial interactions between elements in the mosaic.

It follows that local biodiversity conservation in forest systems requires decision-tools that explicitly account for the effect on ecosystem services of spatial interactions within the forest landscape. In forests that are managed by multiple users these spatial interactions may give rise to spatial externalities. Spatial externalities due to ecological (economic) interactions between landscape components imply the interdependence between users/managers. Gottfried et al. (1996) refer to this as "economies of configuration", which they define as the effect of spatial patterns on the output mix and output costs. In forest systems it implies that outputs depend both on ecological relationships between stands and on the spatial patterns emerging from land use decisions.

This paper considers the policy implications of spatial interactions in forest landscapes. In the next section it addresses the linkage between biodiversity conservation and forest landscape management as reflected in the forestry literature. Next we discuss the way in which spatial interactions in forest landscape management have been analysed in the forest economics and management literatures. Finally, we consider the implications of this for biodiversity conservation policy in forest systems.

## 2 Biodiversity and forest landscape management

Forest landscape management is generally understood to be aimed at achieving a set of environmental conditions, and not a set of outputs such as timber, recreation or wildlife. Nevertheless, by managing the forest in order to yield a set of desired forest conditions it is frequently possible to secure such outputs (Baskent and Yolasigmaz 1999). This broad approach has been adopted by the CBD as the ecosystem approach. One motivation for this is the difficulty in implementing a species approach given the importance of smaller organisms, habitats and processes that are poorly understood (Franklin 1993). A second motivation is the difficulty of meeting conservation goals through a reserve policy – since there can never be enough large and well-distributed protected areas to secure the conservation goal. A third motivation is that the rationale for biodiversity conservation has as much to do with the protection of a range of ecosystem services as it has with the preservation of genetic information.

In other words, the value of the mix of species depends on the value of the goods and services supported by those species. Such indirect use values comprise environmental functions such as nutrient cycling, protection functions such as ground cover for key watersheds, waste assimilation functions such as the retention or detoxification of pollution and wider functions such as microclimatic stabilisation and carbon storage. These functions all indirectly support economic activity and human welfare. The Global Biodiversity Assessment characterised these as regulation, production, carrier and information functions (see Table 1).

Table 1. Forest functions and economic goods and services.

| Regulation Functions Indirect use | Production Functions Consumptive use | Information and carrier functions Non-consumptive use |
| --- | --- | --- |
| – Carbon sequestration<br>– Watershed protection<br>– Erosion prevention and soil protection<br>– Storage and recycling of industrial and human waste<br>– Storage and recycling of organic matter and mineral nutrients<br>– Maintenance of biological and genetic diversity<br>– Biological control<br>– Migratory, nursery and feeding habitat | – Water<br>– Building, construction and manufacturing materials<br>– Energy and fuel<br>– Medicinal resources<br>– Biochemical resources<br>– Genetic resources<br>– Wild living resources | – Aesthetic information<br>– Spiritual and religious information<br>– Cultural and artistic inspiration<br>– Educational and scientific information<br>– Recreation<br>– Habitat |

Source: Adapted from Heywood (1995).

Managing for biodiversity conservation implies an understanding of the effects of forestry practices on the long run dynamics of forests at stand, landscape and even at regional scales. Species-specific approaches, although important, may be too limited (Zavala and Oria 1995, Bengtsoom et al. 2000). Adopting a landscape perspective is considered essential if biodiversity policies are to focus on habitats rather than species (Hunter 1990, Oliver 1992). Habitat in this case refers to the forest structures in which species live. According to these authors, conserving and promoting biodiversity requires a dynamic balance between diverse species composition and different age-classes (i.e. stand-initiation, stem-exclusion, understory-reinitiation, and old-growth) across landscapes. O'Hara (1998) states that biodiversity is maximised with a diversity of structures over a large area including even-aged and multi-aged structures. Hansen et al. (1991) used natural forest in North America as a model to design forest practices consistent with biodiversity conservation objectives. They stated that at the stand level, attention should focus on the role of features of natural areas such as large trees, snags and woody debris, in enriching forest structure; while at the landscape level, practices should involve having a spatial arrangement (size, distribution, edge characteristics, etc.) of forest patches in different successional stages, including old growth stands.

