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Impact Evaluation of Forest Conservation Programs: Benefit-Cost Analysis, Without the Economics

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Abstract Economists are increasingly using impact evaluation methods to measure the effectiveness of forest conservation programs. Theoretical analysis of two complementary economic models demonstrates that the average treatment effect on the treated (ATT) typically reported by these studies can be related to an economic measure of program performance only under very restrictive conditions. This is because the ATT is usually expressed in purely physical terms (e.g., avoided deforestation) and ignores heterogeneity in the costs and benefits of conservation programs. For the same reasons, clinical trials are a misleading analogy for the evaluation of conservation programs. To be more useful for economic analyses of conservation programs, impact evaluations should work toward developing measures of program outcomes that are economically more relevant, data that would enable the evaluation of impacts on forest degradation (not just deforestation) and primary forests (not forests in general), better estimates of spatially disaggregated treatment effects (not program-wide averages), and better information on the accuracy of estimated treatment effects as predictors of future risks.

Keywords Conservation economics · Deforestation · Degradation · Impact evaluation · Primary forest · Protected area

1 Introduction

Environmental economics research has long drawn attention to the important contributions that forests can make to sustainable development (Pearce et al. 1990). Over the past decade, economic studies on forest conservation have increasingly used quasi-experimental impact evaluation methods to measure the effectiveness of conservation programs, such as protected areas, payments for ecosystem services, and community-based management (Pattanayak et al.

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2010; Miteva et al. 2012; Ferraro et al. 2012).[1] In contrast to benefit-cost analysis, which evaluates prospective programs before implementation (*ex ante* analysis), impact evaluation analyzes programs that are complete or have been underway for a period of time (*ex post* analysis). Impact evaluation methods include matching, instrumental variables, regression discontinuity, and difference-in-differences (Ravallion 2008; Imbens and Wooldridge 2009).

Though varied, impact evaluation methods have the common objective of identifying a program's causal effect on observed outcomes, by implicitly or explicitly constructing a counterfactual to which observed outcomes can be compared. Avoided deforestation is the most common outcome measure in impact evaluations of forest conservation programs (Blackman 2013).[2] Construction of a counterfactual enables impact evaluations to eschew before/after comparisons of deforestation rates, which could be confounded by factors correlated with program timing or decisions made by program participants or administrators, in favor of rigorous without-the-program/with-the-program comparisons (Ferraro and Pattanayak 2006). Impact evaluations are thus able to draw quantitative, causal inferences, such as that a conservation program reduced deforestation by X hectares or Y percent.

Studies in this literature argue that progress toward conservation goals requires knowing "what works and where" (Ferraro and Pattanayak 2006, p. 482), but they also report that "the way in which *ex post* impact studies … can inform *ex ante* conservation planning exercises to site new protected areas needs further exploration" (Ferraro et al. 2013, p. 6). I agree with both of these statements, and in this paper I attempt to elucidate the usefulness of information from *ex post* impact evaluations for *ex post* evaluations that define "what works" in economic terms. I assume that conservation has an economic objective: to maximize the net present or future value of the difference between conservation benefits and conservation costs. While actual conservation agencies might pursue noneconomic objectives, my assumption of an economic objective is consistent with a well-developed body of literature on conservation economics (Polasky et al. 2005). In any event, my analysis will reveal the differences between *ex post* conservation-program evaluations that measure a noneconomic outcome, such as the area of avoided deforestation, and *ex post* evaluations that are instead conducted from an economic standpoint and incorporate information on benefits and costs.

In the following two sections, I present two stylized economic models of *ex post* conservation evaluation. The first is a two-period model in which a conservation agency protects a subset of a large number of forested sites that are at risk of deforestation. This model highlights the interrelated influences of heterogeneity of costs, benefits, and deforestation risks on the economics of the agency's protection decisions. The second model reverses the dimensions, with the number of sites reduced to just two but the time horizon running to infinity. Despite the different setup, this model yields results consistent with those from the first model: in particular, avoided deforestation is a poor proxy for an economic measure of a forest conservation program's impact. After presenting these models, I comment on clinical trials as an analogy for evaluations of conservation programs. I wrap up the paper by suggesting ways to make impact evaluations more useful for economic analyses of conservation programs, including a few comments on using *ex post* studies to inform *ex ante* conservation-planning decisions.

[1] Experimental methods have also been used to evaluate conservation programs, but not as commonly. For an early example, see The and Ngoc (2006).

[2] Several impact evaluations of conservation programs have used poverty alleviation as an outcome measure (Sims 2010; Andam et al. 2010; Ferraro et al. 2011; Ferraro and Hanauer 2014).

