Ecological and Economic Dimensions of Fire and Anthropogenic Disturbance in Maquis Woodlands of the Carmel Range: Implications for Planning and Management

Shirra Freeman*
University of Haifa

This paper integrates research in economic valuation, fire disturbance, and land use change in a model that estimates the consequences of man-made and natural disturbance on woodland succession. The proposed expansion of road infrastructure in the Carmel range in northern Israel is developed as a case study. Two woodland succession scenarios are developed, one with existing road infrastructure and one with the proposed expansion. Monetary values are applied to the simulated changes in land cover. The main findings are that the construction of the road in combination with ongoing fire exposures would result in a net loss of 30 ha of natural maquis cover. Of the remaining maquis areas, succession will favour open maquis and there will be a net loss of moderate and dense maquis. The present value of the economic losses associated with these land cover changes is approximately US$640 million including lost benefits from direct and indirect uses as well as option and non-use values. The issue of road expansion is currently the subject of vigorous debate within the conservation and transport planning sectors in Israel. The results of the research have the capacity to support decision making on the scope of public infrastructure projects that have environmental spillovers as well as responses by conservation managers.

Keywords: *Bioeconomics; Economic valuation; Landscape change; Conservation planning; Ecosystem services.*

The Carmel range in northern Israel is one of the country's largest multiple-use woodland ecosystems. It houses large tracts of Mediterranean maquis and planted pine woodland, the Carmel National Park (the Park) and a nature reserve. All are major attractions for tourism and recreation. It contains several towns, a kibbutz, a prison and a quarrying operation. Policy and planning for the range rests with the Nature and National Parks Protection Authority, the Jewish National Fund and a number of government ministries and local authorities. Haifa, Israel's third largest city is on the northern border of the Carmel range, and rapid urbanization and

*Department of Geography and Environmental Studies, University of Haifa, Mt. Carmel, Israel 31905. Email: shirra.freeman@gmail.com

industrialization is ongoing in surrounding areas. Woodland management is a complex undertaking, involving trade-offs among stakeholders such as recreational users, environmental groups, business interests and public bodies. The range is also subject to multiple natural disturbances, especially forest fire. There have been several serious fires in just over 20 years. In 1989, 600 hectares (ha) of woodland, 320 of which were maquis, burned. In 2010, between 4,000 and 7,500 ha were exposed to the worst fire in Israel's history. Smaller wildfires also occurred in 1989 and 2005. Effective planning and management for the range must draw on several disciplines in order to cope with the challenges of multiple jurisdictions, multiple stakeholders and multiple sources of disturbance.

In Israel, multidisciplinary input is mandated in certain planning contexts and the use of decision-support tools such as Environmental Impact Assessment (EIA), Cost-Benefit Analysis (CBA) and conflict resolution is increasing. Nevertheless, consistent integration of these tools within unified frameworks is low with the result that environmental, economic, cultural and other stakeholder concerns tend to be considered in isolation.

As a planning issue, road building within the range has gained prominence since 2000. Initially, the focus was the possibility of transforming an existing low volume road into a major artery connecting Haifa with the northern portion of the Trans-Israel Highway (Route 6) (Tapiero, 2006). Following the 2010 forest fire, several communities in the range requested the paving of existing dirt roads as emergency escape routes (Rinat, 2011). Support and opposition has come from municipalities, citizen groups and environmental interests.

Conflicts can be minimized by expanding the number of stakeholder interests included in decision-making processes in a transparent manner that permits assessment of trade-offs among them. This research addresses the extent to which the choice of units of measure can influence the understanding of trade-offs. Specifically, it examines the scope for measuring ecological change using monetary metrics as part of the assessment of a hypothetical road construction project. The merits of any of the above road projects depend on the costs of construction and maintenance, the benefits of reduced congestion in other parts of the road network, benefits of improved traffic flow to drivers as well as spillovers or externalities (both positive and negative) imposed outside of the transport sector. Each has welfare effects that with the exclusion of spillovers are generally included in CBA and other standard assessments of public projects. Economic valuation of environmental impacts facilitates the inclusion of additional welfare effects. The advantages for planners include improved comparisons of different project specifications and accommodations in woodland management, based on more complete information.

The purpose of this research is to demonstrate a method for integrating environmental and economic factors in assessing spillovers from the road project. The problem of road building in the Carmel range is used because of its policy-relevance and clear economic and environmental dimensions. The two are analyzed using a bioeconomic model that accounts for variability in land-cover density, fire exposures, succession patterns, climax states[1] and the economic value of different maquis habitats.

The conceptual underpinning is the Driver-Pressure-State-Impact-Response (DPSIR) framework that delineates consistent evaluation of different components of an ecosystem over a given period of time (EEA, 1999, 2007). In the context of this research, the drivers are population growth and development. Pressures stem from road expansion. States correspond to the long-term succession and climax of maquis woodlands subjected to different disturbance scenarios. Impacts are the economic values associated with different states. Responses are behaviors of policy makers, planner and managers. The focus of this analysis is on the pressures, states and impacts. The interface between ecological states and economic impacts is central in establishing the links between changes in ecosystem functions (measured in ecological terms) and ecosystem services (measured in monetary terms). The importance of this link allows analysts and decision-makers to deal with two dimensions of the same issue simultaneously. Responses are discussed but not directly analyzed. The main research question is: What are the economic impacts of density changes (biomass and biodiversity) in a maquis woodland landscape subject to pressures associated with forest fire and road expansion? A secondary question concerns the extent to which environmental and economic criteria coincide.