There are certainly some studies that focus on biodiversity conservation in forests as a specific goal. Spellerberg and Sawyer (1996), for example, considered the role of forest management in the restoration of biological diversity in conifer plantations. They argue that forest planning should be located in the landscape of which the conifer plantation is part. Their argument is based on the idea that spatial requirements for maintaining biodiversity differ depending on the scale of the existing management units. Equally, differences may lie in the temporal dimension, i.e. given that the composition of a forest stand changes with time so does the level of biodiversity provided. If the emphasis should be at a landscape rather than a forest stand scale, the concepts of a recent research area, landscape ecology, are important for management of forest resources for multiple values (O'Hara 1998, Bengtsoom et al. 2000). *Landscape ecology seeks to understand the ecological function of large areas and hypothesizes that the spatial arrangement of ecosystems, habitats, or communities has ecological implications'* (Turner 1990, p. 21). In particular, landscape structure influences the movement and persistence of particular species, the susceptibility and spread of disturbances such as fires or pest outbreaks, and the redistribution of matter and nutrients. For instance, the local rate of extinction in landscape patches and the rate of movement of the species among these patches influence species survival. At the same time, the proportion of disturbance-susceptible patches and how they are distributed in the landscape affects the spread of disturbances (Turner 1989).

Managing forests at a landscape level implies focusing on mosaics of patches and long-term changes in these mosaics to integrate ecological values, such as the maintenance of forest ecosystem health and biodiversity conservation, with the production of commodities (Swanson and Franklin 1993, Baskent and Yolasigmaz 1999). Adopting this approach is important because the collective impacts of management practices within individual stands, or even how the different patches are defined, can result in unpredictable effects with important ecological consequences. In addition, management at a single stand level impedes the assessment of the implications of management strategies at a landscape scale. The impacts of forest management practices at different spatial and temporal scales are still not well understood, but are nevertheless essential if we are to be able to predict responses in resource flows, such as timber production and wildlife maintenance (Borgesa and Hoganson 2000, Tang and Gustafon 1997, Baskent and Jordan 1996).

# 3 Spatial interactions in forest landscape management

A number of forestry studies have addressed the tradeoffs between tree diversity and timber revenues under different forest regimes. Buongiorno et al. (1994) and Lin et al. (1996) considered the consequences of different forest practices on tree size (diameter) diversity and on timber economic returns from hardwood stands in Wisconsin. The forestry practices evaluated included differences in the cutting cycles (i.e. intervals between harvests), and in the harvest intensities (e.g. number of trees removed or diameter cutting practices). Later, Lin and Buongiorno (1998) extended this analysis to a landscape approach taking into account stochastic variations in the growth of forest stands and market fluctuations. Biological diversity in their approach was defined as the proportional abundance of stands in different states, depending on the different management strategies. That is, they used a measure of patch diversity.

Other studies have focused on adjacency constraints or modelled species-specific requirements in harvesting schedule problems (see Kurttilla 2001 for a review). Adjacency constraints refer either to an exclusion period between adjacent harvests or to a maximum size of clear-cut which can not be exceeded. For example, Carter (1999) and Tarp and Helles (1997) applied this approach to integrate spatial interactions into the optimisation of forest management. Carter used an integer program to cover a thirty-year planning period to evaluate numerically the impacts on the optimal rotation age arising from spatial constraints. The results are compared with the optimal rotation for a single stand case. Tarp and Helles integrated the trade-offs between economic timber values and adjacency aspects in the selection of an optimal harvest schedule programme. They considered that each stand could be subject to three possible treatments (clear-cutting and re-planting; regeneration felling and regeneration; thinning followed by clear-cutting and regeneration). Following a simulated annealing procedure the value of the objective function with alternative forest configurations, including spatial constraints, was estimated. Hof and Flather (1996) focused on the importance of spatial arrangements of forest patches for the conservation of particular species. They maximised total population, taking into account that the size and location of the habitat patches affect the population through its mean and variance. Fragmentation issues are included based on the probability of patches being connected and this influences the total population mean. Spatial correlation among populations located in different patches of habitat is assumed to affect the total population variance. Simulations for an area with four habitat patches show that the spatial arrangements depend on species dispersal capability and the type of environmental disturbances that affect the correlation between patches.

Most studies of this sort are based on site-specific data and are characterised by large and complex simulation models that aim to capture the complexity of the forest ecosystem and forest treatments. Other studies adopt a more stylised view for approaching forest management with spatial interactions. Based on the traditional Faustmann framework they limit their interest to rotational age decisions. Spatial interactions between forest stands are included, making non-timber benefits dependent on the ages of the stands. The multiple forest stands are assumed to be under sole ownership (i.e. private, public or collective). This implies that the mutual spatial externalities derived of the ecological interactions between the stands are internalised in the management harvesting decisions.

