Endangered Species Conservation on Private Land: Assessing the Effectiveness of Habitat Conservation Plans

Christian Langpap, Department of Agricultural and Resource Economics, Oregon State University*.

Joe Kerkvliet, Department of Economics, Oregon State University.

***** Corresponding author. Department of Agricultural and Resource Economics. Ballard 240 E. Oregon State University. Corvallis, OR 97331.
Ph. 541 737 1473, Fax 541 737 2563, Email: christian.langpap@oregonstate.edu

Abstract

Habitat Conservation Plans (HCPs) have become a key instrument for implementation of the Endangered Species Act (ESA) on private land. However, there is no systematic analysis of their effectiveness in promoting endangered species recovery. This paper is the first to provide a comprehensive analysis of the impact of HCPs on species recovery status. We find evidence that HCPs have a significant positive impact on species recovery. Our results also suggest that the recovery benefits are larger when species have relatively larger plans. However, we fail to find strong evidence that multispecies plans covering more species are more effective than plans which include fewer species.

Key Words: Endangered Species Act, Habitat Conservation Plans, Private Land, Conservation, Fish and Wildlife Service, Recovery.

1. Introduction

Implementation of the Endangered Species Act (ESA, the Act) on private land has been fraught with controversy. A main reason is the perceived conflict between species preservation and land development created by Section 9 of the ESA, which prohibits the "taking" of an endangered or threatened species. The prohibition includes acts that directly harm wildlife or significantly modify or degrade a species' habitat. This perceived conflict generates perverse incentives for private landowners, who may preemptively modify or destroy habitat, or withhold information from government agencies charged with species conservation (Polasky and Doremus 1998; Innes 2000; Michael and Lueck 2003; List et al. 2006). The tradeoff between habitat preservation and competing land uses has been highlighted by high-profile species-versus-development "train wrecks", such as the spotted owl in the Pacific Northwest and the gnatcatcher in southern California. These have made the Act the target of fierce political attacks[1].

To increase flexibility in the implementation of the law, Congress amended the ESA in 1982. Under Section 10(a), the U.S. Fish and Wildlife Service (FWS) and the National Marine and Fisheries Service (NMFS) may issue an incidental take permit (ITP) when a landowner has prepared a satisfactory habitat conservation plan (HCP, plan). In 1994 the Clinton administration developed the "no surprises" rule, which guarantees HCP participants that their obligations will not change even if future circumstances change. By allowing a take to occur legally if the resulting damage is minimized and adequately mitigated, and by providing regulatory assurances to landowners, HCPs seek to balance species' habitat needs with the property rights of private landowners (Moser 2000).

[1] During the early 1990s there was growing support, bolstered by the Republican takeover of Congress in 1994 and the growing property rights movement, for repealing the ESA altogether.

By attempting to address both conservation needs and property rights, HCPs have sparked heated controversy. The FWS claims HCPs have achieved the desired balance (FWS 1996), and the approach is supported by some in the environmental community (Bean and Wilcove 1997). However, many scientists and environmental groups have expressed skepticism and concern about the effectiveness of the HCP process in promoting the ESA's goals (Beatley 1994; Shilling 1997; Honnold et al. 1997; Hood 1998; Watchman et al. 2001).

The HCP process has been in place for over twenty five years, and has become a key vehicle for implementation of the ESA on private land. Whether reliance on HCPs is prudent policy depends on their effectiveness in promoting species recovery and preventing extinctions. However, no systematic assessments of HCP effectiveness have been performed to date. Detailed case studies of individual HCPs have been performed (Beatley 1994; Hood 1998), and thorough analyses of the science used in HCPs (Kareiva et al. 1998; Harding et al. 2001) and the conservation potential of multi-species plans (Rahn et al. 2006) have been conducted, but there are no rigorous empirical examinations of the efficacy of HCPs. In this paper we take an initial step towards filling this gap by statistically evaluating the impacts of HCPs on species recovery[2].