LITERATURE REVIEW

The importance of multidisciplinary approaches in planning has received increasing recognition within the academic, policy, and professional communities since the 1970's (Taplin, 2003). As the field of environmental studies has evolved, methodologies and methods have been formalized (Conrad, 2002; Schroll and Staerdahl, 2001; Vayda, 1983). Bioeconomics, one result of this process, utilizes ecological and economic inputs to develop tools for decision analysis and support (Pearce and Freeman, 1992). Combining economics and ecology is challenging. The former is rooted in a biocentric paradigm with advantages in quantifying risk and physical characteristics as well as changes in both over time and space. The latter is dominated by social-science paradigms stressing theoretical consistency in analyzing agents' preferences and emphasizing choice and ranking of alternatives using the criteria of profit or utility maximization. The synergies between the two come from the capacity of economics to shed light on decision making and ecology's strength in describing the object of decisions (Adamowicz and Veeman, 1998; Brock and Starret, 2000, 2003; Dasgupta and Maler, 2003).

Multiple-use resource management (MURM) is one policy response to the interdisciplinary challenge. MURM facilitated shifts from timber-oriented forest management to management accommodating a range of forest stakeholders in meeting multiple policy objectives (Montgomery, 1996). MURM considerations are implicit in major international guidelines for the planning, construction and use of forest roads (FAO, 1998). Decision tools in MURM include bioeconomic models and monetary valuation of forest ecosystem services (Adamowicz and Veeman, 1998; Bowes and Krutilla, 1989; Crepin, 2003). Typically, timber is not a major disturbance in forests in southeastern regions of the Mediterranean. Rather, the priority is balancing the needs of different stakeholders and understanding trade-offs among land uses given pressures from population growth and development (Palahi et al., 2008).

Forest valuation employs a mix of market prices, public budgets, non-market and quasi-market, hedonic methods depending on the ecosystem services of interest (Pearce and Turner, 1989). Total Economic Value (TEV) is the central concept underlying valuation of projects with environmental spillovers. TEV is the value associated with changes in the quality or quantity of an environmental resource expressed in monetary terms. It is the sum of values associated with direct use (for example, as a factor of production in the case of grazing or as a consumption good in the case of hunting or harvesting biomass for household use), indirect or non-consumptive uses such as certain forms of recreation (e.g.: picnicking and hiking) as well as values associated with systemic ecosystem functions (e.g.: watershed). TEV also includes benefits from retaining the option to use the resource in the future or from the knowledge that it is available to others or simply that it is preserved irrespective of any human use (Pearce and Turner, 1989).

Both economic valuation and ecological studies have been conducted for the Carmel and similar woodlands. Following the extensive fire in 1989, Shechter et al. (1998) conducted a contingent valuation (CVM) study of people's willingness to pay (WTP) to protect the Park. They found that WTP for both use and non-use benefits were quite high, but not entirely reflective of actual behaviors exhibited by donations collected for a fund dedicated to protection. These results are consistent with those from other forest valuation studies (For example: Bateman and Willis, 1999; Carson, 2000). Riera et al. (2007) conducted a choice modeling (CM) study of Catalonian shrubland to elicit public preferences for landscape attributes. They found that in general WTP was highest for denser landscapes and active fire mitigation and soil preservation interventions. Becker and Freeman (2009) valued the benefits of rare, old and ancient trees in Israel using CVM. Shamir et al. (2005) conducted the most comprehensive valuation of maquis areas of the Park to date. It includes values associated with forestation, pasture, medicinal uses of plants, inputs in pharmaceuticals, benefits of an agricultural gene reservoir, landscape amenity, endemism and rarity. They compiled an inventory of vegetation in different maquis habitats. Each species was assessed according to a set of attributes and ranked on a five-point ascending scale. For example, trees that provide shade were ranked more highly than shrubs in the forestation category because of potential contributions to recreation and harvest. Species having uses in multiple categories received weightings in each relevant category. Each habitat type was described by the abundance of species and their related biodiversity rankings. Economic values for individual species and specific attributes were estimated using market and non-market methods. This work is notable for the large number of benefits valued, the incorporation of multiple valuation techniques coupled with the assessment of biodiversity.

Mediterranean maquis is resilient in the long-term. Recovery of ecosystem functions, following single disturbances occurs within 30-40 years in the absence of additional major perturbation (Carmel and Flather, 2004; Diaz Delgado et al. 2002; Trabaud and Lepart, 1980). There is uncertainty as to exact succession trajectories because of stochastic processes governing climate variables, fire and disease. With respect to fire, Diaz Delgado et al. (2002) report that multiple exposures within a short period of time lead to lower biodiversity and dominance by a small number of shrubs and trees. In the Carmel range, multiple exposures within a twenty year span resulted in lower biomass density (van Leeuwen et al. 2010; Malkinson et al. (2011); Malkinson, 2010). Historically, the main anthropogenic pressures have come from wood-cutting, livestock grazing and agriculture (Lehouerou, 1974; Naveh and Carmel, 2003; Zavala and Burkey, 1997). The creation of the Park and reserve in 1972 reduced the intensity of many of these pressures (Kadmon and Harari-Kremer, 1999). Concurrently, socioeconomic and demographic changes have increased pressures associated with industrialization and urbanization. In addition, the incidence and severity of forest fires, another major disturbance, have increased over the last half century. Though fire is traditionally regarded as an environmental disturbance, evidence points to man-made causes as well. Conservation management has increased biomass and therefore fuel and this may also be a contributing factor in the number and severity of fires (Shoshany and Goldshleger, 2002). Wittenberg and Malkinson (2009) find a significant association between proximity to roads and footpaths and the number of fire ignition points.