2 A Two-Period Model of Conservation Evaluation with Many Sites

A simple two-period model is sufficient for demonstrating the relationship between an impact evaluation that determines the causal effect of a conservation program on avoided deforestation and an evaluation of the same program in economic terms. Suppose that a public conservation agency has a budget, C, to be spent on protecting forested sites at risk of conversion to agriculture or other land uses. There are n sites, and a given site i costs c_i to protect (e.g., the cost of acquiring the site). The agency's budget is not sufficient to protect all the sites:

$$C < \sum_{i=1}^{n} c_i. \tag{1}$$

The agency therefore needs to decide which sites to protect. It makes this decision and incurs protection costs in the first period, and it observes which sites were deforested in the second period. The second period is also when the conservation program is subjected to an *ex post* impact evaluation.

2.1 Impact Evaluation of the Program

In the language of impact evaluation (see Ch. 2 in Angrist and Pischke 2009), protection is the *treatment* in this model. A site's treatment status is given by the binary variable s_i, with $s_i = 1$ indicating protection and $s_i = 0$ indicating no protection. Deforestation is the *outcome*. There are two potential outcomes for any site, with d_{1i} denoting the outcome if a site is protected ($s_i = 1$) and d_{0i} denoting the outcome if it is not protected ($s_i = 0$). Assume that the outcome is also binary: $d_{1i} = 1$ and $d_{0i} = 1$ indicate that a site was deforested, while $d_{1i} = 0$ and $d_{0i} = 0$ indicate that it was not deforested. This binary treatment/binary outcome framework is common in the impact evaluation literature on forest conservation programs.[3]

For any given site, the evaluator observes only one of the two potential outcomes, d_{1i} or d_{0i}. The observed outcome, d_i, is related to the two potential outcomes by the relationship,

$$d_i = d_{0i} + s_i(d_{1i} - d_{0i}), \tag{2}$$

which yields d_{0i} if $s_i = 0$ (the site is not protected) and d_{1i} if $s_i = 1$ (the site is protected). The difference in parentheses, $d_{1i} - d_{0i}$, is the *causal effect* of protection on deforestation. This is what the impact evaluator would like to know, but direct measurement of it is impossible with only one of the two potential outcomes being observed.

Instead, the evaluator must infer the *average* effect of protection from observed outcomes for the two different groups of sites: those that were protected, and those that were not. Using $i = P$ and $i = NP$ to denote these respective groups, the average difference in observed outcomes between the two groups is given by $\mathrm{E}[d_P] - \mathrm{E}[d_{NP}]$. With the deforestation outcome being binary, this difference can range from -1 (deforestation occurs in no protected sites and all unprotected sites) to +1 (deforestation occurs in all protected sites and no unprotected sites). The difference can be decomposed into two components,

$$\mathrm{E}[d_P] - \mathrm{E}[d_{NP}] = \{\mathrm{E}[d_P] - \mathrm{E}[d_{0P}]\} + \{\mathrm{E}[d_{0P}] - \mathrm{E}[d_{NP}]\}, \tag{3}$$

where d_{0P} is the unobserved potential outcome for a protected site if it had not been protected. The first term on the right side, $\mathrm{E}[d_P] - \mathrm{E}[d_{0P}]$, gives the average causal effect of protection.

[3] One could instead employ a continuous outcome measure, such as the proportion of a spatial unit that was deforested.

It is known as the *average treatment effect on the treated* (ATT) in the impact evaluation literature. The second term, $\mathrm{E}[d_{0P}] - \mathrm{E}[d_{NP}]$, is the selection bias. If its value is nonzero, then it drives a wedge between the average difference in observed outcomes and the ATT, with the former exceeding the latter when $\mathrm{E}[d_{0P}] > \mathrm{E}[d_{NP}]$ and the opposite when the inequality is reversed.

Impact evaluation methods aim at estimating the ATT. Demonstrating the ATT's economic interpretation is facilitated if we make one additional assumption: protection is fully effective, with no protected sites being deforested ($d_P = 0$). Under this assumption, the ATT simplifies to $-\mathrm{E}[d_{0P}]$. As noted in the introduction, the impact evaluation literature typically defines the ATT for forest conservation programs in terms of avoided deforestation, not deforestation. Applying this definition, the ATT is given by the positive quantity $\mathrm{E}[d_{0P}]$, which corresponds to

$$\frac{\sum_{i=1}^{n} s_i d_{0i}}{\sum_{i=1}^{n} s_i}. \tag{4}$$

This is just the fraction of protected sites that would have been deforested in the absence of protection. (Unprotected sites have $s_i = 0$and thus drop out.) A larger fraction—i.e., a larger ATT—indicates that the conservation program caused a larger reduction in deforestation.

