Assessing the Effectiveness of Tradable Landuse Rights for Biodiversity Conservation: an Application to Canada's Boreal Mixedwood Forest

Marian Weber,
Sustainable Ecosystems, Alberta Research Council, 250 Karl Clark Road
Edmonton, Alberta, CANADA T6N 1E4
E-Mail: weber@arc.ab.ca

Fourth BIOECON Workshop on
The Economics and Biodiversity Conservation
Venice, 28th-29th August, 2003.

Abstract

Ecological reserve networks are an important strategy for conserving biodiversity. One approach to selecting reserves is to use optimization algorithms that maximize an ecological objective function subject to a total reserve area constraint. Under this approach, economic factors such as potential land values and tenure arrangements are often ignored. Tradable landuse rights are proposed as an alternative economic mechanism for selecting reserves. Under this approach economic considerations determine the spatial distribution of development and reserves are allocated to sites with the lowest development value, minimizing the cost of the reserve network. The configuration of the reserve network as well as the biodiversity outcome is determined as a residual. However cost savings can be used to increase the total amount of area in reserve and improve biodiversity outcomes. The appropriateness of this approach for regional planning is discussed in light of key uncertainties associated with biodiversity protection. A comparison of biodiversity outcomes and costs under ecological versus economic approaches is undertaken for the Boreal Forest Natural Region of Alberta, Canada. We find a significant increase in total area protected and an increase in species representation under the TLR approach.

Keywords: biodiversity conservation, reserve design, tradable landuse rights.

JEL Classification Codes: Q28, Q21, Q44, H41

1. Introduction

The Rio Convention on Biodiversity (1992) commits governments to recognize and protect biodiversity values in land management decisions. Ecological reserves are protected areas where development and other anthropocentric activities are minimized or prohibited. They provide sanctuary for species at risk and are the cornerstone of any biodiversity protection strategy. Article 8 of the Convention calls for signatories to establish, regulate and manage networks of protected areas (reserve networks) to promote the protection of ecosystems, natural habitats and the maintenance of viable populations of species in natural surroundings (IUCN 1992). The goal of a reserve network is to maximize species representation and probability of persistence (Margules and Pressey 2000). Implied in this approach is the idea of efficiency, or that reserves should be selected to minimize the amount of land required to achieve these objectives. Improvements in computational power in the last decade have been accompanied by a corresponding emphasis on optimization approaches for designing reserve networks. Optimization approaches include the maximal coverage (MC) approach, which maximizes a biodiversity metric subject to an aggregate reserve area constraint (*c.f.* Underhill 1994; Church, Stoms, and Davis 1996; Camm, Polasky, and Csuti 1996; Csuti et al. 1997). Maximal coverage is useful for countries or regions attempting to fulfill percentage based protected area targets such as the 12% guideline interpreted from the 1987 Brundtland Commission Report (Sanjayan and Soule 1997).

Unfortunately there are numerous practical constraints to implementing "optimal" reserve networks. These include existing property right structures, social attitudes, costs, and political opposition (Pressey, Possingham, and Margules 1996). The result is that planned networks are often not feasible. In reality, reserves tend to be allocated to low value sites in an ad hoc fashion and the goals of the reserve network are compromised (e.g. Pressey 1994; Simon et al. 1995; Thomas et al. 1997). This problem is particularly acute on public lands where resource rights are not comprehensive and multiple use problems abound. However when economic considerations are considered explicitly in conservation planning reductions in costs can lead to an increased level of protection. Recent studies demonstrate that the costs of achieving biodiversity targets are reduced if one accounts for land values in designating ecological reserves. Under the "budget constrained" (BC) approach, reserves are selected by minimizing the land costs associated with achieving a particular biodiversity metric (e.g. Faith, Walker, et al. 1996; Ando, Camm, et al. 1998; Polasky, Camm et al. 2001). However, implementation of the budget constrained approach requires appropriate price signals to assign values to land use alternatives. These price signals are biased or non-existent on publicly owned lands where much conservation effort is applied.

Finding appropriate mechanisms for assessing values and compensating costs is a persistent problem in biodiversity conservation (Panayotou 1997). As a result there is interest in the use of market mechanisms such as tradable permits that have the potential to reveal privately held values associated with public goods (e.g. Shogren et al. 1999; Tietenberg 2000). Under a tradable permit system the regulatory authority sets a cap on environmental resource use and allows the market to ration access to the resource so that the resulting pattern of use minimizes the cost of achieving the environmental objective (Montgomery 1972). Weber and Adamowicz (2002) propose tradable landuse rights (TLRs) as an economic mechanism for managing cumulative environmental effects on public forest lands. Under this system the regulator sets a cap on the amount of land that can be developed in a region and allocates the area available for development through TLRs. Directing development towards sites with the highest economic potential minimizes the opportunity cost associated with the land use constraint. The potential cost savings associated with this approach can then be used to increase the amount of land set aside in habitat protection. The landscape ecology literature emphasizes the importance of the spatial configuration of habitat in determining the quality of reserves. However there is a tradeoff between the total amount and spatial configuration of habitat for biodiversity protection and the magnitude of the tradeoff is subject to debate with at least some ecologists suggesting that configuration matters only when there is very little habitat left (e.g. Fahrig 2001; Trzcinski, Fahrig, and Merriam 1999). By exploiting the tradeoff between total amount and configuration of area in reserve, the TLR approach may result in improved biodiversity outcomes.

In this paper we apply the concept of TLRs to the problem of reserve design for biodiversity protection in a managed forest. We develop a case study for the boreal mixedwood ecological region of Alberta's Boreal Forest Natural Region (BFNR) and numerically compare biodiversity and economic outcomes under TLRs relative to the maximal coverage and budget constrained approaches. Over the last twenty years the BFNR has come under unprecedented pressure from industrial development. While ecological reserves are viewed as a key component for achieving ecological sustainability, the BFNR remains unprotected in a meaningful way (WRI, 2001). In the following section we present a formal model for selecting reserves using TLRs and discuss the robustness of this framework when the underlying assumptions are relaxed. The benefits of the TLR approach are discussed relative to MC and BC approaches, with an emphasis on the key uncertainties and risks associated with each. In section 3 we develop the case study and compare biodiversity and cost outcomes of each approach through numerical simulation. We find large cost savings under TLRs that can be used to increase the total amount of area in reserve and increase species representation relative to the MC approach. This is an important finding since cost savings and flexibility in choosing the pattern of development may increase the political feasibility of completing a protected area network in this region.