THE STUDY SITE: INITIAL STATE DESCRIPTION

Woodland areas of the Carmel total approximately 15,700 ha. The Park accounts for 8,500 ha, of which one third (2,833 ha) is a nature reserve (Figure 1). The Park as a whole and many of the forested areas of the range are generally managed according to conservation objectives. This heterogeneous landscape is composed of planted pine forests and Mediterranean maquis of different densities. The three main types considered in this study are open, moderate and dense maquis. Open maquis is dominated by herbs and grasses with scattered dwarf shrubs such as thorny burnet. Tree cover is no more than 20%. Moderate maquis is shrub dominated and contains many of the same species as open maquis, but in higher density. Small species of Palestinian oaks are also found. Tree cover is between 40% and 50 %. Dense maquis is tree dominated with 80%-90 % tree cover. It contains a number of mesophillic species such as sweet laurel and Judas. Shrubs and annuals are found only in well-lit locations (DiCastri et al., 1981). According to Kadmon and Harari-Kremer (1999) maquis accounted for approximately 13,308 ha of the range in 1992. The remainder was planted pine. Pine succession differs from that of maquis and no economic valuation was available for pine areas. Therefore, in this research, pine areas are used in calculating forest composition and fire distribution but excluded from the analyses of succession scenarios and economic impacts.

Ideally for a study of this type, the initial case would be based on data from a single ecological survey mapping the range according to succession stages and fire exposures. Because such a survey was unavailable, two separate sources were used. The base case distribution of habitats is assumed to be that found by Kadmon and Harari-Kremer (1999) -- 23% dense maquis, 24% moderate maquis and 53% open maquis. The base case fire distribution corresponds to the findings of Malkinson et al. (2011). Between 1983 and 2006, 15% of the landscape or 2,400 ha had been exposed to at least one fire. Ignition points were identified in 71% of the cases. Of these, 88% had been exposed once; 10% had had two exposures and 2% had been exposed three or more times. A partition of this data shows that pine woodland is far more susceptible to fire than maquis. Thirty-two percent of planted pine woodland in the range has been exposed to fire on more than one occasion. This amounts to 769 ha or 45% of all burnt areas, a disproportionately large amount given that pine cover is only 15.5%. Tables 1 and 2 summarize forest composition and fire exposures. At the time of writing distributions following the 2010 fire were unavailable.

Table 1: Forest composition with habitat type expressed as a percentage of the total landscape and maquis areas

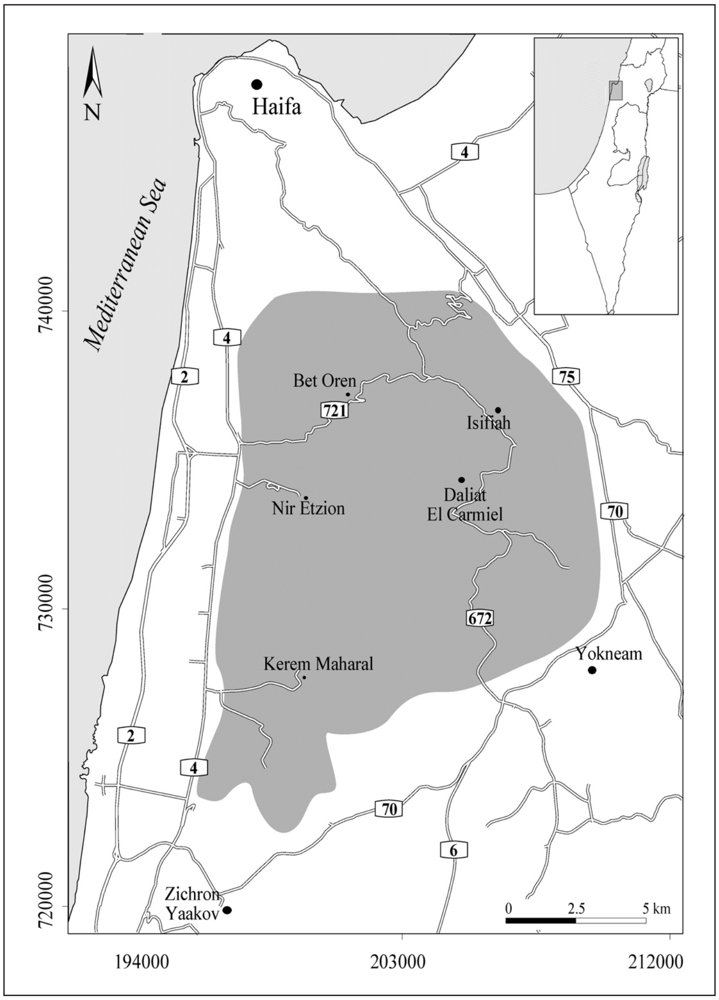
Composition	Areal coverage	% Of total	% Of maquis
Total forest	15,700	100	--
Planted pine	2,392	15	--
Total maquis	13,308	85	100
Dense maquis	3,061	20	23
Moderate maquis	3,194	20	24
Open maquis	7,053	45	53

Table 2: Fire exposures for the Carmel range.