2.2 Economic Evaluation of the Program

Now consider an evaluation with an objective expressed in economic rather than physical terms. Assume that the evaluator knows the conservation value of each site, v_i, which is expressed in the same units as the protection cost (i.e., money or another numeraire).[4] The evaluator also knows the cost of protecting each site. The gross value associated with protecting a given site is $v_i - c_i e^{\rho}$, where ρ is the social discount rate, while the gross value associated with leaving the same site unprotected is $(1 - d_{0i})v_i$, which yields 0 or v_i depending on whether the site would have been deforested ($d_{0i} = 1$) or not ($d_{0i} = 0$). These values are expressed in future terms, from the *ex post* standpoint of the second period. The net future value of protecting the site is the difference between these two expressions, $d_{0i} v_i - c_i e^{\rho}$.

The aggregate net future value across all the protected sites is

$$NFV = \sum_{i=1}^{n} s_i d_{0i} v_i - \sum_{i=1}^{n} s_i c_i e^{\rho}. \tag{5}$$

Assuming that the budget constraint is binding, i.e., $C = \sum_{i=1}^{n} s_i c_i$, this can be rewritten as the benefit-cost ratio for the agency's program,

$$1 + \frac{NFV}{Ce^{\rho}} = \frac{\sum_{i=1}^{n} s_i d_{0i} v_i}{\sum_{i=1}^{n} s_i c_i e^{\rho}}. \tag{6}$$

The benefit-cost ratio is greater than one when *NFV* is positive and less than one when *NFV* is negative.

[4] The assumption that the evaluator has information on site-specific values is a big one, of course, easier to make in a conceptual setting than in practice. Improvements in spatial datasets and ecosystem service models are making the estimation of conservation benefits at large spatial scales increasingly possible, however. For examples, see Naidoo and Ricketts (2006) and Bateman et al. (2013).

2.3 Economic Interpretation of the ATT

The right side of (6) is similar to the ATT given by (4) in that both expressions are ratios of sums, with the numerator involving s_i and d_{0i} and the denominator involving s_i. The ATT contains none of the economic information on the right side of (6), however: no benefits (v_i), no costs (c_i), and no discount rate (ρ). In the general case, there is thus no reason to expect the ATT to equal the benefit-cost ratio for the conservation program.

Further insight can be gained by considering the special case in which neither benefits nor costs vary across sites, so that $v_i = v$ and $c_i = c$. In that case, (6) can be rewritten as

$$1 + \frac{NFV}{Ce^{\rho}} = \frac{v}{ce^{\rho}} \left(\frac{\sum_{i=1}^{n} s_i d_{0i}}{\sum_{i=1}^{n} s_i} \right). \tag{7}$$

The right side is now the product of the constant benefit-cost ratio for an individual site and the ATT. It implies that, in the absence of heterogeneous benefits and costs, differences in the benefit-cost ratio across alternative conservation programs are driven entirely by differences in the average avoided deforestation rates for the programs. In this special case, one could therefore use the ATT to evaluate the agency's program against other feasible programs that the agency could have implemented instead (i.e., ones that were within its budget constraint). The agency's program would be revealed as the best program, in a benefit-cost ratio sense,[5] if and only if no other feasible program had an average avoided deforestation rate that exceeded its ATT.[6] Conducting this analysis would require estimates of not only the ATT for the agency's program but also the site-level avoided deforestation rates (the d_{0i}s) for all n sites, including ones not included in its program. These estimates would be needed to calculate the average avoided deforestation rates for the other programs—essentially, simulated ATTs.

Even in this highly restrictive case, the ATT would not provide sufficient information for determining if the agency's program passed a benefit-cost test: was the benefit-cost ratio at least as great as one? Calculating the benefit-cost ratio requires information on the economic parameters in $\frac{v}{ce^{\rho}}$; hence, an economic evaluation requires valuation. It follows that a low ATT does not necessarily imply that a program failed a benefit-cost test: a program with a low ATT, even one approaching zero, could nevertheless have a benefit-cost ratio of at least one if $\frac{v}{ce^{\rho}}$ were sufficiently large. Conversely, a program with a high ATT could fail a benefit-cost test if $\frac{v}{ce^{\rho}}$ were small. On its own, an ATT based on avoided deforestation thus provides little information that is useful for economic evaluation even when the economic characteristics of protected sites do not vary.

In practice, the economic characteristics of sites do vary, and this obviates the usefulness of simulated ATTs for ranking programs according to their relative economic returns. Site-level variation in conservation benefits and costs has long been emphasized in the conservation planning literature. The "hot spots" approach developed by conservation biologists (Myers 1988; Myers et al. 2000) emphasized the first term on the right side of (5), by advocating

[5] It is well-known that maximizing a benefit-cost ratio does not necessarily lead to the same ranking of alternative investment projects as maximizing NPV or NFV. Inspection of (7) indicates that the conservation program that maximized the benefit-cost ratio would be the same as the one that maximized NPV or NFV if the two programs were constrained to have the same aggregate cost. Otherwise, the programs could differ. In fact, in the simple context of (7), maximizing the benefit-cost ratio would imply protecting just the single site with the highest deforestation probability, which is hardly a reasonable conservation policy recommendation.