2. TLRs and the Reserve Selection Problem.

Assume that a fixed percentage of land in a region is to be set in reserve for habitat protection. This is a coarse filter approach to biodiversity protection and a realistic representation of actual strategies for regional biodiversity conservation (e.g. Faith 1997; Sanjayan and Soule 1997). Biodiversity objectives are addressed in the selection of the amount and type of land to be set aside. Although the model implicitly assumes one habitat type the regulator can create separate markets for different habitat types (Weber and Adamowicz 2002).

Assume that the region is divided into *n* regular parcels of land (for example townships) that can be bought or sold. Let A be equal to the total area of the region. Each land parcel h_i is spatially referenced by *i* so that $\sum_i h_i = A$. Economic attributes at site *i* are described by the vector $X_i = [x_i1, x_i2, \ldots x_{iK}]$. In general economic benefits are unevenly distributed over space due to variations in transportation costs, the heterogeneous distribution of extractable resources, and landscape characteristics that affect the cost of access to resources. Firms are denoted by $j=1, \ldots j$. Each firm obtains a benefit of :

$B_j(h_{ij}) = B_j(X_i, \sum_j h_{ij})$, if right to develop at site h_i is allocated to *j*;

$B_j(h_{ij}) = 0$ otherwise.

The benefits from developing site *i* are a function of the attributes at site *i* as well as the total number of rights already held at alternative sites.

The reserve design problem is described as follows. A fixed percentage of the landscape, α, is slated for development. Therefore the development constraint is $\bar{A} = \alpha A$. The remaining percentage, $(1 - \alpha)$, is the basis for a reserve network. The objective is to maximize the benefits from land use subject to the above land use constraint:

(P1) $$\underset{h_{ij}}{\text{Max}}\ W = \sum_j \sum_i B_j(h_{ij})$$

subject to

(P1.1) $$\sum_{i=1}^{n} h_{ij} \leq \bar{A}; \qquad (\lambda \geq 0);$$

(P1.2) $$\sum_{j}^{j} h_{ij} \leq 1; \qquad (\mu_i \geq 0);$$

(P1.3) $$h_{ij} \in \{0, 1\}.$$

(P1) is equivalent to minimizing the opportunity costs of habitat protection. The first constraint limits the total amount of land available for development, while the second constraint allocates each site to only one firm.[1] The last constraint requires that a site be either fully developed or set aside as reserve. The solution to this problem is given by

[1] The assumption that only one activity takes place at each site is motivated by the static framework and the fact that some activities, such as agriculture and forestry, at a particular site may be mutually exclusive. Nonetheless, we can extend this framework to the case where multiple firms and sectors may operate at each site without affecting the main results.

(2) $$\left[\frac{\partial B_j}{\partial X_i} + \frac{\partial B_j}{\partial H_j} \right] - \mu_i - \lambda \leq 0 \,; \qquad \forall \; i, \; j \,,$$

where $H_j = \sum_i h_{ij}$, and $\frac{\partial X_i}{\partial h_{ij}} = 1$.

Equation (2) sets the marginal benefit of site *i* to the user *j* that obtains development rights equal to the shadow price $\mu_i + \lambda$ where λ represents the cost of the aggregate reserve area constraint and μ_i represents the marginal value of the set of attributes found at site *i*. The marginal benefit in (2) includes benefits arising from quality attributes at site *i* (the first term on the LHS) as well as the marginal benefit of bringing an additional unit of land into production (the second term on the LHS). If the marginal benefit of developing site *i* is less than λ, then site *i* is set aside as reserve.

We show that a market for development rights can lead to the cost minimizing distribution of activities described by (2). Assume that firms bid for each parcel of land in the region, but only those $\overline{A}$ units associated with the highest bids are allocated for development. If the bidding process is competitive, the decision rule facing each firm in choosing a bid for the right to develop site *i* is given by

(3) $$\left[\frac{\partial B_j}{\partial X_i} + \frac{\partial B_j}{\partial H_j} \right] - p_i \leq 0$$

where p_i is the price paid by firm *j* for the right to develop site *i*. Equation (3) is equivalent to the decision rule given by equation (2) as long as $\mu_i + \lambda = p_i$. Proof of this result is provided in the appendix. As long as (3) holds, TLRs minimize the opportunity cost associated with a reserve network, and cost savings can be used to increase the amount of land set aside in reserve and potentially improve biodiversity outcomes relative alternative reserve selection approaches.

The cost minimization result obtained above is derived from a highly stylized model. In particular, the cost effectiveness of TLRs depends on the assumption that there are no spatial or network cost spillovers for firms. This allows each parcel of land to be treated as a homogeneous commodity whose economic value does not depend on the context of other sites in the network. In practice, transportation networks and fixed investment costs will lead to a violation of this assumption. However, the fact that auctions are common in natural resource markets where transportation costs are significant suggests that the magnitude of the inefficiency caused by network externalities might be small compared to the benefits of using auctions to allocate resources. Market power is another potential problem for TLRs in regions where a few large players dominate land use.[2] Issues of market power and spatial externalities can be addressed to some extent in the design of the permit market itself but are beyond the scope of this paper. It is important to note, however, that the costs associated with a reserve network are most likely to be greater than the minimum, but lower than under alternative approaches.

The information environment in which conservation decisions are made is incomplete. Because of key uncertainties, MC and BC approaches may be inherently more risky than the TLR approach. Consider the issue of biodiversity measurement. Biodiversity criteria used in most MC and BC studies are based on species metrics and require data on the presence or absence of species as well as their geographical distribution. Species based diversity metrics

[2] For example, market power is a potential problem in the BFNR where a few large forestry and oil and gas firms dominate land use (Weber and Adamowicz 2002).

vary according to the weight given to individual species. Weighting functions derived from "uniqueness" characteristics such as phylogenic distinctiveness (e.g. Vane-Wright, Humphries, and Williams 1991; Weitzman 1992; Faith 2002) have been supported by underlying utility theory (Metrick and Weitzman 1998; Nehring and Puppe 2002). Unfortunately, many diversity metrics are not operational due to the high cost or impossibility of obtaining relevant information (Weikard 2002), and there is little empirical evidence backing the selection of one diversity measure over another. Since species classification is incomplete, surrogates such as umbrella species with large spatial requirements are often used to capture the presence of other species (Simberloff 1998). However tests on the use of species based indicators as surrogates for biodiversity have not been promising. Reserves selected for conservation of different taxa or based on higher taxon surrogates often do not overlap, undermining their usefulness for biodiversity planning (e.g. Jaarsveld et al. 1998; Faith and Walker 1996). Process based indicators such as habitat and ecosystem type have been advocated because they capture the conditions necessary for maintaining ecosystem function and data are readily available (Margules and Pressey 2000).