Times exposed	Forest (ha (%))	Maquis (ha (%))	Pine (ha (%))
Never exposed	13,900 (89.0)	12,368 (93.0)	1,523 (67.7)
One	1,504 (9.6)	825 (6.2)	675 (28.2)
Two	171 (1.1)	97 (0.7)	79 (3.3)
Three	34 (0.3)	18 (0.1)	18 (0.8)
Total	15,700 (100)	13,308 (100)	2,392 (100)

The human dimensions of the range include its built aspects and a variety of uses. With respect to roads, the existing system connects with Haifa at two points, in Nesher at sea level (Route 7212) in the northeast and in the Denia neighborhood at the top of the ridge in the north of the city (Route 672). Paved roads also connect the range to Route 4 at the southwest border (Route 721) and Route 70 at the northern border (Route 672). Within the range, these roads connect the two towns, the prison and all major recreation sites. An extensive network of unpaved paths serves hikers, grazers and limited vehicular traffic.

Figure 1: The study area



RESEARCH METHODS

The bioeconomic model is elaborated in two stages. The first computes and compares two ecological states, one, the baseline case in which there is no change in road infrastructure and another, the perturbed case in which road infrastructure is expanded. The second stage of the model estimates the economic impacts or TEV of the state changes.

The ecological baseline is estimated using a forward projection from the initial state using transition probabilities formulated using techniques detailed in Malkinson et al. (2011). These account for numbers of fire exposures in twenty years and the succession stage at the time of the last exposure. For areas not exposed to fire, the base year is 1990. In the perturbed case, 10 kilometers (km) of existing road along the northeast ridge is widened and another 10 km is constructed through woodland. Two effects are considered: Disturbance caused by construction including the permanent conversion of woodland to roadway and ongoing disturbance caused by open access and vehicular traffic that increases air and soil pollution in 2,000 ha along the road's length. In addition to the three main maquis categories (open, moderate and dense), an additional habitat category, "roadside" is added to reflect the likely increase in highly degraded areas adjacent to newly constructed portions of the road. The economic values associated with each case are computed by multiplying Shamir et al.'s (2005) per ha dollar value times the number of ha of each habitat type and then aggregating over all four habitat types in the landscape. The TEV of the environmental changes is the difference between the climax states in the status quo and perturbed cases.

The main assumptions underlying the model are:

- The temporality of the initial distribution of succession stages and fire exposures correspond. The former is based on a 1992 survey and the latter on a 2006 survey.
- The succession stage and fire distributions from these surveys can be generalized over the entire range and the areas affected by the road.
- Fire occurrence is uniformly distributed over all maquis categories.
- The historical transition probabilities are stable over the projected period.
- Succession in all areas follows that of north-facing slopes.

Assumptions one and two permit the nesting of the woodland patchwork and fire distributions. Assumption one is reasonable given the relatively small number of multiple exposures. Assumption two should be taken with some caution. Evidence of increased fire rates over time means projecting the 2006 fire map onto the 1992 patchwork may overestimate the extent of fire in earlier periods. Assumptions two and three were needed to accommodate gaps in available data on changing trends in fire incidence and distribution. Although fires incidence is highest in the northern ridge and eastern slopes of the range, incidence data for different maquis types and time periods were unavailable at the time of writing. Assumption five was made for simplification. Kadmon and Harari-Kremer (1999) and Malkinson et al. (2011)

observe that north-facing slopes have more forward succession trajectories than do south-facing slopes. This assumption may have led to an overestimate of moderate and dense maquis climax states.

THE MODEL

Base Case

Table 3 shows that in 2006, 93% of maquis woodland was unexposed to fire between 1983 and 2006. Of the areas exposed, most had experienced a single fire. Areas exposed more than once were smaller by nearly an order of magnitude or more. The fire distribution over the three types of maquis follows Assumption 3 - areas were exposed in proportion to their share of the landscape.

Table 3: Base case initial state classified by density and number of fire exposures.

Times exposed	Total maquis %	Total maquis Area (ha)	Open (53%)	Moderate (24%)	Dense (23%)
Never exposed	93.0	12,368	6,555	2,068	2,845
One to two	6.9	922	488	221	212
Three	0.1	17	9	4	4
Total exposed	7.0	940	498	226	216
Total maquis (all types)	100.0	13,308	7,053	3,194	3,061

The transition probability matrices in Tables 4.a – c detail succession trajectories observed in selected 30 m X 30 m maquis polygons with different fire exposures. Base year states in the left-hand column are mapped into climax states indicated in the top row. For example, according to the control case observations in 1990 and 1995, moderate maquis remained unchanged with a 69% probability. Succession to open and dense maquis was expected with 3% and 28% probabilistic respectively. Because succession is similar for areas with one and two, the matrix for a single exposure was applied to both.

Based on the matrices and consultation with Malkinson (2009), when there were between zero and two exposures, succession favored moderate and dense maquis. The control case, converged to moderate maquis, unless the initial state was dense, in which case the latter dominated. Following one or two fires, successions from moderate to dense maquis and dense to moderate were slightly higher than in the control. Following three exposures, trajectories shifted dramatically. Zeros in Table 4.c. indicate net losses of maquis woodland. Only open maquis was resilient following three fires.

Table 4: Transition probability matrices.