[6] I note in passing that the existence of a feasible program with an average avoided deforestation rate higher than the agency's program does not necessarily imply that the agency behaved irrationally by selecting the latter program. The agency's program might have been the rational choice given the *ex ante* information on expected deforestation rates that was available to the agency when it selected sites for the program.

protection of sites that had both a high risk of deforestation (a high d_{0i}) and high conservation value (a high v_i). There is much evidence that conservation values vary greatly across forests, for example with regard to carbon sequestration (Asner et al. 2010), watershed services (Pattanayak and Kramer 2001; Brauman et al. 2007), and biodiversity habitat (Gibson et al. 2011; Le Saout et al. 2013).[7]

An economic evaluation that considered only the variables in the "hot spots" approach would yield the same result as one based on (5) if costs did not vary across sites ($c_i = c$ for all i), but much evidence indicates that conservation costs are also highly heterogeneous (Naidoo et al. 2006; Polasky 2008). In response to this evidence, the economics literature on conservation planning has emphasized cost-effectiveness as a criterion for evaluating alternative plans (Ando et al. 1998; Polasky et al. 2001, 2008). This approach is the dual to the maximization of NFV: it treats the second term on the right side of (5) as the objective (minimize aggregate costs) and the first term as a constraint (achieve a given level of aggregate benefits).

In sum, economic evaluations of conservation programs require the application of expressions like (5) or (6), because those expressions include economic variables and account for heterogeneity in not only the avoided deforestation rate, which is the basis of the ATT in the typical impact evaluation, but also the benefits and costs of conservation, which typically do not enter into the ATT's calculation. The typical ATT is not useful for applying (5) or (6) because those expressions require site-specific estimates of avoided deforestation rates, not average rates. Heterogeneity rules in the economic evaluation of conservation programs.[8]

3 An Infinite-Horizon Model of Conservation Evaluation with Two Sites

The site-level avoided deforestation rates (d_{0i}), benefits (v_i), and costs (c_i) were arbitrarily given in the two-period model. A possibility not considered there was that the dynamics of deforestation could cause d_{0i} to be positively correlated with v_i and negatively correlated with c_i, which would result in the ATT being positively correlated with the benefit-cost ratio. If so, the ATT could be a useful proxy for the latter, at least for ranking an agency's conservation program relative to other feasible ones.

In this section, I present a model that sheds light on the expected relationship among the deforestation rate and the benefits and costs of protection. The model assumes a process of conversion of forests to agriculture that is socially optimal within the countries where forests are found, with social optimality defined as the maximization of the discounted sum of agricultural land rents. There are two such countries, both autarkic, one with more forest (the "forest-rich country") and one with less forest (the "forest-poor country"). Two other countries ("donor countries") lack forests but receive positive externalities from forests in the forested countries. At the same point in time, one donor country decides to use its foreign aid

[7] See Syrbe and Walz (2012) for a general discussion of spatial heterogeneity in the values of services provided by forests and other ecosystems. The most careful empirical analysis of the relationship between land cover, including forest cover, and the spatial distribution of multiple ecosystem services is probably a study by Eigenbrod et al. (2010) for England, which found that land cover (such as area forested) was a poor proxy for the value of services provided.

[8] Costello and Polasky (2004, Table 1) provide a simple example that nicely illustrates this point. Their example includes three sites, which offer different benefits (numbers of species present) and face different conversion probabilities. Protection costs are uniform across the sites. The authors demonstrate that the dynamically optimal decision is to select the site that has the intermediate conversion probability, not the highest probability.

to protect a marginal area of forest in the forest-rich country, while the other uses its aid to protect a marginal area in the forest-poor country. Can an ATT based on avoided deforestation be used to determine which country made the better aid decision?

3.1 Deforestation Dynamics

The dynamics of deforestation are identical in the forested countries. The countries differ only in that they are at different points along the same deforestation path. Historical accident is responsible for this difference; both countries began with the same amount of forest, but agricultural conversion began earlier in one than the other. My model of deforestation dynamics is identical the classic Eisner–Strotz investment model (Eisner and Strotz 1963). I use the version of this model presented in Chiang (1992, pp. 106–110), which corrects a minor error in the original paper. To facilitate comparison, I use the same notation as Chiang does, with two inconsequential differences: I use dots instead of primes to denote time derivatives, and I add a variable L to denote the constant total land area of each forested country. Land area is the same in the two countries and is allocated completely between agriculture and forests.