A significant problem in reserve design is that it is difficult to incorporate factors important to species persistence such as the size, shape, or quality of habitat in selected sites. Part of the problem is that theory suggests conflicting requirements. Biogeography theory suggests large habitat patches support large populations of species for longer periods of time than small patches, and increased patch connectivity facilitates dispersal and improves persistence (van Langevelde et al. 2000).[3] However, this implies a tradeoff between a few large

[3] Patch connectivity is defined by the distance between patches.

reserves that favour the persistence of some species versus more smaller reserves that together are more representative of biodiversity but are individually less effective for persistence (Margules and Pressey 2000). Furthermore, increased connectivity may increase species risk by making populations more vulnerable to spatially correlated environmental stresses such as fire, extreme weather, and disease (Hof and Flather 1996).

Overall ecological theory suggests tradeoffs between reserve size, patch size, and connectivity. Unfortunately the tradeoffs are difficult to empirically assess.[4] In addition responses to fragmentation tend to be species specific and sensitive to the scale at which habitat is defined (Cumming and Schmiegelow 2001). Although it is generally accepted that habitat quantity and configuration are substitutes for species persistence, the degree of substitutability between these two attributes is unclear. Nonetheless, there is at least some evidence that the amount of habitat is of greater importance that spatial configuration in forested landscapes, with spatial configuration becoming important only when there is very little habitat left (Fahrig 2001; Schmiegelow and Makkonnen 2002; Trzcinski, Fahrig, and Merriam 1999).

The above discussion suggests that implementation of MC and BC approaches using species based biodiversity indicators may pose more risk than focusing on habitat or ecosystem protection. Moreover there is evidence that the cost effective TLR approach can improve biodiversity protection by increasing the total area in reserve. This suggests that the economically driven TLR approach may actually be less risky than alternative approaches based on species indicators. Practically the efficacy of the TLR approach depends on the magnitude of

[4]For example, the debate between one single large reserve versus several small reserves has no single answer and depends on the taxa, geographic locality and position of the reserves (Prendergast, Quinn, and Lawton 1999).

cost savings that can be reinvested in habitat protection. Expected cost savings are greatest when there is large spatial variation in resource values. If site values are uniform the mechanism for selecting the network is irrelevant in terms of costs. Cost savings will decrease as the size of the reserve network increases. This is because the opportunity to substitute low value for high value reserves is greatest when there are fewer reserves. In other words, the costs of achieving spatial objectives in terms of foregone habitat area are greatest precisely when it is most difficult to achieve spatial objectives - i.e. when the number of sites available for constructing contiguous habitat patches is low. Finally, transportation and networking costs play an important role in the location decisions of firms and the spatial arrangement of economic activity. This implies a tendency for industrial activities to be clumped around sites with high development value. The corollary is that protected areas are likely to be spatially clumped under the TLR approach.

The appropriate role for economic criteria in reserve design is controversial. In practice the selection of protected areas because of their relative lack of development value has led to biased content in regional reserve systems, leaving species, communities, and ecosystems in the greatest need of reservation without protection (Pressey 1994). While the habitat value of potential reserves is heterogeneous, markets treat ecological attributes as homogeneous. Confining the spatial extent of the TLR market to a regional scale where habitat type is relatively homogeneous is necessary to minimize the potential for systematic bias in the selection of landscape attributes in the reserve network. The quality of the reserve network may still vary due to the spatial distribution of protected habitat generated by the reserve network. However, as we show below, exploiting the tradeoff between total area protected and configuration can improve biodiversity outcomes for a given conservation budget.

3. Data Description and Land Value Calculations

The above discussion highlights some of the problems in choosing appropriate objectives for reserve selection algorithms and lends credibility to the economic approach to reserve selection. In this section we quantify the economic and ecological tradeoffs associated with alternative approaches to reserve design for protecting avian species in Alberta's Boreal Forest Natural Region (BFNR). The BFNR covers 52% of the provincial land base and constitutes most of the provinces forested land, the majority of which is publicly owned and allocated through a variety of lease agreements based on approved plans. Land values, where they exist, do not represent the true opportunity costs of setting a site aside for reserve since they are distorted by regulatory terms and conditions of leases that attenuate leaseholder rights. The study area chosen for analysis consists of approximately 114,000 square km, or 1137 townships, in the boreal mixedwood ecological region of Northeastern Alberta.[5] The boreal mixedwood forest is a mosaic of stands of different ages and species composition. Each township represents a unit for the reserve selection problem.

Alberta Breeding Bird Atlas data recording the presence or absence of a suite of bird species were collected between 1987 and 1991 by the Alberta Federation of Naturalists and used to construct a biodiversity index. Multiple logistic regression models were use to develop prediction probabilities for the distribution of 27 bird species in the study area. Detection probabilities are based on habitat abundance and configuration metrics derived from 1993 snapshots of Alberta phase 3 forest inventory data (Alberta Forest Service 1985). A complete

[5]A township is equivalent to an area of 10 square km or 10,000 ha.

description of the methods used to generate the data can be found in Vernier, Schmiegelow and Cumming (2002). At the time the data were collected over half of the total extent of the boreal mixedwood region in Canada was still forested and the main Forest Management Agreement covering the study area had only just been approved. Therefore the prediction probabilities are assumed to represent the natural range of species detection rates across the study area. The "productivity" of a particular site is defined by the average of predicted detection probabilities across species given by:

(4) $$d_i = \sum_{l=1}^{m} \rho_{li} / m .$$

The index specified in (4) weights species equally so that a comparison across sites reveals information about the total population of birds, but not particular species.[6]

The main contributors to forest disturbance in the study area are forestry, and oil and gas sector activities. Resource rents are equal to the discounted present value of the stream of future net revenues expected from resource exploitation. In a competitive market these rents would be capitalized in land prices that could then be use to calculate the opportunity cost of the reserve network. However, actual land prices are not available on public lands, and much of the deciduous timber resource in the study area is tied up in a forest management agreement and is not allocated through a market. In the remainder of this section we construct a proxy for land values using resource inventories and prices paid at auction for rights to exploit resources. Due to data limitations, estimates of resource values are derived from different time snapshots for

[6] See Hof and Bevers (1998) for a discussion of probabilistic representation of species richness criteria in objective functions.

forestry (1993) and oil and gas (2001). For consistency, all prices used in the rent calculations are in 1992 constant dollars.