We combine data on HCPs, species recovery, and species' characteristics and use econometric methods to evaluate the treatment effects of HCPs on species' recovery status. We also explore the effects of characteristics of HCPs, such as size and number of additional species included in a plan. Our results indicate that HCPs are effective in promoting the ESA's goals: species that have an HCP are less likely to become extinct or decline and more likely to be stable

[2] This type of analysis may also be relevant for contexts beyond endangered species protection, such as land use or water quality, where HCP-like policies could be considered to address conflicts between environmental quality and economic activity. It could also be relevant for other countries addressing species conservation on private land, such as Canada, Australia, and the European Union.

or improving. We also find that plans covering larger areas have bigger positive impacts, but the results for the effects of plans that include more species are inconclusive.

In the next section we provide additional background on HCPs and review the literature. In section 3 we describe the data used, and in section 4 we discuss the empirical models. In section 5 we present our main results and sensitivity analysis. In section 6 we discuss the effects of HCP characteristics. Section 7 contains discussion and a conclusion.

2. Background

2.1 Habitat Conservation Plans

The HCP program has its origins in an early-1980s planning process for San Bruno Mountain, near San Francisco (California). Development on the mountain was restricted by the ESA due to its potential effect on the endangered Mission Blue butterfly and other endemic species. Developers, the FWS, environmental activists, and other stakeholders cooperated in preparing a management plan that allowed some take of butterflies, but also set aside a large area as protected habitat. The plan was preceded by biological studies that identified adequate habitat and included long-term habitat management stipulations as well as a steady funding source. This planning effort was hailed as a model to address conservation-versus-development conflicts through negotiation rather than litigation, and prompted the 1982 amendment of the Act. The first ITP was issued for San Bruno Mountain in 1983 (Beatley 1994).

Implementation of the HCP program began slowly. Only 14 HCPs were approved between 1983 and 1992. However, prompted by the “no surprises” policy, 225 plans were approved between 1994 and 1997. The program has continued growing, and by early 2012 there were over six hundred seventy five HCPs and nearly eight hundred ITPs covering over 40 million acres and hundreds of species (FWS 2012). Most HCPs have dealt with commercial development or forest management. Although there are plans across the U.S., most are in areas with both large numbers of endangered species and high demand for land development: the southwest (mainly Texas), the southeast (mainly Florida and Alabama), and California.

The ESA stipulates minimum requirements for HCP approval. A plan must: i) specify the likely impact of the proposed taking, the measures that will be taken to mitigate that impact, and monitoring procedures to assess the effectiveness of these measures; ii) identify reliable funding[3]; iii) list the alternatives to the proposed taking and explain why they were not chosen; iv) lay out procedures to deal with unforeseen circumstances, as well as additional measures that may be required by the FWS or NMFS; and v) demonstrate that the proposed activities will not appreciably reduce the likelihood of the survival and recovery of the species.

Some common mechanisms for mitigating the impacts of a taking include limiting the geographical extent or timing of harmful activities, acquisition of existing habitat, protection of habitat through conservation easements or similar instruments, enhancement or restoration of damaged or former habitat, management of habitat to achieve characteristics required by the species, and creation of new habitat. Alternative forms of mitigation have included allowing permitees to fund recovery plan efforts or to contribute or participate in habitat mitigation banks. Funding requirements are met with mitigation fees on new development in or near habitat areas, letters of credit, escrow accounts, or performance bonds, as well as federal, state, and local government funding (Beatley 1994; Ruhl 1999).

HCPs may further ESA goals for several reasons: (i) HCPs may add flexibility to ESA implementation, promoting negotiation instead of confrontation and litigation. Mitigating some of the adversarial nature of ESA implementation may reduce the incidence of pre-emptive

---

[3] Failure at any time during the duration of the plan to provide required funds is reason for revocation of the ITP (Ruhl 1999).