In Bowes and Krutilla's (1985) model a public manager optimises the mixed age-class distribution of the forest stands by choosing the proportion of each age-class that should be harvested in each period. Forest conditions, given by the mix of ages in the stock, influence non-timber values. The standing stock may include one or several age-classes. This analysis shows empirically the potential influence of the mix of ages in the forest stock on the harvesting age decisions and the difficulty on establishing rules of the thumb. Swallow et al. (1997) generalised the Faustmann analysis to the forest level. Location aspects are included in this model assuming that the non-timber benefits of each individual stand depend on a vector of ages of stands nearby or ecologically linked. The holding owner maximises the summation of timber and non-timber benefits of all the stands. Numerical simulations are carried out setting up the multiple-stand model in a linear programming framework, representing a forest ecosystem with two substitute stands providing non-timber services. Swallow et al. (1997) show that the optimal sequence of rotations for any individual stand may differ from the Faustmann-Hartman stationary harvesting schedule. The optimal rotational periods are not necessarily constant because the manager's emphasis on timber and non-timber benefits may change over time depending on the age of the adjacent stand. For different combination of timber and non-timber qualities of the two stands, their numerical results show that optimal harvest patterns in which specialisation of the forest stands across space and time may occur. In particular, the stands alternate between short (forage production) and long (timber production) rotations. Substitution and wealth effects analysed theoretically in a two-stand setting in Swallow and Wear (1993) may be expected. Substitution effects occur because, if a neighbouring stand is clear-cut, this encourages the manager to increase the rotational period of the focal stand. Wealth effects occur because harvest in an adjacent stand causes an increase in the present value of the future environmental benefits of the focal stand and therefore also in the opportunity costs of delaying its harvest. Due to the trade-off between these two effects a clear-cut of an adjacent stand may increase or decrease the rotational age of the focal stand.

An important point here is that in multiple stand forests two adjacent stands can be spatial substitutes, complements or independent (Koskela and Ollikainen, 2001). That is, an increase in the rotation age of an adjacent stand decreases, increases or leaves unchanged the marginal amenity values of the focal stand, respectively. In a single rotation maximisation problem, if the stands are substitutes/complements, the private harvesting age will be shorter/longer, the longer the rotation age in the adjacent exogenous stand. In ongoing rotation maximisation problem the effects of these interactions is more complex. Temporal interdependence between the two stands can be constant, increasing or decreasing depending on how the spatial interdependence of the stands' changes with increases in the rotation age of the endogenous stand.

Tahvonen and Salo (1999) followed the Scandinavian tradition, which focuses on the management problem of non-industrial private forest owners, using a jump-control modelling approach. Their model is defined by three state variables: forest biomass, age of the tree and owner's financial assets. The jump points or discontinuities in the state variables are related to clear-cutting activities in the forest. Clear-cutting activities are modelled through downward jumps in the timber biomass and the age of the trees and upward jumps in the forest owner's financial assets. If spatial interactions influence forest non-timber benefits, they illustrated that a heterogeneous forest structure, in which each stand is harvested approximately halfway through the rotation of its neighbouring stand is optimal. This has the effect of distributing timber and non-timber benefits evenly through time.

Touza-Montero and Perrings (2002, 2003) used a similar approach to analyse rotation periods in a forest landscape. They adapt previous bioeconomic forest models (Termansen 2001) to include information on the age-class structure of forests. Stand interactions are assumed to influence the flow of non-timber benefits provided for the entire forest. The dynamic optimal cutting rule derived suggests that the harvest age of any stand depends on the overall condition of the surrounding forest landscape. In particular, it indicates that a stand's optimal harvesting age is a function of the importance of the stand's non-timber benefits relative to the non-timber benefits derived from the entire forest.

As expected, the optimal harvest age of any stand – for example stand $i$ – is shown to balance the forest marginal timber and non-timber benefits of delaying the stand harvest with the forest marginal costs of waiting (i.e. forest MB = forest MOC). The marginal non-timber benefits of postponing the harvest of stand

*i* is determined by the difference between the forest non-timber benefits before and after the harvest of stand *i*. This difference may be denoted as "stand net contribution" of stand *i* to the non-timber benefits of the whole forest. This "stand net contribution" term indicates that the optimal harvest age of any stand depends not only on the provision of forest non-timber benefits if the harvest of the stand is delayed, but also on how the forest non-timber revenues would be affected if harvest had occurred. Therefore, it represents the difference between forest non-timber benefits with and without the harvest of the stand. If the stands are substitutes in the provision of the non-timber benefits, the "stand net contribution" has a smaller value if other stands are acting as substitute sources of non-timber benefits at the harvest instant than if they are not. For example, if the stands in the collective forest are mature stands, the marginal value of the non-timber benefits from delaying the harvest of stand *i* would be relatively small. While if stand *i* is a mature stand and the remaining stands are young – for instance, they are just being planted – the impact of harvesting stand *i* on the whole-forest non-timber benefit would be higher. Similarly, when stands are complementary, the "stand net contribution" is higher when other stands are acting as complementary sources of non-timber revenues at the moment of harvest.