Forestry rents are defined as the discounted present value of standing timber. A proxy for standing timber values was obtained from the Revised Alberta Timber Damage Assessment (TDA) updated for 2000-01 (Alberta Environmental Protection 1995). The TDA is used to compensate forest tenure holders for losses in trees arising from oil and gas sector activities. TDA values are derived from bonus prices paid in Alberta for commercial timber permits for coniferous and deciduous species. Bonus bids incorporate all private costs, including imputed transportation costs, and represent the expected discounted present value of the timber resource to private firms. In addition, the TDA includes an adjustment for annual allowable cut effects over the whole forest.[7] The TDA was applied to a 1993 snapshot of Alberta's phase 3 inventory. TDA values do not include dues paid to the crown.[8] Dues in Alberta vary by volume, wood type, product, and price. Average timber charges for 1999 by species type and volume were applied to the Phase III Inventory in order to calculate maximum potential foregone crown dues. These were added to TDA values to derive full forestry sector rents.

Similarly, energy sector rents are equal to the discounted present value of remaining marketable oil and gas reserves, and consist of both private returns as well as royalty payments received by the government from energy production. Data on remaining marketable reserves and bonus bids for petroleum and natural gas (PNG) leases were obtained by the Alberta Energy

---

[7] The annual allowable cut effect arises in regulated forests that prescribe constant harvest rates throughout a forest rotation. The allowable cut effect spreads any loss of inventory over the whole rotation by reducing the amount that can be harvested in subsequent years (Pearse 1990).

Utilities Board and used to estimate energy rents. Private returns are inferred from the price firms are willing to pay for PNG leases and are assumed to be equal to the total value of remaining marketable reserves net of royalties and costs. The leases grant surface rights for both the exploration and production of subsurface energy resources and cover all hydrocarbon resources except tar sands, and natural gas associated with coal seams. PNG bonus sales take place every two weeks and data were collected for the 5 year period April 1, 1996 to March 31, 2001.[9]

Royalties generated by the energy sector constitute a large percentage of Alberta's provincial budget. In general, approximately 30% of the market value of a typical oil or gas reserve will be paid out to the province in royalties. Foregone potential royalties were calculated by applying formulae published by the Alberta Government to the oil and gas reserve data base (Alberta Resource Development 1999). The existence of remaining marketable reserves does not guarantee that they will be exploited. Therefore it is necessary to weight potential royalty values by the probability that they will be realized. We assume that a firm purchases a lease if the price of the lease exceeds the discounted present value of the future net revenue stream. If there is no lease on a particular parcel of land, this implies that exploitation of underlying remaining reserves is not economically viable in the current period, ie., costs exceed revenues. Therefore the price paid for a PNG license incorporates the explicit cost of exploration and production, as

[8] Alberta crown dues for forest resource improvement were not considered resource rents.

[9] There is a four year grace period in Northern Alberta between the date that licenses are issued and the date that drilling must commence. Therefore bid data for the 5 year period leading up to 2001 was felt to be a fair representation of the *current* expected discounted present value of remaining energy reserves. This period coincidentally encompasses both a trough (1998) and peak (2001) in energy prices.

well as the implicit cost associated with discounting the future revenue stream. Note that the greater the exploration risk, the more uncertain the future revenue stream implying lower prices for PNG licenses. Thus even if remaining marketable reserves have positive value in the current period, the discounted value may be zero if revenues are realized several years in the future or are highly uncertain.

Because bid prices for PNG licenses incorporate exploration costs, risk, and discounting of future revenues, we can use these prices to discount the value of royalties from remaining marketable reserves. For example, if the value of PNG leases associated with remaining marketable reserves is zero, then the current discounted present value any associated royalties is also zero.[10] The expected value of a PNG lease at site i is given by:

(5) $$X_i = \beta_1 \; x_{i1} - \beta_2 \; x_{i2} + \mu$$

where $x_i 1$ represents the total value of remaining reserves less royalties and $x_i 2$ represents the expected total costs associated with exploitation. The value of X is equal to the price paid for the PNG lease. The total current value of remaining reserves, $x_{1,}$ is calculated using current oil and gas par prices. Although we do not have direct information on expected costs, x_2 , low bid prices relative to the value of remaining reserves is indicative of high costs.

The regression of X on x_1 generates a forecast error

(6) $$\hat{X} \; - \; X = \varepsilon = \beta_2 \, x_2 - \beta_2 \, P_{12} \, x_1$$

---

[10] Although the private sector discount rate is not necessarily the social discount rate, it is the defacto discount rate for this problem since the reserves must be produced privately in order to generate any public revenue.

where $P_{12} = Cov(x_1 x_2)/Var(x_1)$.[11]

Assuming constant or increasing returns to scale, the covariance between costs and reserve size is non-negative. Therefore the error term varies systematically and is a positive function of the cost associated with exploiting the reserve. In particular, The shape of the distribution of (ε / x_1) is the same as that of $\beta_2 (x_2 / x_1) - \beta_2 P_{12}$, and the percentile rank of (ε / x_1) will be the same as the percentile rank of (x_2 / x_1), or costs relative to the value of remaining marketable reserves. We use the distribution of (ε / x_1) across townships as a weighting factor to discount potential royalties based on the distribution of cost shares over the study area. The weighting index is equal to the percentile rank of each (ε / x_1) within the distribution. Note that as relative costs decrease, the discount factor applied to royalties also decreases zero.

The total value of the energy sector was obtained by summing lease values and weighted royalty values by township. These were added to TDA values to obtain resource rents by township across the study area. These values are displayed in Table 1.

[11] The sign of $P_{12} = Cov(x_1 x_2)/Var(x_1)$ is a function of returns to scale since the size of remaining marketable reserves is the only source of variation in x_1.