habitat destruction. (ii) HCPs have set aside considerable amounts of habitat, often in a coordinated, interlinked manner (Moser 2000). (iii) HCPs may generate funding and political support for preservation of habitat that may otherwise not exist given the political and practical enforcement difficulties of implementing the Act on private land. Furthermore, in negotiating a plan the FWS will often elicit more conservation effort than the landowner is legally required to provide (Moser 2000). (iv) HCPs direct additional resources to the covered species, as funds are expended to assess the occurrence of species, the quality of habitat, the quantity of take expected, mitigation activities, and periodic monitoring[4]. (v) HCPs help inform landowners of their impacts on wildlife. Small landowners have heterogeneous management objectives that often include wildlife, and providing information to these landowners about the impacts of their actions can induce voluntary changes (Lehmkuhl et al. 2004). (vi) HCPs provide public and private landowners with the opportunity to combine objectives and take advantage of complementarities. For many species, objectives can be achieved at the landscape level that cannot be achieved on small acreages (Lehmkuhl et al. 2004). Finally, (vii) HCPs help provide strategies and organizational structures for conservation efforts that are absent without a plan.

Nevertheless, critics argue that there are significant flaws in the way the HCP process is implemented in practice. One concern is the context of biological uncertainty in which HCP decisions are usually made. Key uncertainties include the extent of necessary habitat, the appropriate size, number, and configuration of preserves, baseline populations, and species ecology (Beatley 1994; Shilling 1997; Watchman et al. 2001). For example, Harding et al. (2001) find "a striking lack of information on the basic biology of many species" for which ITPs have been granted. Hence, these critics argue, the planning process is largely conducted on the

[4] The evidence suggests that more expenditures have a positive impact on species recovery (Kerkvliet and Langpap 2007; Ferraro et al. 2007; Langpap and Kerkvliet 2010).

basis of short-term considerations of the stakeholders rather than on the basis of biologically relevant criteria (Bingham and Noon 1997; Hood 1998; Kostyack 1998).

Shortcomings in HCP preparation may translate into deficiencies in implementation and results. The quantity and quality of habitat conservation and species protection requirements contained in plans may be inadequate, mitigation guidelines can be arbitrary and lack empirical foundation, and expected conservation benefits may be vague or meaningless[5] (Bingham and Noon 1997; Kostyack 1998). Furthermore, critics argue that these shortcomings are compounded by the inadequacy of long-term funding, monitoring arrangements, and adaptive management strategies, as well as by limitations imposed by the no-surprises policy (Hood 1998).

Finally, a potentially critical limitation is that the FWS requires that the taking authorized by an HCP not "appreciably" reduce the likelihood of survival of the affected species, but it does not require that the plan contribute to recovery. This lack of a clear legal mandate that HCPs do not undermine the ESA's recovery goal could mean that, even after all mitigation efforts are implemented, the net effect of an HCP may be to permanently reduce the population of a species, either by direct take or by modification of habitat (Ruhl 1999). The FWS has insisted that recovery is an important consideration in HCPs, and that contribution to recovery is often an integral product of a plan. The HCP Handbook (FWS 1996) specifies that, whenever feasible, the FWS should encourage plans that result in net benefits to species, but the service has rejected arguments that recovery should be a requirement for HCP approval. Critics argue that this places a disproportionate amount of risk on the species (Hood 1998) and ultimately implies that HCPs do not protect endangered species (Shilling 1997).

---

[5] For instance, Beatley (1994) questions the adequacy of the Coachella Valley HCP, which allowed an 85%-90% reduction in the existing habitat of the fringe-toed lizard, and Honnold et al. (1997) point out that sometimes HCPs specifically assume species will not persist on lands covered by the plan.

Perhaps not surprisingly given these criticisms, there is much uncertainty and skepticism about the effectiveness of HCPs in promoting the ESA's goals. For instance, Shilling (1997) argues that the emphasis placed by the FWS on habitat conservation planning, combined with the weak recovery goals in these plans, "puts virtually any listed species in jeopardy of extinction." Brussard et al. (1997) maintain that "HCPs have the potential to become habitat giveaways that contribute to, rather than alleviate, threats to listed species and their habitats." Kostyack (1998) states that HCPs have created "new and unprecedented risks for imperiled species" and that, without appropriate safeguards, they "will likely do more harm than good."