4a: Control (no fire exposure)

2005 2000	Open	Moderate	Dense
Open	0.40	0.49	0.11
Moderate	0.03	0.69	0.28
Dense	0.00	0.09	0.91

4b: One and two fire exposures

2005 2000	Open	Moderate	Dense
Open	0.74	0.15	0.11
Moderate	0.11	0.49	0.39
Dense	0.02	0.30	0.68

4c: Three fire exposures

1995 1990	Open	Moderate	Dense
Open	0.83	0.16	0.01
Moderate	0.00	0.00	0.00
Dense	0.00	0.00	0.00

The base case climax woodland distribution was estimated by multiplying the 3X3 matrix on the right hand side of Table 3 by the appropriate transition probability matrix. Table 5 summarizes the results:

Table 5: Base case climax state classified by density and number of fire exposures.

Times exposed	Total maquis		Open	Moderate	Dense
	%	Area (ha)	(25%)	(43%)	(35%)
Never exposed	93.00	12,368	2,787	5,361	4,220
One to two	6.93	922	392	245	285
Three	0.07	9	8	1	0
Total exposed	7.00	932	400	246	285
Total maquis (all types)	100.0	13,000	3,187	5,608	4,505

Maquis areal coverage falls by 8 ha permanently degraded following three fires. Succession of the remaining 13,000 ha is forward with proportions of moderate and dense maquis increasing and the proportion of open maquis falling by more than half.

Perturbed Case

The project's specifications are assumed to follow Israeli and international guidelines for secondary roads and rural highways[2]. The road's width will average 18 m and will accommodate four lanes and shoulders. Deviations from the average will depend on terrain. Some of the route will follow the ridge but portions, especially in the east will follow steep, curving slopes.

The immediate effect of road construction will be the permanent removal of 42 ha, or 0.3%, of the base case maquis habitat. Thirty ha of woodland will be eliminated; 18 due to new construction and 12 due to widening. An additional 12 ha will be permanently converted to roadside habitat. Roadside habitat is a term used by Shamir et al. (2005) to describe areas adjacent to roads and paths that have low vegetative cover because of heavy trampling.

Table 6: Perturbed case - distribution in climax state.

Times exposed	Total maquis		Open	Moderate	Dense	Roadside	Loss
	%	Area (ha)	(27%)	(44%)	(29%)		
Never exposed	89.55	11,872.38	2,603.65	5,319.51	3,949.22		
One to two	6.68	885.14	376.13	234.82	274.18		
Three	0.07	9.09	7.54	1.42	0.12		
Total exposed	6.75	894.22	383.67	236.25	274.12		
Non-succeeding open maquis	3.71	491.31	491.31				
Total	100	13,257.94	3,478.63	5,555.76	4224.52	12	30

In the long-run, soil pollution from runoff, possible air pollution effects and trampling resulting from easier access to more areas of the woodland are expected as a result of ongoing use of the road. These effects are assumed to extend 500 m on either side of the road. At any point in time, the effect of the disturbance depends on the time since the last fire exposure. During the first four years after exposure to fire, woodland is highly sensitive to disturbance and unlikely to succeed beyond open maquis if subjected to the disturbances considered here (van Leeuwen, 2010). On average, in any year, 41 ha of woodland (940 ha exposed over 23 years) will be exposed to fire. This means that each year, 164 ha or 1.23% of the landscape will be in a critically sensitive state. Over the entire 20 km length of the road, a total of 2,000 ha is subject to ongoing disturbance and 1.23% of this area, or 24.57 ha will be in the critical state at any point in time. Projecting this over 20 years of road usage, approximately 491 ha will be subject to disturbance in the critical state.

The climax state for the perturbed forest reflects the removal of 491 ha from succession and assumes that these areas persist as open maquis. The remaining 12,775 ha of maquis succeeds according to the transition probabilities. Table 6 summarizes the perturbed climax state.

Comparing the Base and Perturbed Outcomes

Comparing the climax landscapes in the base and perturbed cases indicates the expected change in the composition of the Carmel woodland that could be attributed to the proposed road. Table 7 details the woodland losses and transition favoring open maquis. Fourteen percent or 491 ha of this open maquis are not expected to succeed. In total, 543 ha have been removed from the maquis succession patterns given in the transition matrices. This ecological shift is the basis for assessing the marginal economic impact stemming from the proposed road.

Table 7: Comparison of land cover in the base and perturbed cases.

	Total land cover (Ha)	Open maquis (ha)	Moderate maquis (Ha)	Dense maquis (ha)	Roadside (ha)
Base case	13,300	3,187	5,608	4,505	--
Perturbed case	13,270	3,479	5,554	4,224	12
Marginal change	(30)	292	(54)	(281)	12

ECONOMIC BENFITS ASSOCIATED WITH LAND COVER

In this section, the habitat values estimated in Shamir et al. (2005) are applied to the changes in the woodland landscape. The TEV estimate includes the following ecosystem services:

1. **Forestation** derived from the standing woodland and harvest -- Benefits include recreation, gathering and timber harvest as well as the benefits from systemic functions such as watershed, habitat and biodiversity. Benefit estimates for timber and gathering were based on market prices and the remaining forestation benefits had non-market sources of value.
2. **Pasture** measured as the net savings to ranchers from grazing instead of buying commercial feed.
3. **Medicinal** from the direct use of herbs and plants – Comparable commercial products were used as a proxy for the value of medicinal uses of species.
4. **Agricultural gene reservoir** as a partial measure of the value of genetic diversity – This estimate used the commercial value of cultivated seeds as a proxy for the market value of genetic diversity.
5. **Landscape** derived from the benefits of soil conservation and from the benefits associated with the entire landscape, for example, visual amenity. Non-market stated preference and hedonic methods were used in the estimation.
6. Pharmaceutical from the use of indigenous species as inputs in the production of pharmaceutical products – These were based on the value to firms from using specific plant species.