Table 1. Land Rents per Township by Sector ($1992)[12]				
	Forestry	Energy Royalties	Energy Leases	All Sectors
Average	$9,891,194	$13,107,440	$219,607	$23,218,242
Minimum	$1,435,152	0	0	$1,435,152
Maximum	$33,438,156	$1,020,000,000	$5,983,663	$1,027,664,204

The total value of land rents for the study area is $26.4 billion. Table 1 shows that the average land value per township is equal to $23,218,242 ($2321 per ha.) and ranges from a minimum of $1,435,152 ($144 per ha.) to a maximum of $1,027,664,204 ($102,766 per ha.). It is interesting to view the relative rents from the forestry and energy sectors. The perception in Alberta is that the contribution of the forestry sector to the provincial economy is dwarfed by the contribution of the energy sector. However, the average forestry rent for the study area is $9,891,194 per township while the average energy sector rent is $13,327,048 per township. In spite of the large contribution of the energy sector to provincial revenues *ex post*, the average capitalized value of energy resources in land values *ex ante* is similar to that in the forest sector. This reflects the riskiness of energy exploration - the returns from PNG lease sales represent a negligible portion of energy rents (~1%) collected by the province. In addition, the low average value for energy sector rents arises from the spatial distribution of energy resources in the region as energy leases currently exist on only 55% of townships in the study area. There are several

[12] Per hectare land values can be obtained by dividing by 10,000.

omissions in the land value estimates worth mentioning. First in spite of large bitumen deposits in the study area, we made no attempt to value the rents associated with tar sand development due to their relative complexity and lack of data. In addition, we don't include option values for future resources that may be developed such as coal-bed methane, peat, and gas shale.

4. Simulation

We compare the TLR approach to reserve selection with the maximal coverage and budget constrained approaches. Under the maximal coverage approach reserve sites are selected to maximize biodiversity subject to a constraint on the number of sites available for habitat protection. The maximal coverage (MC) problem is given by (P2):

(P2) $$\underset{y_i}{\text{Max}} \sum_{i=1}^{n} y_i \; Z_i(M)$$

subject to

(P2.1) $$\sum_{i=1}^{n} y_i \leq \bar{k} = A - \bar{A}, \text{ and}$$

(P2.2) $$y_i \in \{0, 1\}.$$

The problem in (P2) is an integer program where y_i=1 if site i is selected for reserve, and zero otherwise. The total amount of area in reserve, $\bar{k}$, is constrained in (P2.1) to be no greater than the total amount of area in the region, A, less the amount available for development, $\bar{A}$. The biodiversity index for each site, $Z_i(M)$, is defined on the set of species M to be represented in the network. In this simulation the objective is to maximize the expected probability of species detection over the entire reserve network. Therefore $Z_i(M) = \sum_{1}^{m} \rho_{im}$.[13] To ensure that all species are represented in the reserve network we add the additional constraint:

(P2.3) $$\sum_{i=1}^{n} y_i \, s_{ij} \geq 1$$

where $s_{ij} = 1$ if species *j* is present in township *j*, and zero otherwise. We make the arbitrary assumption that $s_{ij} = 1$ if probability of detection is above the 50th percentile in the distribution of detection probabilities for that species over the study area. The solution to MC is given by $Z_i^*(M \mid \bar{k})$.

Since (P2) does not consider the relative opportunity cost associated with the various townships, the coverage problem is reformulated as a cost minimization problem to determine whether an alternative set of reserves could achieve $Z^*(M)$, the biodiversity outcome under (P2), at a lower cost. This is the basis of the budget constrained (BC) approach given by:

(P3) $$\underset{y_i}{\text{Min}} \; C(y) = \sum_{i=1}^{n} v_i \, A_i - \sum_{i=1}^{n} (1 - y_i) \, V_i \, A_i$$

subject to

(P3.1) $$\sum_{i=1}^{n} y_i \; Z_i(M) \geq Z^*(M)$$

where *C(y)* is the opportunity cost associated with the reserve network *y*, and V_i is the land

[13] Note that since *A* is a constant, maximizing expected detection over the network is equivalent to maximizing (P2). In addition, the formulation of (P2) suggests that species are "perfect substitutes" in the objective function. This is true for all metrics based on unweighted species richness criteria.

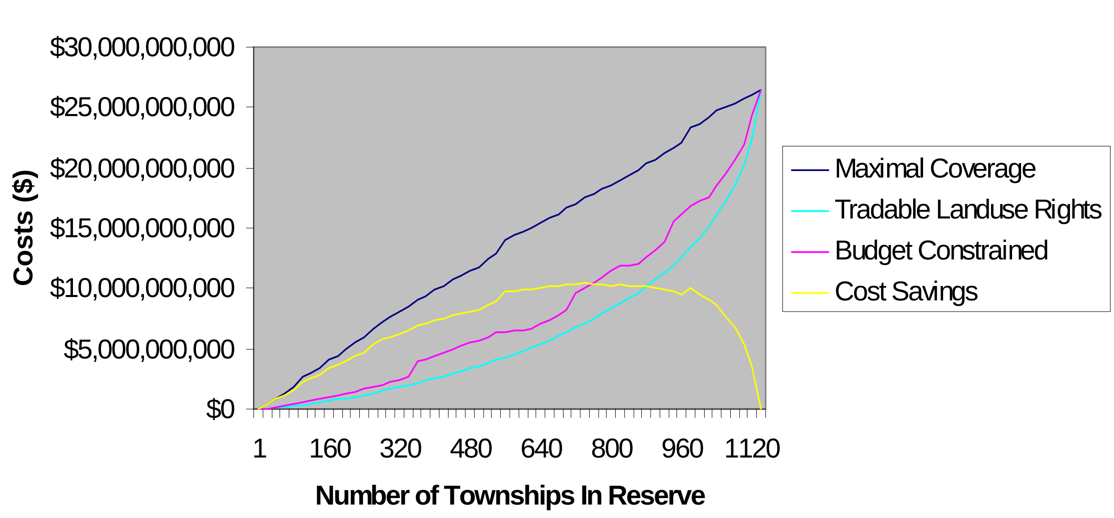
value associated with township *i*. The reserve network solution to (P3) is given by $\tilde{y}$. By definition, $C(\tilde{y}) - C(y^*) \leq 0$, in other words the BC approach requires more area of lower value to attain the same level of biodiversity generated by the MC approach. Finally, in order to numerically compare the MC and BC approaches to the TLR approach, (P1) is reformulated as

$$\text{(P1')} \qquad \underset{y_i}{\text{Min}}\, C(y) = \sum_{i=1}^{n} V_i A_i - \sum_{i=1}^{n} (1 - y_i) V_i A_i$$

subject to (P2.1) and (P2.2). The TLR alogrithm results from the fact that TLRs find the least cost set of reserves for a particular reserve area constraint.