Research on this topic has focused on case studies of specific plans and assessments of the scientific adequacy of HCPs, but there are no systematic empirical evaluations of the overall effectiveness of these plans in promoting species recovery. Indeed, Honnold et al. (1997) argue that perhaps the most troubling aspect of the growing number of HCPs is the lack of analysis of existing plans to establish whether they are protecting biodiversity. Watchman et al. (2001) assert that HCPs remain controversial mostly because of the uncertainty about their ultimate impacts on recovery of species. In this paper we address this uncertainty by providing the first systematic empirical analysis of the effectiveness of HCPs in promoting species recovery.

2.2 Literature

This paper contributes to three distinct but closely related strands of literature. First, the economics literature has focused on understanding the perverse incentives created by the take prohibition in section 9 of the ESA (Polasky and Doremus 1998; Innes 2000; Michael and Lueck 2003; List et al. 2006), but has not focused specifically on HCPs. Innes et al. (1998) discuss HCPs in the broader context of incentives for conservation on private land. Langpap and Wu (2004) examine the level of conservation elicited by agreements with regulatory assurances such as the no surprises policy. However, these theoretical papers do not provide empirical evidence of the effectiveness of HCPs. Langpap (2006) presents empirical evidence that assurances-based incentives can promote conservation on private land, but does not specifically examine HCPs.

Second, the conservation biology and policy literatures have paid considerable attention to HCPs. Beatley (1994) and Hood (1998) conduct detailed reviews of several plans, but these assessments take a case-study approach and lack rigorous empirical analysis. Kareiva et al. (1998) and Harding et al. (2001) conduct a more systematic analysis of a broad spectrum of plans. They examine how well the available scientific data were used during HCP development, whether the conclusions in the plans are supported by the scientific literature, and whether the data proposed for monitoring are relevant. They conclude that overall the available scientific information was adequately used, but for many species more in-depth analysis was required to justify issue of an ITP. Rahn et al. (2006) focus on the scientific quality of multi-species HCPs and on the conservation potential of these plans for the individual species they cover. They find that many plans are excessively broad, and conclude that multi-species planning will not always guarantee effective conservation. These papers conduct more systematic assessments than those provided by case studies, but they do not focus on the effects of HCPs on species recovery.

Finally, four papers have empirically examined the effectiveness of the ESA. Male and Bean (2005) find that taxa, funding, extinction risk, and recovery potential are correlated with species' recovery. Kerkvliet and Langpap (2007) find evidence that increased spending reduces the probability of a species being classified as extinct or declining. Ferraro et al. (2007) find that ESA listing is detrimental to recovery on average, but species receiving substantial funding tend to improve. Langpap and Kerkvliet (2010) explore alternative criteria for allocating funds to listed species. None of these papers examine the impact of HCPs on species recovery.

By conducting the first rigorous empirical analysis of the effectiveness of HCPs, this paper contributes to the economic literature on incentives for conservation on private land, to the conservation biology and policy literature that analyzes HCPs, and to the economics and policy literature that assesses the efficacy of ESA implementation. Additionally, by examining the effectiveness of a policy to elicit conservation on private land, this paper also informs the broader literature on property rights, land use policy and conservation, and takings (see for example Blume et al. 1984 or Innes 1997).

3. Data

We use species-level data on recovery status, ESA implementation activities including HCP preparation, and species characteristics. We also use information on relevant characteristics of the states in which species' habitat is located. We have panel data for ESA listed vertebrates for the years t = 1990, 1992, 1994, 1996, 1998, 2000, 2002, 2004.[6]

We measure endangered species recovery with the population status reported by the FWS (U.S. FWS 1990a-2004a). This is the only broad-based measure of recovery status available over a time frame long enough to meaningfully discuss species recovery. It has been used in the ecological and economics literatures for similar purposes (Taylor et al. 2005; Male and Bean 2005; Kerkvliet and Langpap 2007; Langpap and Kerkvliet 2010). The FWS status reflects population size and threats. It is determined by field and regional staff based on the best available information from recovery planning and implementation efforts, consultation with other federal and state agencies, and the FWS' permitting program (U.S. FWS 2002a).