7. Endemism and rarity and their inter-relationship – Endemic species may be rare or abundant, but because they are uniquely suited to a particular environment, loss is irreversible if the species become extinct. The loss of species that are locally rare, but not endemic is not irreversible, because reintroduction from other areas is feasible. Measurements were based on non-market valuation methods.

TEV of the Carmel Forest Landscape

The mixture of plant species and densities provides a distinct set of benefits for each maquis habitat. In Shamir et al.'s model, the TEV of the landscape is the sum of the products of the value of each habitat times its areal (ha) coverage in the landscape. The value of each habitat type is the sum over all species of the products of each species' biodiversity-weighted economic value times the number of plants found in 4,000 m^2 sample plots for each habitat.

A species or species group is denoted by *h*; *i* represents habitat type: (1) dense maquis, (2) moderate maquis, (3) open maquis and (4) roadside. Specific attributes or benefits associated with species are denoted by *j*. The abundance of a species or group per ha of a given habitat is a_{hi} and the ecological weight as a function of habitat type and species' attributes is w_{hj}. The economic value associated with attribute *j* is denoted by q_j. The TEV of one ha of woodland habitat of type *i*, tev_i is given by:

1. $$tev_i = \sum_{h=1}^{m} a_h \sum_{j=1}^{n} w_h \, q_j$$

If A_i is the total number of ha of habitat i, then the TEV for each habitat type over the landscape is

2. $$TEV_i = (tev_i)(A_i)$$

Aggregating over all land cover types gives the TEV of the range:

3. $$TEV = \sum_{i=1}^{4} TEV_i$$

Habitat values, capitalized to infinity at 3% are summarized in Table 8. Greater diversity and density confer more benefits and are valued more highly. Thus, dense maquis has the highest economic value. Nevertheless, even the more degraded roadside habitat has significant economic value because it contains a number of commercially valuable species.

Table 8: Habitat values (Source: shamir et al., 2005).

Land cover type	Value (us$mm/ha)
Dense maqus	15.00
Moderate maquis	14.50
Open maquis	12.00
Roadside	10.25

Table 9 summarizes the expected TEV of the changes detailed in the comparison of the base and perturbed cases (see Table 7). It is the product of per ha values times the loss or gain in areal coverage.

Table 9: Total economic value of woodland change.

	Open Maquis	Moderate maquis	Dense Maquis	Roadside
Land cover change (ha)	291.58	(51.82)	(281.43)	12.00
Δ*Tev* (us$ mm/ha)	3,498.98	(751.34)	4,221.44	123.00
Δ*Tev current value* (Us$ mm/ha)		(1,350.79)		
Δ*Tev present value (3% over 25 years)* (us$ mm/ha)		(640.00)		

The current value of the landscape changes due to the environmental disturbance caused by the construction and operation of the road are in the range of US$1.35 billion. Because the climax states will be realized 20-30 years in the future, this sum must be discounted to reflect the passage of time. The last row of Table 9 is the present value of this future TEV discounted at a rate of 3%. This amounts to US$640 million or US$32 million on an annualized basis. Given the 2011 Israeli population of 7.5 million, the present value of woodland losses per capita are in the range of US$4.27 per annum.

DISCUSSION

This research demonstrates a method for assessing several dimensions of man-made pressures on the Mediterranean maquis landscape in the Carmel range. An ecological model of the effects of fire disturbance on woodland succession is layered with a model of man-made disturbance and both are embedded in an economic valuation model. Demonstrating the potential of using this bioeconomic modeling platform to support integrated transport and conservation planning was made possible by the availability of quantitative, empirical analyses in Shamir et al. (2005) and Malkinson et al. (2011). The results show considerable scope for integrating economics and ecology; however economic and ecological criteria do not always coincide and this must be taken into account in designing and assessments public projects.

The base and perturbed cases offer a plausible scenario for landscape change following road infrastructure expansion. This scenario indicates that long-term succession is expected to favor open maquis with declining biomass density and biodiversity over the landscape. Overall, the density reduction causes significant economic losses. A closer inspection of the results shows that economic and ecological metrics diverge for the most degraded habitat (roadside). In general, more dense and diverse habitats are more highly valued. Nevertheless the relatively high value (US$10.25 million per ha) assigned to roadside habitat does not reflect its ecological value. Similarly, the valuation may not take into account the overall landscape which because of systemic ecosystem functions contributes to overall ecological integrity.

For forest managers, these results open the question of appropriate responses to pressures and changing states, for example, more intensive silviculture and fire management. For road planners considering environmental spillovers, these results open the possibility of alternate routes and road design; for example, minimizing the conversion of standing woodland by including tunnels and the incorporation of existing unpaved paths (Tapiero, 2006).