At time t, $K(t)$ hectares in a given forested country are in agriculture, with the remaining $L - K(t)$ hectares in forest. Agriculture generates profits (land rents) of $\pi(t) = \alpha K(t) - \beta K(t)^2$ in period t, with $\alpha, \beta > 0$. This convex relationship could reflect, among other things, a diminishing supply of ecosystem services from forests to agriculture as forest area declines; such production externalities are the only benefits of forests to the forested country.[9] Land cannot be converted from agriculture back into forest: $\dot{K}(t) \geq 0$. So, deforestation is possible, but reforestation is not. Converting forest to agriculture costs $C(t) = a\dot{K}(t)^2 + b\dot{K}(t)$ in period t, with $a, b > 0$. Because both parameters are positive, the marginal cost of conversion is positive and increasing in the current area deforested: $\partial C(t)/\partial \dot{K}(t) = 2a\dot{K}(t) + b > 0$, and $\partial^2 C(t)/\partial \dot{K}(t)^2 = 2a > 0$.

In the absence of conservation aid by a donor country, a forested country will maximize the present value of agricultural profits net of conversion costs,

$$\Pi = \int_0^{\infty} \left(\pi\left(K(t)\right) - C\left(\dot{K}(t)\right)\right) e^{-\rho t} dt, \tag{8}$$

subject to $K(0) = K_0$, where K_0 is given. Chiang shows that the optimal area of agricultural land (capital, in his model) rises over time at a diminishing rate, $\dot{K}(t) > 0$ and $\ddot{K}(t) < 0$: i.e., the area deforested declines over time. Early in the country's development, a large area is deforested each period; later in its development, a small area. Chiang also shows that agricultural land converges to a maximum area of $\overline{K} = (\alpha - b\rho)/2\beta$. I assume that $\overline{K} < L$ (an interior solution), which conforms to Chiang's assumption that $\overline{K}$ is not bounded from above.

[9] A more realistic model would include harvesting of roundwood (fuelwood or timber) in the country's forests, as an economic activity that generates a valuable extractive resource but degrades the forests' conservation value. The model in this section ignores degradation resulting from wood harvesting because nearly all impact evaluations of forest conservation programs have also ignored it. I return to this point in the final section of the paper.

3.2 Economic Interpretation of the ATT

Assuming that protection is fully effective, the ATT in this model corresponds to the probability that a marginal area of forest would be deforested if it were not protected by conservation aid.[10] This probability is, as in Sect. 2, the deforestation rate,[11]

$$\frac{\dot{K}(t)}{L - K(t)}. \tag{9}$$

From above, the area deforested in a given period is positive but diminishing over time. This implies that both the numerator and the denominator of (9) are larger in the forest-rich country, which is at an earlier point on the deforestation path, than in the forest-poor country, which is at a later point. The ranking of the deforestation rates in the two countries would thus appear to be ambiguous. It can be shown, however, that the numerator of (9) declines more rapidly than the denominator (see the appendix), which implies that the deforestation rate is higher in the forest-rich country. If the ATT were used to evaluate the donor countries' aid decisions, one would conclude that the donor that protected a marginal area in the forest-rich country earned a higher return on its aid than the donor that protected a marginal area in the forest-poor country.

Consideration of economic factors undermines this conclusion. A forested country would agree to refrain from converting a marginal area of forest to agriculture only if aid from a donor country were at least as large as the forgone agricultural value of that area. This opportunity cost is equivalent to marginal conversion cost, $\partial C(t)/\partial \dot{K}(t)$.[12] From earlier, marginal conversion cost is given by $2a\dot{K}(t) + b$, which implies that the change in marginal conversion cost over time is given by $2a\ddot{K}(t)$. Also from earlier, $a > 0$ and $\ddot{K}(t) < 0$, and so this expression is negative: marginal conversion cost declines over time. The cost to a donor country of protecting a marginal area of forest is thus higher in the forest-rich country, which makes the forest-rich country less attractive for conservation aid than the forest-poor country. The effect of conservation cost on the economics of conservation aid thus runs in the opposite direction as the effect of the ATT.

I have made no assumptions so far about the relationship between the forest-related benefits received by a donor country and the area of forest in a forested country. In principle, this relationship could have a variety of forms. A reasonable default assumption is that benefits are increasing in forest area but at a diminishing rate (i.e., a convex benefit function). Under this assumption, the marginal benefit of protection is lower in the forest-rich country, which again makes the forest-rich country less attractive for conservation aid than the forest-poor country. This further undermines the economic validity of using the ATT to evaluate the aid decisions.[13]

[10] The implicit assumption here is that a donor country randomly selects the area to be protected in a forested country, which is the only possible selection procedure in this aspatial model.

[11] Because the donor country selects a single area for protection, unlike in Sect. 2 the deforestation rate is not averaged across multiple protected areas. Put another way, the deforestation rate given by (9) is the average across the landscape within which the donor country randomly selects a single area to protect.