The FWS classifies each listed species as Extinct (E), Declining (D), Stable (S), Improving (I), Recovered (R), found only in captivity (C), or Uncertain (U). We use this

[6] Data on recovery status is reported by FWS every two years since 1990, with 2004 the last year available when this analysis was conducted. We limit the analysis to vertebrates because of data availability.

classification to construct a discrete, ordered measure of recovery: $Status_{it}$ = 0 if species *i*'s status at time *t* is E, $Status_{it}$ = 1 if status is D, $Status_{it}$ = 2 if status is S, or $Status_{it}$ = 3 if status is I or R. We exclude status C because only four species are in this class. We set $Status_{it}$ = 3 for species in both I and R because only nine species have R status. We omit status U because it cannot be meaningfully ordered among the others[7].

We obtain data on HCP coverage and key characteristics of plans from the FWS' Conservation Plans and Agreements Database (FWS 2010). The database includes the HCP's starting date, acreage, and the species covered by the plan. The listing of species included in each plan allows us to cross-reference this database to the FWS' Threatened and Endangered Species Database System. This lets us identify, for each listed species, all the HCPs in which it is included in a given year, along with the size of each plan and the total number of additional species included in the plan. For each species, we construct an HCP indicator variable: HCP_{it} = 1 if species *i* is included in at least one plan in year *t*, HCP_{it} = 0 otherwise. We also calculate the total size (acreage) of all the HCPs in which species *i* is included, and the mean number of other species covered by the plans that include species *i*. Summary statistics for *HCP* and other key variables are in table 1. The table shows an increasing trend in the proportion of species that have an HCP, plan acreage per species, and average number of additional species covered by a plan.

To control for ESA implementation, we use cumulative federal and state spending on each species for each year (U.S. FWS 1989-2004b)[8] and indicator variables for final recovery

[7] This raises the possibility of sample-selection bias if factors that determine whether the status of a species is U are correlated with factors that determine recovery. We tested for this possibility by defining a dichotomous recovery variable and estimating a Heckman selection model with selection based on the U status. The results presented here hold, and there is no evidence of selection.

[8] Agencies are required to report annually all "reasonably identifiable expenditures" that can be traced back to specific species. This includes spending for items such as refuges, land acquisition, law enforcement, research, surveys, listing, recovery, and consultation. Opportunity costs such as the value of unsold power, timber, or water are generally not included. We use cumulative spending because some recovery actions are likely to yield cumulative improvements in recovery status.

plan preparation and critical habitat designation (U.S. FWS 1990a-2004a). We also control for species characteristics that could influence recovery. Indicator variables $Mammal_i$, $Bird_i$, $Reptile_i$, and $Amphibian_i$ are set equal to one if species i is in that taxonomic group and zero otherwise (fish is the benchmark). We account for species distinctiveness with the variable $Distinct_i = 1$ if the species is monotypic (the only species in its genus) or belongs to a small genus (2 – 5 species), and $Distinct_i = 0$ otherwise. Longevity and growth rate are highly correlated with body size, so we include $Body\ Length_i$. We construct indicator variables for migratory habits ($Non\text{-}Migratory_i$, $Local\text{-}Migratory_i$, $Distance\text{-}Migratory_i$), phenology ($Diurnal_i$, $Nocturnal_i$), feeding habits ($Carnivore_i$, $Herbivore_i$), and predominant type of habitat ($Terrestrial_i$, $Subterranean_i$, $Estuarine_i$, $Lacustrine_i$, $Palustrine_i$, $Marine_i$, $Riverine_i$), as well as for whether a species has special habitat needs. Data on species characteristics were obtained from the Database on the Economics and Management of Endangered Species (Cash et al. 1998) and NatureServe (2007).