Figure 1 illustrates the costs of biodiversity protection under the three alternative reserve selection approaches represented by (P1)-(P3). The cost of reserves under MC increases linearly, while the cost under TLRs is convex indicating an initial abundance of low value land that can be cheaply set aside in habitat protection. Cost savings are quadratic in the size of the land constraint. As the size of the reserve network increases, there are fewer cheap substitute sites available, and costs under all approaches converge. The cost curve associated with BC indicates the least cost associated with obtaining $Z_i^*(M \mid \bar{k})$. If we compare the TLR and BC approaches in terms of biodiversity outcomes, the costs under TLRs will always be at least as high as under BC by definition of BC. Recall, however, that practical implementation of BC is prohibited by lack of information on land values and species.

Figure 1. Opportunity Costs Under Alternative Reserve Selection Algorithms



Although more land can be set aside under the TLR approach there is no guarantee that this will result in an increase in biodiversity protection as measured by *Z(m)*. Figure 2 illustrates biodiversity outcomes under the TLR and MC approaches.[14] We see that for a given land constraint, the MC approach always outperforms the TLR approach but at a decreasing rate. More importantly, we show that for a given budget constraint, TLRs outperform the MC approach in terms of biodiversity protection because the total amount of area available for habitat protection under is greater.

[14] The BC approach yields the same biodiversity outcome as MC by definition.

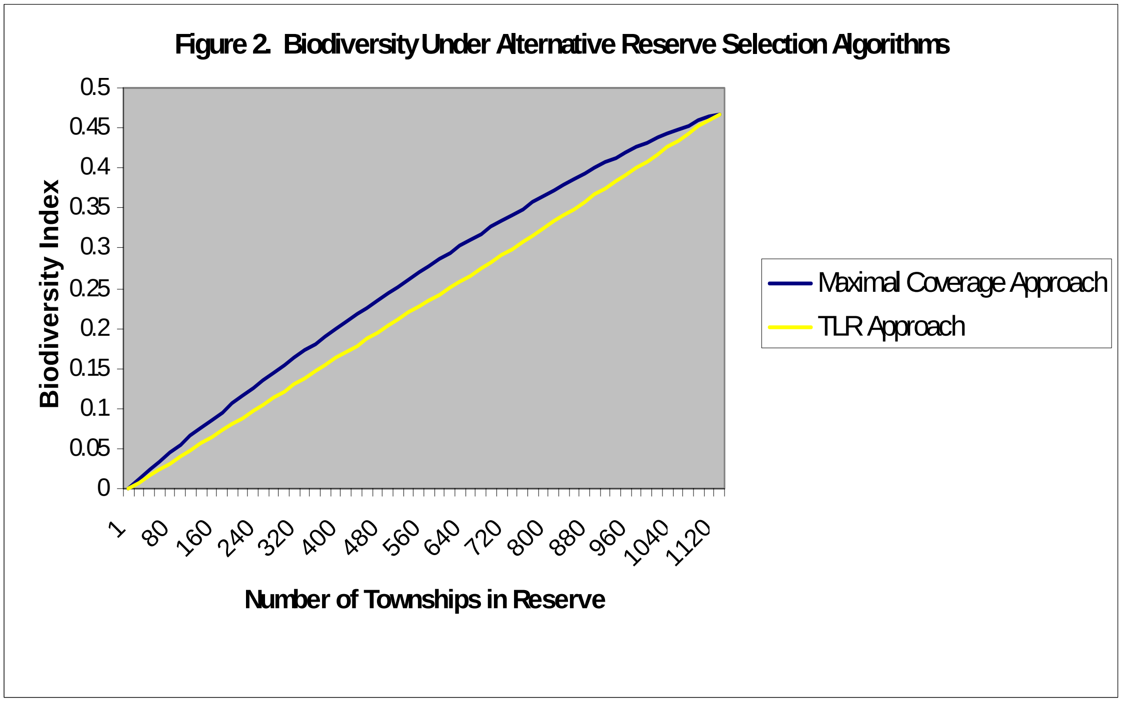


Table 2 illustrates the potential benefit of TLRs using a 12% rule for protected area representation. Costs of a 12% reserve network under MC are $3.4 billion compared to $555 million under TLRs while the level of biodiversity protection under MC is .076 relative to .056 under TLRs.[15] Another way of looking at this is that a 512% increase in costs buys only a 38% increase in the level of biodiversity protection for a 12% landuse constraint. Alternatively these cost savings could be used to increase the level of protection under TLRs to 480 twps, or approximately 42% of the study area. This increases the amount of habitat protected by 243% and the level of biodiversity protection to .195, or by 157% relative to MC. In order to buy an equivalent level of protection under MC, it would be necessary to spend $9.4 billion - a potentially prohibitive amount.

[15] The results are comparable to other estimates of habitat protection costs in the boreal mixedwood. Armstrong, Adamowicz et al. (2003) calculate the discounted present value of lost forestry opportunity from coarse filter management of 5 preferred habitat types over a 200 year

Table 2. Outcomes Under Alternative Reserve Selection Approaches			
Approach	Land Constraint	Cost ($M)	Biodiversity Index
MC	140 twps (~12%)	$3,410	.076
BC*	173 twps (~15%)	$807	.076
TLR	140 twps (~12%)	$555	.056
TLR	200 twps (~18%)	$917	.081
TLR	480 twps (~42%)	$3,340	.195
MC	380 twps (~34%)	$9,360	.190

* Note that BC is not attainable due to incomplete information about true land values.