[12] The current-value Hamiltonian for land conversion in the country is $\pi(K(t)) - C(\dot{K}(t)) + \lambda(t)\dot{K}(t)$, where $\lambda(t)$ is the costate variable. The costate variable gives the present value of current and future agricultural profits from a marginal unit of agricultural land. The first-order condition with respect to the control variable $\dot{K}(t)$ is $\lambda(t) = \partial C(\dot{K}(t))/\partial \dot{K}(t)$. Hence, the marginal value of agricultural land equals the marginal cost of conversion: forest is converted to agriculture up to the point where the long-run marginal benefit of conversion equals the long-run marginal cost.

[13] The marginal benefit described here is a current value, received only in the period when the donor country makes its protection decision. In contrast, the marginal cost of protection described in the previous paragraph

In sum, consideration of both the marginal costs and the marginal benefits of protection indicates that an ATT that is defined in terms of avoided deforestation overstates the case for protecting forests in the forest-rich country instead of the forest-poor country. Determining how much the ATT overstates it would require making explicit assumptions about the relative magnitudes of the parameters a, b, α, β, and ρ, along with explicit assumptions about the points that the forested countries are at on the deforestation path. But that's my point: one cannot assume that an ATT defined in terms of avoided deforestation is sufficient for determining whether economically efficient protection decisions have been made.[14] As in Sect. 2, one would need to conduct additional analysis to determine which donor country obtained a higher net value from its aid decision, and the analysis would need to include information on costs and benefits, not only information on deforestation rates used to construct the ATT.

4 Clinical Trials: A Misleading Analogy for Conservation Evaluation

The impact evaluation literature sometimes portrays clinical trials as the "gold standard" for causal inference (Deaton 2010). This view derives from the random assignment of subjects to treatment and control groups, which eliminates selection bias [the second term in (3)], at least in double-blind trials. The elimination of selection bias through random assignment is indeed a powerful principle. Despite this, clinical trials provide a misleading analogy for the evaluation of conservation programs. A lack of recognition of this point perhaps helps explain the emphasis on physical outcome measures, such as avoided deforestation, in applications of impact evaluation methods to conservation programs.

The key issue, as in Sects. 2 and 3, is heterogeneity of costs and benefits. The cost of administering a medical treatment in a clinical trial is the same across subjects: all subjects in the treatment group receive the same dose or procedure, and all subjects in the control group receive the same placebo. The effect of a treatment on health outcomes might differ across subjects—for example, a given treatment might reduce mortality more in men than in women—and investigation of these differences might even be built into the design of the trial. Clinical trials do not value effects differently across subjects in clinical trials, however, say by assigning a higher value of statistical life to men than to women. Instead, benefits are defined by some purely physical measure of reduced morbidity or mortality, without any weighting by subject-specific economic values. Heterogeneity of costs and benefits across subjects thus does not enter into the evaluation of clinical trials, and this necessarily results in a focus on outcomes expressed in noneconomic terms.

Footnote 13 continued
is a long-run value that is equivalent to the present value of current and future marginal agricultural profits. An apples-to-apples comparison requires expressing the marginal benefit to the donor as a present value, too. All that needs to be shown here, however, is that the ranking of long-run marginal benefits is the same as the ranking of current marginal benefits, and doing this does not require formal analysis. The present value of future benefits from a marginal forest area is higher in the forest-poor country than in the forest-rich country because economic conditions in the forest-poor country at time t are equivalent to those in the forest-rich country at a later time $t + x$. So, period-by-period into the future, the current marginal benefit of forest protection cannot be lower in the forest-poor country than in the forest-rich country, and so neither can the long-run marginal benefit calculated at a common point (i.e., the same time period) in the two countries.

[14] Similarly, I might have obtained different results if I had developed a more complicated model than the simple one in this section, but that would just reinforce this point: the relationship between the ATT and economically rational conservation decisions is context-dependent. The ATT does not necessarily provide a useful proxy for the latter.

This focus may well be appropriate in the context of clinical trials, but the context for conservation evaluation could not be more different. Both costs and benefits can vary across "subjects," even for subjects who receive the same "treatment" or experience the same "outcome." Costs and benefits, and not just treatment effectiveness in physical terms, must therefore be considered when evaluating conservation programs from an economic standpoint.

5 Discussion

Using two simple but complementary models, I have demonstrated that impact evaluations of forest conservation programs that measure outcomes by avoided deforestation can be related to an economic measure of program performance only under very restrictive conditions. My findings provide conceptual support for the caution expressed by Joppa and Pfaff (2010, p. 5) about the policy implications of results from their matching study on global protected areas (PAs):

> Such results do not imply criticism of existing PAs' locations or management. … For instance, a PA targeting a region of dense and highly valued biodiversity might well be worthwhile even far from roads and cities, as blocking a low threat (i.e. low impact) could provide benefits above all costs. Further, targeting high threats will sometimes be discouraged by correlated high costs.