We account for several additional factors that could affect recovery status. We include an indicator variable for whether FWS considers a species' recovery to be in conflict with economic activity. To control for time-dependent factors, such as changes in conservation attitudes and priorities, political factors, or the decision-making processes of state and federal agencies, we include a trend variable. We control for development pressure by including mean population growth and mean land values in the states in a species' range (USDA 1990-2004). Finally, we use the League of Conservation Voters' (LCV) annual ranking of members of Congress on their votes on environmental legislation (LCV 1990-2004) to account for the influence of political factors (Cash 2001; DeShazo and Freeman 2006; Ferraro et al. 2007). We use the mean LCV score for U.S. Congress Interior Subcommittee members from the states in each species' range.

4. Empirical Models

In this section we describe the empirical strategies we use to assess effectiveness of HCPs. The economics literature has taken two distinct approaches to identifying the ESA's effects. One uses ordered discrete dependent variable models to estimate parameters measuring impacts (Kerkvliet and Langpap 2007; Langpap and Kerkvliet 2010). The other relies on treatment evaluation methods, specifically on matching estimators (Ferraro et al. 2007). Each approach has strengths and weaknesses, so we use both to examine HCP effectiveness. This comprehensive estimation strategy allows us to evaluate the robustness of our results. We conduct additional robustness checks for both approaches, which we discuss in section 5.

We focus on listed species and compare recovery outcomes for species with and without HCPs. Hence, the counterfactual for both approaches is ESA protection but no HCPs. We control for other ESA implementation actions, for which all listed species are eligible independently of HCP preparation. In the ordered dependent variable approach, identification comes from the use of instrumental variables and is based on the assumption that exclusion restrictions for the instruments are satisfied. In the treatment evaluation approach, identification is based on the assumption that, given the set of observed covariates, whether a species has an HCP or not is independent of its recovery status.

4.1 Ordered Dependent Variable Model

In this approach, the basic econometric model of species recovery is

$$Status_{it} = \alpha_0 + \alpha_1\, HCP_{it} + \boldsymbol{\alpha}_2 \boldsymbol{X}_{it} + \alpha_3 t + \lambda_i + \varepsilon_{it} \qquad (1)$$

where α_1 is the main parameter of interest, as it measures the difference in recovery status between species that have an HCP and species that do not. $\boldsymbol{X}_{it}$ is a vector containing all control variables, including ESA implementation activities, species characteristics, and other factors that may affect recovery. λ_i is a species-specific effect that controls for unobserved species heterogeneity, t is a time trend that is common to all species, and ε_{it} is an idiosyncratic unobserved error term.

The dependent variable $Status_{it}$ is a discrete, ordered variable. We therefore estimate model (1) using an ordered probit. To accommodate the panel structure of our data, we use panel methods. Given that there is no consistent estimator of the parameters for fixed-effects probit models (Wooldridge 2002; Cameron and Trivedi 2005), we use Chamberlain's conditional random-effects (CRE) ordered probit[9] (Chamberlain 1980; Wooldridge 2002). This model includes persistent species-specific effects, which yield the same intuitive outcome as fixed effects. The CRE model conditions on the sample averages of the regressors of most theoretical concern rather than on the averages of all variables, as fixed effects do. We condition the distribution of these species-specific effects on sample means of the HCP indicator and cumulative spending by including these means as if they were additional explanatory variables.

An important consideration is that whether a species has an HCP, as well as the expenditure on a species, may be simultaneously determined with its recovery status. In this case parameter estimates are biased. To control for this possibility, we take an instrumental variables approach by estimating prediction models for HCP preparation and cumulative spending, obtaining their predicted values, and replacing actual values with predicted values when estimating equation (1).