If decision makers had appropriate price signals, they could implement the BC approach and achieve the level of biodiversity protection afforded by the MC solution at the least cost (807M). Alternatively, TLRs could provide approximately the same level of coverage for 917 million. The TLR approach will always be more expensive than BC for achieving a particular level of biodiversity protection. However one should be cautious about making such comparisons. In the absence of complete species information, the biodiversity measure used in the reserve selection algorithm is only a surrogate for actual biodiversity. Even if land values

rotation. They find the cost of maintaining 12.5% of the natural distribution of each preferred habitat type in the study area is equal to $4453 per ha.

were known, the benefit of the BC approach would depend on the degree to which chosen indicators are successful biodiversity surrogates. Similarly, the cost difference between the TLR and BC programs is the cost of not having the correct land value information to link incentives directly to the biodiversity targets. On the other hand, increasing the level of habitat protected under TLRs reduces the level of risk associated with the use of species based biodiversity indicators.

The TLR algorithm illustrates the benefit of substituting more sites in the reserve network with lower productivity in terms of resource and biodiversity values. In this study TLRs provide large benefits because of the overlap of very productive sites for resource development and biodiversity. Ideally scenario analyses should be undertaken as part of a conservation planning exercise prior to choosing reserve size constraints and the approach for selecting reserves to determine the relative risks associated with different approaches and targets.

5. Conclusion

There are significant obstacles to the implementation of optimal reserve networks. As forested land becomes more scarce, issues of "price, tenure, availability, present and future uses of adjacent land, access, and protection regimes ... are likely to dominate what might be perceived as minor biological differences between sites" (Prendergast, Quinn, and Lawton 1999). In this paper we examine the potential for tradable landuse rights to implement biodiversity objectives through the selection of an ecological reserve network. TLRs are a coarse filter approach to conservation where biodiversity is maintained indirectly through maintenance of habitat. Fine filter systems may still be necessary for species whose requirements are not met through the coarse filter approach. This paper shows that while markets do not allow the regulator to directly control the selection of sites in the ecological reserve network, there is a systematic economic tradeoff between the total amount of habitat protected and the selection of high quality sites for biodiversity protection. By allocating habitat patches to areas with the lowest opportunity cost in terms of development, the total amount of habitat that can be protected at a given cost is increased. Simulation results suggest that the increase in habitat may be quite significant. Furthermore, potential cost savings will be greater in markets where there is substantial heterogeneity in land values. Given the uncertainty surrounding biodiversity outcomes resulting from traditional approaches to reserve design, further research is warranted to better understand the efficacy of decentralized instruments for implementing conservation objectives.

6. References

Alberta Environmental Protection, *Implementing the Revised Timber Damage Assessment: 1995*, Alberta Environmental Protection, Edmonton, A.B. (1995).

Alberta Forest Service, *Alberta phase 3 forest inventory: an overview*, Alberta Energy and Natural Resources, Edmonton, A.B. (1985).

Alberta Resource Development, *Oil and gas fiscal regimes of the western Canadian provinces and territories*, Alberta Resource Development, Edmonton AB. (1999).

Ando, Amy et al., "Species distributions, land values, and efficient conservation," *Science* 279: 2126-2128 (1998).

Armstrong, G.W., Adamowicz, W.L., Beck J.A., Cumming S.G., and Schmiegelow, F.K., "Coarse filter ecosystem management in a non-equilibrating forest," *Forest Science* (in press).

Camm, Jeffrey D., Stephen Solow Andrew Polasky, and Blair Csuti, "A note on optimal algorithms for reserve site selection," *Biological Conservation* 78: 353-355 (1996).

Church, Richard L., David M. Stoms, and Frank W. Davis, "Reserve Selection as a Maximal Covering Location Problem," *Biological Conservation* 76: 105-112 (1996).

Csuti, Blair et al., "A comparison of reserve selection algorithms using data on terrestrial vertegrates in Oregon," *Biological Conservation*: 83-97 (1997).

Cumming, S. G. and Schmiegelow, F. K. A. Effects of the forest matrix, habitat abundance and fragmentation on distributional patterns in bird atlas data. Sustainable Forest Management Network, Working Paper SFM WP 2001-01, Edmonton, Alberta (2001).

Fahrig, L. "How much habitat is enough?" *Biological Conservation* 100:65-74 (2001).

Faith, D.P. and Walker, P.A., "Integrating conservation and development: effective trade-offs between biodiversity and cost in the selection of protected areas," *Biodiversity Conservation* 5:431-446 (1996)

Faith, D.P., Walker, P.A., Ive, J.R., and L. Belbin, "Integrating conservation and forestry production: exploring trade-offs between biodiversity and production in regional land-use assessment, *Forest Ecology and Management* 85:251-260 (1996).

Hof, J. and C. H. Flather, "Accounting for connectivity and spatial correlation in the optimal placement of wildlife habitat," *Ecological Modeling* 88: 143-155 (1996).

Jaarsveld, Albert S. et al., "Biodiversity assessments and conservation strategies," *Science* 279: 2106-2108 (1998).

Margules, C. R. and R. L. Pressey, "Systematic conservation planning," *Nature* 405: 243-253 (2000).

Metrick, Andrew and Martin L. Weitzman, " Conflicts and choices in biodiversity preservation," *Journal of Economic Perspectives* 12: 21-34 (1998).

Montgomery, W., "Markets in licenses and efficient pollution control programs," *Journal of Economic Theory* 5: 395-418 (1972).

Nehring, Klaus; Puppe, Clemens, "A theory of diversity," *Econometrica* 70:1155-1198 (2002)

Panayotou, Theodore. Reducing biodiversity expenditure needs: reforming perverse incentives. from *Investing in Biological Diversity*. OECD, Paris, France, pp. 217-233 (1997).

Pearse, P.H., *Introduction to Forestry Economics*, University of British Columbia Press, Vancouver, B.C. (1990).

Prendergast, John R., Rachel M. Quinn, and John H. Lawton, "The gaps between theory and practice in selecting nature reserves," *Conservation Biology* 13: 484-492 (1999).

Pressey, R.L., "Ad-hoc reservations: forward or backward steps in developing representative reserve systems," *Conservation Biology* 8(3): 662-668 (1994).

Pressey, R. L., H. P. Possingham, and C. R. Margules, "Optimality in reserve selection algorithms: when does it matter and how much," *Biological Conservation* 76: 259-67 (1996).

Polasky, S., J. D. Camm, and B. Garber-Yonts, "Selecting biological reserves cost-effectively: an application to terrestrial vertebrate conservation in Oregon," *Land Economics* 77 (1): 68-78 (2001).