My findings also underscore the importance of related points made by several recent studies in the impact evaluation literature on forest conservation programs. For example, Ferraro et al. (2013, p. 6) state that "there is a glaring lack of cost data in the literature on environmental impact evaluations."[15] Ferraro et al. (2012, p. 39) complement this point by drawing attention to benefits: "Future studies must more tightly integrate policy and research by combining nonmarket valuation and program evaluation." Miteva et al. (2012, p. 86) appropriately call for a "*Conservation Evaluation 2.0*—that not only uses theory of change to better characterize the mechanisms that trigger heterogeneous impacts varying with context, but also then conducts evaluations of different instruments in different contexts to permit cost–benefit comparisons of conservation instruments."

A good first step toward this new version of impact evaluation would be to develop outcome measures that relate more directly to conservation benefits or costs. Consider the example of the UN Collaborative Program on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries (UN-REDD; http://www.un-redd.org/). An impact evaluation conducted in the style of most of those to date would estimate the program's impact on reduced deforestation (emphasis on the first "D"). The program's objective is to reduce greenhouse gas emissions, however, not deforestation, and emissions are not uniform across deforested sites. A more suitable outcome measure would be defined in terms of reduced emissions (emphasis on the "E"), which could be developed using a map of forest carbon (e.g., Saatchi et al. 2011) instead of forest cover. Given that the economic impact of an additional molecule of carbon dioxide in the atmosphere is independent of where the molecule was emitted, such a measure would be perfectly correlated with the program's climate-related benefits. An impact evaluation similar to this has in fact been conducted on "blue carbon" in Indonesia (Miteva et al. 2014). While not a complete economic analysis,

15 A study on community-based forest management by Somanathan et al. (2009) provides a rare example of combining impact evaluation results with information on conservation costs.

which would require evaluation of emission reduction costs too, this type of impact evaluation is economically more relevant than ones that measure avoided deforestation.[16]

Developing outcome measures related to avoided degradation (the second "D" in "REDD") and not just avoided deforestation would enhance the policy relevance of impact evaluations. Somanathan et al. (2009) is one of the few impact evaluations on forest degradation. Impact evaluations have evidently emphasized deforestation more than degradation because remote sensing, which is the standard data source for evaluations, is considered to be more reliable for deforestation than for degradation (Blackman 2013, footnote 10). The neglect of degradation has two important policy implications. One is that evaluations are stacking the deck against forest conservation programs, which are not always primarily motivated by a concern about deforestation. Malaysia illustrates this point well. A mid-1960s land-use planning exercise in the more developed part of the country, Peninsular Malaysia, allocated land at lower elevations and on gentler slopes to agriculture and human settlement and land at higher elevations and on steeper slopes to forests (Wong 1971). The Malaysian government has adhered to this plan reasonably closely, with the consequence that nearly all of the lowland rainforests in the peninsula have been converted to agriculture (Vincent and Rozali 2005). Forests remain mainly in upland regions, and in those regions the government's 1978 National Forestry Policy designates a substantial forest area as Protection Forests. The meaning of "protection" in the Policy was not so much protection against deforestation, in the sense of conversion to agriculture or another nonforest land use (which the land-use plan already disallowed), but rather protection against logging, which in the tropics typically degrades forests but does not result in deforestation (Geist and Lambin 2010). If an impact evaluation that used avoided deforestation as the outcome measure were conducted for Peninsular Malaysia's Protection Forests, it would likely generate a misleading, negligible ATT, because negligible deforestation would have occurred in the absence of protection. Proper evaluation of this protection policy would require an outcome measure related to the presence or intensity of logging, to determine the policy's impact on avoided degradation.

The second implication of the neglect of forest degradation is that impact evaluations are ignoring a conservation issue that might be globally more important than generic deforestation. This issue is the loss of primary forests (Mackey et al. 2014): "forests of native species in which there are no clearly visible signs of past or present human activity" (FAO 2010, p. 11). These unlogged, virgin forests are globally significant repositories of biodiversity (Gibson et al. 2011) and carbon (Asner et al. 2010). Their area is declining at an annual percentage rate that is nearly triple the annual global deforestation rate (FAO 2010, Tables 2.4 and 3.3). Logging is the main cause of their decline (FAO 2010, p. 27), not agricultural conversion as in the case of deforestation (although conversion can follow logging). Forests typically remain after logging, but their high conservation value for biodiversity, carbon, and other ecosystem services has been degraded. There is an urgent need to develop datasets that would enable impact evaluations to evaluate programs that aim at protecting these valuable forests.