The marginal costs of postponing the harvest of a single stand in the forest landscape includes the opportunity cost of investment of timber revenues plus a term analogous to the concept of "site value" in the traditional Faustmann framework. This includes the impact of delaying the harvest of stand *i* on the forest timber and non-timber returns of the following rotation. This impact is not only the opportunity cost of a delay in future benefits but also the value of altering the age of stand *i* relative to the age of other stands in the forest landscape. Delaying the harvest decision therefore may cause a shift in the distribution of ages in the forest, altering the interactions between the stands and affecting timber and non-timber benefits in the future.

# 4 Implications for policy

The central stylised fact of this paper is that in forest landscapes, property boundaries do not correspond with ecological boundaries. Therefore the delineation of forest ecosystems at an ecologically significant scale includes many owners and may involved mixed types of ownership (public and private). It follows that the management decisions of any one landowner may have consequences for the decisions of neighbouring landowners. That is, it may imply the existence of spatial externality. If the spatial externalities derived of ecological (economic) interactions are neglected, and if the stands are independently managed, the optimal harvest ages will be determined by the Faustmann-Hartman rule.

However, when spatial interactions between forest stands are endogenous to the harvest decisions, recent studies have shown that optimal harvesting ages no longer follow the Faustmann-Hartman rule (Swallow and Wear 1993, Swallow et al. 1997, Tahvonen and Salo 1999, Touza-Montero and Perrings 2002, 2003). The "optimal" harvesting strategies at a single-stand scale are not necessarily optimal when a larger spatial scale is adopted and when spatial interactions are included in the management decisions. Specifically, when a single stand is managed independently, the Faustmann-Hartman rule indicates that is the flow of the stand's non-timber benefits that influences when the stand should be harvested (Hartman 1976). When spatial interactions are taken into account, it is the relative contribution of each stand to the non-timber benefits provided by the whole forest that determines the rotation intervals (Touza-Montero and Perrings 2003). That is, whether an individual stand is harvested depends on its role in fulfilling in forest landscape objectives. This conclusion may imply that non-harvesting policies may be optimal in stands that contribute highly valuable forest landscape benefits. In addition, if non-timber benefits increase with the age of the stands and are significant with respect to other uses, it may be optimal never to harvest any of the stands in the forest. Most importantly, it implies that the ecological and economic consequences of alternative actions taken at small scales (i.e. stands) on a wider spatial context (i.e. forest landscape) must be understood for optimal forest management.

What implications do these findings have for policy? At one level the implications are quite generic. Spatial externalities and spatially defined public goods are still externalities and public goods. In the case of

externalities, the first-best solution is the internalisation of externalities through either the appropriate assignment of property rights, or the use of mechanisms that confront users with the social opportunity cost of their actions. Both result in an efficient allocation of resources. World markets for fibres drive both the choice of cultivated species in forestry and the privately optimal rotation period, but do not signal the costs to society of the resulting forest structure. In principle there exists a set of prices and hence a set of market based instruments that would induce a socially optimal forest structure.

Property rights solutions to the problem of spatial externality include both the merging of rights to interacting stands and the assignment of rights to the external effects between stands. The first converts an economic problem to a management problem by directly internalising the externality. The second creates a market in the externality by allowing the source and victim of an effect to trade. While we are unable to cite examples of the development of markets in spatial externalities within forests, there is clearly considerable scope for this. Markets in such effects would involve small numbers of parties and so few transactions costs.

Other market based instruments that are currently used to address spatial externalities in forests include taxes (royalties/stumpage fees), charges, pest control subsidies, afforestation subsidies and grants. It is not hard to find examples of public payments to private landowners for biodiversity conservation through forest afforestation. In the UK the woodland grant scheme is a case in point. In this scheme, payments for reforestation depend on the potential for biodiversity conservation, and are graduated depending on the location of existing forest. Grants are highest where the proposed stand is contiguous to a large existing block precisely because of the greater biodiversity benefits this offers (Forestry Commission 1997).

The second-best solution applies where the first-best solutions are infeasible because there is insufficient information, transaction costs are too high, wealth effects are too severe, administrative capacity is too low, or because there are threshold effects involving irreversible change. In this case the solution involves direct regulation supported by enforcement and penalties for non-compliance, and the economic test is one of cost-effectiveness. Direct regulation is almost always the instrument of choice where there are large numbers of forest owners (since that implies the transactions costs are high).