Rusco, F.W. and Walls, W.D. "Competition in a spatial repeated auction market with an application to timber sales," *Journal of Regional Science* 39:449-465 (1999).

Sanjayan, M.A. and Soule, M.E., *Moving Beyond Brundtland: The Conservation Value of British Columbia s 12 Percent Protected Area Strategy,* Greenpeace Canada, Vancouver B.C. (1997).

Schmiegelow, F.K., and Makkonnen, M., "Habitat loss and fragmentation in dynamic landscapes: avian perspectives from the boreal forest," *Ecological Applications* 12:375-389.

Shogren, Jason F.; Tschirhart, John; Anderson, Terry; Ando, Amy, Whritenour; Beissinger, Steven R.; Brookshire, David; Brown, Gardner M. Jr.; Coursey, Don; Innes, Robert; Meyer, Stephen M.; Polasky, Stephen, "Why economics matters for endangered species protection," *Conservation Biology*,13:1257-1261 (1999).

Simberloff, Daniel, "Flagships, umbrellas, and keystones: is single-species management passe in the landscape era?," *Biological Conservation* 83: 247-257 (1998).

Simon, B.M, Leff, C.S., and Doerkson, H., "Allocating scarce resources for endangered species recovery," *Journal of Policy Analysis and Management* 14(3):415-432 (1995).

Tietenberg, Tom. The tradable permits approach to protecting the commons: What have we learned? Proceedings from the CREE 2000 Workshop. Canadian Resource and Environmental Economics Study Group (2000).

Thomas, R.C., Kirby, K.J., and Reid, C.M., "The conservation of a fragmented ecosystem within a cultural landscape - the case of ancient woodland in England, *Biological Conservation* 82:243-252 (1997).

Trzcinski, M.K., Fahrig, L., and Merriam, G., "Independent effects of forest cover and fragmentation on the distribution of forest breeding birds," *Ecological Applications* 9:586-593 (1999).

Underhill, L.G., "Optimal and suboptimal reserve selection algorithms," *Biological Conservation* 70: 85-87 (1994).

van Langevelde, Frank et al., "Competing land use in the reserve site selection problem.," *Landscape Ecology* 15: 243-256 (2000).

Vane-Wright, R. I., C. J. Humphries, and P. H. Williams, "What to protect? - Systematics and the agony of choice," *Biological Conservation* 55: 235-254 (1991).

Vernier, P., Schmiegelow, F.K.A., and S.G. Cumming. "Modeling bird abundance from forest inventory data in the boreal mixedwood forests of Alberta." in Scott, J.M., P.J. Heglund, M.L. Morrison, J.B Haufler, M.G. Raphael, W.A. Wall, F.B. Samson (eds.), *Predicting Species Occurrences. Issues of Accuracy and Scale*. Island Press. Covelo, CA. pp. 559-571 (2002).

Weber, M.. Adamowicz W., "Tradable landuse rights for cumulative environmental effects management," *Canadian Public Policy* (2002)

Weikard, H.P., "Diversity functions and the value of biodiversity," *Land Economics* 78:20-27 (2002).

Weitzman, Martin L., "On diversity," *Quarterly Journal of Economics* 107 (2): 363-405 (1992).

7. Appendix

Proof that market solution is optimal requires the assumption of constant returns to scale for land holdings so that $B_{i'}\ (\sum_j h_{ij}) = c$ is constant. Assume constant returns to scale. Consider the pricing strategy of two firms 'a' and 'b'. The FOC defined by (3) shows the marginal benefit to firm *i* of obtaining the *j*th piece of land at price pj. Let H_i^* denote the optimal portfolio of land assets held by each firm, and $\hat{H}_i$ represent an alternative portfolio.

Lemma 1. If $\hat{H}_i = H_i^*$ for all *i*, then no firm has an incentive to reorganize its land portfolio.

Proof: Assume the contrary. From (3) firm *i* has an incentive to reorganize its portfolio as long as

A1. $$[\ \frac{\partial B_i}{\partial X_j} + \frac{\partial B_i}{\partial H_i}] - p_j > 0\,.$$

However from (2)

A2. $$[\ \frac{\partial B_i}{\partial X_j} + \frac{\partial B_i}{\partial H_i}\ ] - \mu_j\ -\ \lambda \leq 0$$, for all *i* and *j*.

At the optimum assume that firm *a* owns parcel *j*. Then

A3. $$[\ \frac{\partial B_b}{\partial X_j} + \frac{\partial B_b}{\partial H_b}\ ] \leq [\ \frac{\partial B_a}{\partial X_j} + \frac{\partial B_a}{\partial H_a}\ ] = \mu_j\ -\ \lambda\ .$$

Firm *a*'s reservation price is given by $p_j^a\ =\ \mu_j\ +\ \lambda$. Therefore Firm *b* never has an incentive to purchase land from Firm *a* and vice-versa. Finally neither *a* or *b* have an incentive to trade a parcel of land in their portfolio for a parcel of reserve land. From (A3), and the fact that $\mu_k\ =\ 0$ if parcel *k* is in reserve

A4. $$[\ \frac{\partial B_i}{\partial X_k} + \frac{\partial B_i}{\partial H_i}\ ] \leq [\ \frac{\partial B_i}{\partial X_j} + \frac{\partial B_i}{\partial H_i}\ ]$$

Therefore no firm will prefer a parcel in reserve to their portfolio.

Lemma 2. If $\hat{H}_i\ \neq\ H_i^*$, firms will have an incentive to reorganize their portfolio.

Proof. If $\hat{H}_i\ \neq\ H_i^*$, then either

1. the inequality in A3 is reversed and firm *b*'s willingness to pay for site *j* exceeds firm *a*'s reservation price in which case the two firms will make a Pareto optimal sale;

or,

2. the inequality in A4 is reversed and firm *i* is willing to trade a parcel of land from $\hat{H}_i$ for a piece of vacant land.

Proposition. The optimal allocation of land, H_i^*, is a Nash Equilibrium.

Proof: Follows from Lemmas 1 and 2. This outcome depends on constant returns to scale otherwise the marginal value of each site depends on the temporal and spatial order in which parcels are purchased. Under constant returns firms costlessly reorganize their portfolios, building clusters of activity until there is no incentive to reorganize.