Heterogeneity matters for the evaluation of programs aimed at reducing degradation as much as it matters for programs aimed at reducing degradation.[17] In the case of avoided

[16] Similarly, spatial data on the ranges of threatened species from IUCN (http://www.iucn.org/) or NatureServe (http://www.natureserve.org/) could be used to develop outcome measures related to reduced species loss. Species loss correlates less closely with biodiversity-related benefits than reduced carbon emissions correlate with climate-related benefits, however, because different species or assemblages of species do not necessarily have identical values.

[17] I am grateful to an anonymous reviewer for very helpful suggestions regarding the issues discussed in this paragraph.

deforestation, impact evaluations of conservation programs are paying increasing attention to spatial heterogeneity in programs' impacts, by estimating conditional treatment effects that vary with site characteristics instead of just a single program-wide ATT (Pfaff et al. 2009; Sims 2010; Ferraro and Hanauer 2011; Ferraro et al. 2011; Alix-Garcia et al. 2012; Robalino and Pfaff 2012, 2013).[18] This has been done in different ways, for example by including interaction terms in linear regression models (Pfaff et al. 2009); combining matching-based preprocessing of data with partial linear modeling (Ferraro et al. 2011); and combining imputation methods adjusted by regression with a variance estimator based on matching (Ferraro and Hanauer 2011). This work faces statistical and data-related challenges, but it represents important progress toward generating the information needed for economic evaluations. Although an economic evaluation would ideally be based on site-specific information [see (6)], a partially disaggregated evaluation based on average benefits, costs, and avoided deforestation rates for classes of relatively similar sites could still be very useful.

As Ferraro et al. (2013, p. 6) suggest, a big challenge facing impact evaluation is to be relevant for forward-looking conservation planning, not just backward-looking evaluations of decisions that have already been made. In keeping with the impact evaluation literature, the models in Sects. 2 and 3 focused on the latter, and they assumed that conservation benefits and costs and avoided deforestation rates were observable or at least capable of being estimated. For conservation planning, these variables need to be predicted, so that planners can select the set of sites that maximizes expected net present value. Impact evaluation research could play a role here by investigating the accuracy of disaggregated ATTs as predictors of future avoided deforestation rates, using standard techniques such as out-of-sample forecast errors (Kennedy 2008, Ch. 20). A problem, however, is that many impact evaluations analyze the effects of conservation programs on avoided deforestation during time intervals as short as 3–5 years (Blackman 2013, Tables 1 and 2).[19] Given the great economic, social, and political changes that occur as countries develop, the analysis of such short intervals surely does not provide a reliable basis for predicting deforestation rates very far into the future. This argues for extending the time intervals analyzed by impact evaluations from a few years to several decades or more. Augmenting satellite data with information on forests in earlier periods drawn from historical sources, such as colonial archives and military maps,[20] would be one way to do this.

6 Appendix

Differentiating (9) with respect to time yields:

$$\frac{\ddot{K}(t)\,(L - K(t)) + \dot{K}(t)^2}{(L - K(t))^2} \tag{10}$$

[18] Ferraro, Hanauer, Miteva et al. (2013, p. 2) state, however, that "most [impact evaluations] treat 'protection' as if it were homogeneous."

[19] It follows that the ATTs estimated by these studies will necessarily be small in most cases: annual deforestation rates are only a few percentage points even in countries with "rapid" deforestation, and so nearly all of the land that was forested at the start of the interval will still be forested at the end, regardless of protection status. It also follows that a conservation program with an ATT = 0 during such a short interval could nevertheless reflect smart economic decisions: protection could generate a positive net return in the long run, after accounting for benefits and costs that occur beyond the interval evaluated.

[20] For example, see the University of Texas at Austin's Perry-Castañeda Library Map Collection (http://www.lib.utexas.edu/maps/ams/).

Chiang (1992) demonstrates that the following relationship holds along the optimal path,

$$K(t) = \overline{K} - \left(\overline{K} - K(0)\right) e^{r_2 t},$$

where r_2 is a negative constant, $\frac{1}{2}\left(\rho - \sqrt{\rho^2 + \frac{4\beta}{a}}\right)$. This simplifies to $K(t) = \overline{K}\left(1 - e^{r_2 t}\right)$ if $K(0) = 0$ (all land is initially in forest). It follows that $\dot{K}(t) = -r_2 e^{r_2 t}\overline{K}$ and $\ddot{K}(t) = -r_2^2 e^{r_2 t}\overline{K}$.

Substituting these expressions into (10) and simplifying, we obtain

$$\frac{r_2 \dot{K}(t)\left(L - \overline{K}\right)}{(L - K(t))^2}.$$

This expression is negative because r_2 is negative and all the other terms are positive. Hence, the deforestation rate declines over time.

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