In the context of decision-making frameworks, applying an "environmental" CBA rule to maximize the net benefit of the project taking into account the conventional costs of road construction and maintenance, the benefits of reduced congestion and improved traffic flow, environmental impacts and the costs of conservation management would also be made possible by integrating the results of valuation. Valuation could also inform policy on the sharing of costs and benefits among different stakeholders. Stakeholders are clearly delineated and if the net benefits of the project are positive and the road is built, then options for distributing benefits and costs can be identified. For example, changes in conservation management may mitigate the damages caused by the road and lessen the US$640 million burden that will be borne by the Israeli public under the scenario above. Interventions effectively shift the burden from forest stakeholders to public budgets. Similarly, alterations in the specifications of the road that reduce environmental spillovers would shift part of costs to the transport sector and away from forest users.

There are several caveats regarding the results. First valuation and CBA can inform policy but do not provide the normative basis for decisions on questions of distribution (Randall, 2002). Nor do they offer guidance on resolving the divergence between ecological and economic criteria observed in the results.

A third caveat is contextual. The analysis is static and non-spatial. The model compares two woodland climax states without taking into account the accumulation of value over time. While this is sufficient to compare the base-case and perturbed scenarios, it is inadequate for the purposes of a CBA and more dynamic analyses. In order to compare all costs and benefits of the project, the same time frame must be applied to the road project and its impacts. Thus, the value of changes from the time at which the project is implemented must be computed otherwise, the environmental impacts will be undervalued. In addition, the analysis was conducted prior to the fire of 2010. In order to be useful for planners, an updated spatial mapping of succession stages and fire distributions are needed to accurately parameterize the initial state (length of time since last fire and succession state). Similarly, the specifications for the project are hypothetical and the single perturbed scenario will be too limiting.

Finally, the valuation data used as input may require further consideration. In particular, while the valuation study used is the most complete available, it was restricted to a limited sampling within the Park. While generalizing market values is probably reasonable, recreation and non-use values may not be the same over the entire range. Similarly, wider sampling of vegetation may reveal differences between the Park and the other parts of the range.

The limitations above are specific to this case study and can be resolved in future research. Moreover, the framework elaborated is a potentially powerful addition to planners' toolkits since it can be easily integrated with EIA, CBA and other existing tools. Valuation is already widely used in settling environmental damage claims[3] and increasing its use in wider planning contexts is a logical step forward.

NOTES

1. Succession refers to the transformation of woodland over time according to a combination of physical factors, structure and species composition. Climax refers to a state that is stable over a relatively long period if there are no major disturbances. The classification is not rigid and does not imply the absence of change. The period varies from landscape to landscape and stability is relative to the composition and density compared with earlier succession stages (Knowler and Lovett, 1996).
2. The specifications in this study are based roughly on Israel's Ministry of National Infrastructure's, Department of Public Works' guidelines for secondary roads In Israel (http://www.mni.gov.il/mni/he-il) and FAO guidelines for forest roads (FAO, 1998).
3. Programmes such as the U.S. Comprehensive Environmental Response, Compensation and Liability Act (CERCLA/".Superfund".) and its European counterpart, REMEDE base part of their compensation calculations on environmental valuation (REMEDE, 2007).

REFERENCES

Adamowicz, W. and Veeman, J. (1998) Forest policy and the environment: Changing paradigms. *Canadian Public Policy*, XXIV. Supplement no. 2.

Bateman, I. and K. Willis (eds.) (1999) *Valuing Environmental Preferences: Theory and Practice of the Contingent Valuation Method in the US, EC and Developing Countries.* Oxford, UK: Oxford University Press.

Becker, N. and Freeman, S. (2009) The economic value of old growth forest assets. *Journal of Forest Policy and Economics,* 11(8):608-15.

Bowes, M. and Krutilla, J. (1989) Economics and public forestland management. *Natural Resources Journal,* 29(3):737-749.

Brock, W. and Starret, D. (2000) Nonconvexities in ecological management problems. Stockholm: Beijer Institute for Ecological Economics, Working Paper.

_____ . (2003) Managing systems with non-convex positive feedback. *Environmental and Resource Economics*, 26: 575-602.

Carmel, Y. and Flather, C. (2004) Comparing landscape scale vegetation dynamics following recent disturbance in climatically similar sites in California and the Mediterranean Basin. *Landscape Ecology*, 19:573-590.

Carson, R.T. (2000) Contingent Valuation: A User's Guide. *Environmental Science and Technology,* 34: 1413-1418.

Conrad, J. (2002) Limitations to interdisciplinarity in problem oriented social science research. *Journal of Transdisciplinary Environmental Studies*, 1:1-15. available online at: http://www.journal-tes.dk.

Crepin, A.-S. (2003) Multiple species boreal forests - what Faustmann missed. *Environmental and Resource Economics*, 26:226-246.

Dasgupta, P. and Maler, K.-G. (2003) The economics of non-convex ecosystems: An introduction. Environmental and Resource Economics, 26:499-525.

DiCastri, F., Goodall, D. and Specht, R. (1981) *Mediterranean-Type Shrubland, First Edition*. Amsterdam: Elsevier Science.

Díaz-Delgado, R., Lloret, F., Pons, X., and Terradas, J. (2002) Satellite evidence of decreasing resilience in Mediterranean plant communities after recurrent wildfires. *Ecology,* 83: 2293–2303.

EEA (1999) Environmental indicators: Typology and overview. Technical report No 25. Copenhagen: European Environment Agency.