Forest regulations are currently widespread. Regulations at international, national, regional or local level currently deal with a range of management issues including timber harvesting, reforestation, fire hazard, deforestation in catchments, habitat and species conservation While these can be useful tools for achieving social objectives in forest landscapes that are owned by many individuals they may not induce an efficient use of forest resources for reasons that are well understood. They do not provide users with information on resource scarcity until the penalty for non-compliance is activated. This is frequently unconnected to the opportunity cost of non-compliance. They tend to treat all users identically. Enforcement is frequently weak or non-existent and is data demanding. Spatial inventories of private forest ownerships are required. In areas such as in Galicia where there are 673 000 single forest owners with holdings of about two hectares (GEPC, 2002) these inventories can be administratively demanding.

The use of mixed instruments in forestry tends to be restricted to the allocation of harvest rights, in which a total allowable harvest is allocated between foresters through a market for harvest rights. This applies to the exploitation of natural forests in the public domain, but has little interest in areas where forests are privately owned.

If the ecological services provided by forests with multiple owners include local public goods such as watershed protection, soil conservation, habitat provision and the like, the policy problem is to persuade landowners to cooperate in the provision of such services. In certain of European forest systems this is achieved by assigning management responsibilities to a forest management company. As a result the forest is run as a single entity – which effectively internalises the externalities between stands.

Where incentives have been offered to private landowners to cooperate in the management of forest resources the results have been mixed. Klosowski et al. (2001) carried out a conjoint analysis to study the probability that coordinated management programmes would be undertaken. Results from a survey, carried out to landowners in New England, showed that this probability is very small even when substantial incentives (e.g. property tax reductions) are offered. Even though, larger tax incentives and short commitments increase

this likelihood. Landowners already involved in forestry-related organisation are more likely to enroll in these programmes.

Whether private landowners will cooperate in the absence of incentives depends on the likelihood that an agreement between them would otherwise be self-enforcing. This depends on a number of factors of which the most important are the number of parties and the incentive to defect from the agreement (which is related to the difference between the payoffs under the Nash and cooperative equilibria). Collective property regimes have been suggested as a way of promoting forest management for multiple private and public goods and services in Europe (Gluck, 2000). This requires community members to integrate both timber and environmental interests by negotiation and cooperation. It is argued that it guarantees rights without parcelling the forest, and unparcelled areas are better for biodiversity conservation. It will also internalise externalities by ensuring that decisions are made jointly.

If we now try to pull together the policy implications of spatial interdependence, the central point is that decisions should be made (or at least coordinated) at the geographical scale at which the interdependent activities operate. Since the boundaries of individual land titles almost never coincide with the boundaries of the impacts of land use, it follows that decisions should be made at (or should take into account the effects on) the geographical area in which activities are interdependent. But this is precisely the meaning of the European principle of subsidiarity. The principle is typically used to distinguish between decisions that are most appropriately taken at European and national levels. But it applies across all scales. Within individual countries the question is whether decisions should be taken at the national, regional, local, firm or household level. The existence of spatial externalities and spatially defined public goods implies that the appropriate geographical scale of decision-making is wider than the firm or household level. But how much wider? This depends of the nature of the ecological services supported by the forest system in question. Carbon sequestration implies that management should be coordinated at the global level. Watershed protection implies that it should be coordinated at the scale of the catchment. Pollination services imply that it should be coordinated over the habitat of the relevant pollinators.

In many European forests spatial interactions tend to be limited in their geographical extent. In some cases they may be limited to neighbouring stands. However, if each stand interacts with all neighbouring stands then the appropriate geographical scale at which the forest should be managed is still the whole system. Just as an individual with a contagious disease is a threat not only to the people with whom come into contact directly, but to all people with whom they come into contact indirectly – often the whole population. The relevant policy questions then concern the best mechanism for ensuring the coordination of decisions at the relevant geographical scale. As always these tend to be case specific, and sensitive to the existing structure of property rights and regulatory regime, as well as the characteristics of existing markets. Given the kinds of spatial interactions observed in temperate forest systems, and given the nature of activities undertaken in those systems, conservation policies in temperate forests generally attempt to maintain a patch structure or mosaic of land uses. The appropriate geographical scale is that over which the components in the mosaic support complementary ecosystem services.

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*Footnotes*

[1] Over 40% of GEF funding on biodiversity was directed to humid tropical forests in its first two phases (Pearce et al. 1999).