EEA (2007) EEA Integrated Assessment Portal: The DPSIR framework used by the EEA. available at: http://ia2dec.ew.eea.europa.eu/knowledge_base/Frameworks/doc101182

FAO (1998) Report on seminar on environmentally sound forest roads and wood transport. FAO Forestry Division. http://www.fao.org/docrep/X0622E/x0622e01.htm

Kadmon, R. and Harari-Kremer, R. (1999) Landscape scale vegetation dynamics of perturbed Mediterranean maquis. *Journal of Vegetation Science*, 10(3):393.401.

Knowler, D. and Lovett, J. (1996) Training manual for environmental assessment in forestry. Prepared for FAO Regional Project Forestry Planning and Policy Assistance in Asia and the Pacific (GCP/RAS/137/JPN); available at http://www.fao.org/docrep/005/ac625e/ac625e00.htm.

Lehouerou, H. (1974) Fire and vegetation in the Mediterranean basin. In *Proceedings Annual Tall Timer Fire Ecology Conference*. 3:237-277.

Malkinson, D. (2009) pers corr.

Malkinson, D., Wittenberg, L., Beeri, O. and Barzilai, R. (2011) Effects of Repeated Fires on the Structure, Composition and Dynamics of Mediterranean Maquis: Short- and Long-Term Perspectives *Ecosystems*, 14: 478–488.

Montgomery, C. (1996) Forest policy and risk: Issues and recent trends in the U.S. *Ecological Economics*, 16:65-72.

Naveh, Z. and Carmel, Y. (2003) The evolution of the cultural Mediterranean landscape in Israel as affected by fire, grazing and human activities. In Wasser, S. (ed.) *Evolutionary Theory and Processes: Modern Horizons Papers in Honour of Eviatar Nevo*. Dordrecht: Kluwer, 337-409.

Palahi, M., Mavsar, R., Gracia, C., and Birot, Y. (2008) Mediterranean forests under focus. *International Forestry Review*, 10(4):676-688.

Pearce, D. and Freeman, S. (1992) Theme paper 1: Informational requirements for policy decision-makers. In Proceedings of the OECD International Forum on Environmental Information for the Twenty-First Century. OECD. Paper presented at the OECD International Forum on Environmental Information for the Twenty-First Century. May 21-24, 1991.

Pearce, D. and Turner, K. (1989) *Economics of Natural Resources and the Environment.* Hemel, Hempstead, UK: Harvester Wheatsheaf.

Randall, A. (2002) Benefit cost considerations should be decisive when there is nothing more important at stake. In Bromley, D. and Paavola, J. (eds.) *Economics, Ethics and Environmental Policy: Contested Choices.* Malden, Mass.: Blackwell Publishing, 53-68.

REMEDE (2007) Resource Equivalency Methods for Assessing Environmental Damage in the EU (REMEDE) website: http://www.envliability.eu/

Riera, P., Penuelas, J., Farreras, V. and Estiarte, M. (2007) Valuation of climate-change effects on Mediterranean shrublands. *Ecological Applications*, 17(1):91-100.

Schroll, H. and Staerdahl, J. (2001) The concept of transdisciplinary environmental studies. *Journal of Transdisciplinary Environmental Studies*, Preview 6. also available online at: http://www.journal-tes.dk.

Shamir, S., Zeitouni, N. and Shechter, M. (2005) Economic valuation of habitats - The case of the Mediterranean grove. In Proceedings of the Seventh Annual BIOECON Conference *on Economics and the Analysis of Ecology and Biodiversity*. Paper presented, 20-21 September 2005. Available at: http://www.bioecon.ucl.ac.uk/10papers7.htm.

Shechter, M., Reiser, B., and Zaitsev, N. (1998) Measuring passive use value: Pledges, donations and CV responses in connection with an important natural resource. *Environmental and Resource Economics, 12(4):457-478.*

Shoshany, M. and Goldshleger, N. (2002) Land-use and population density changes in Israel, 1950-1990: Analysis of regional and local trends. *Land Use Policy*, 19(1):123-133.

Tapiero, A. (2006) Road 6 - conflicts along its length and breadth. Presentation at the Conference: Road 6 - Lessons from the Past, Directions for the Future. Haifa: Technion (Hebrew)

Taplin, R. (2003) Defining the environmental doctorate: Education for sustainable development. Presentation at: Defining the Doctorate: AARE (Australian Association for Educational Research) Mini-Conference, 2-4 October 2003.

Trabaud, L., and Lepart, J. (1980) Diversity and stability in garrigue ecosystems after fire. *Plant Ecology*, 43: 49–57.

van Leeuwen, W.J.D., Casady, G.M., Neary, D.G., Bautista, S., lloza, J.A., Carmel, Y., Wittenberg, L., Malkinson, D., and Orr, B.J. (2010) Monitoring post-wildfire vegetation response with remotely sensed time-series data in Spain, USA and Israel. *International Journal of Wildland Fire,* 19:75–93

Vayda, A. (1983) Progressive contextualization: A method for integration of social and biological research in the man and the biosphere programme. *Human Ecology*, 11(3):265-281.

Wittenberg, L. and Malkinson, D. (2009) Spatio-temporal perspectives of forest fire regimes in a maturing Mediterranean mixed pine landscape. *European Journal of Forest Research,* 128(2):297-304.

Zavala, M. and Burkey, T. (1997) Application of ecological models to landscape planning: The case of the Mediterranean basin. *Landscape and Urban Planning*, 38:213-227.