We use two instruments for HCPs.[10] The first instrument is the number of previously approved plans in a species' lead FWS administrative region. Regional and field offices are

[9] We conduct a Hausman test and cannot reject the null hypothesis of no correlation between the unobserved species-specific effects and the regressors ($p = 0.13$).
[10] We thank an anonymous referee for suggesting these instruments.

responsible for assisting with HCP preparation, as well as for review and approval[11]. HCP preparation involves substantial time and effort on the part of landowners, FWS, and other parties involved (Beatley 1994; Ruhl 1999). Past experience with HCPs will reduce the transactions costs of additional HCPs and increase the experience of FWS staff in facilitating agreement and obtaining approval. Hence, conditions in regional offices which have handled more HCPs in the past will be more conducive for future plans. Since these conditions are related to transactions costs and agency staff expertise, but not on-the-ground recovery methods, we do not expect this variable to have a direct impact on species status[12].

The second instrument for HCPs is an indicator for whether the “no surprises” rule was in effect in period *t*. This policy was announced in 1994, and was the single most important reason for the dramatic increase in the number of HCPs starting the following year (Moser 2000). This is illustrated in Figure 1, which shows the number of HCPs per year and reveals a substantial jump in the number of plans after 1994, with the total number of plans more than doubling between 1994 and 1995 and more than tripling by the end of 1996. Hence, we expect this variable to be highly correlated with the likelihood of an HCP. Given that the “no surprises” ruling is specific to HCP preparation, but does not indicate changes in other efforts to improve species’ status, it should not have a direct impact on species recovery.

To instrument for cumulative spending we rely on previous work establishing that political influence affects federal spending decisions. A commonly used measure of political influence is the LCV score for U.S. Congress Interior Subcommittee members from the states in each species’ range (Cash 2001; DeShazo and Freeman 2006). This score is constructed as the

[11] There are eight administrative regions (Pacific, Southwest, Great Lakes-Big Rivers, Southeast, Northeast, Mountain-Prairie, Alaska, and California-Nevada).

[12] The impact of any regional differences in other aspects of ESA implementation that could impact recovery are removed from the residual of model (1) by including the ESA implementation control variables in $\boldsymbol{X}_{it}$.

percentage of pro-environmental votes by members of Congress on a set of key legislative votes selected by experts from environmental organizations. To the extent that some of these votes are on legislation that could directly impact species recovery, the LCV score in its original form may not satisfy exclusion restrictions in our model, and hence is not an adequate instrument. To address this limitation, we reviewed every LCV Scorecard report between 1990 and 2004 and recalculated the scores for the relevant members of Congress by excluding votes on legislation that could directly impact species recovery. For example, the score for 1995 included votes on reducing the FWS' listing budget and on red wolf recovery. These votes could directly impact our recovery measure, and were therefore excluded. The 1995 score also included votes on issues such as procedural hurdles to new health and environmental protection standards, drinking water, and the Community Right to Know Act, which do not impact our recovery measure and are therefore included. We excluded votes on issues not directly related to the ESA but potentially having an indirect impact on species recovery, such as timber subsidies. This modified LCV score should reflect political pressure affecting spending decisions, but is plausibly exogenous to species recovery. We further discuss the validity of exclusion restrictions in section 5.

We predict HCP_{it} and cumulative spending as a function of these instruments and all other exogenous variables. Parameter estimates appear in table 2, along with an F-test for the joint significance of the instruments and an R^2 statistic of overall fit of the first stage models. The prediction models yield the expected results and perform well, generating confidence in the relevance of the instruments and in the resulting predicted values. The instruments have statistically significant impacts on the endogenous variables and the signs of their estimated coefficients are as expected. Species with habitat in regions with more previous HCPs are more likely to have a plan, and HCPs are more likely after the "no surprises" policy. Species from states whose representatives receive high LCV scores receive more funding[13].

The advantages of the ordered probit approach are that it makes the most of the discrete, ordered nature of the recovery measure and that it allows us to use information for every period and take full advantage of the panel structure of the data. The disadvantages of this approach are the highly parametric nature of the estimator and the inherent distributional assumptions.

4.2 Treatment Evaluation Models

The objective with this approach is to evaluate the effect of a treatment on an outcome of interest when there are observations on individuals exposed, and not exposed, to the treatment. Assignment to treatment is not random, but rather depends on observable covariates. The evaluation is based on a comparison of outcomes for treated and not-treated individuals. Since for any individual we can only observe an outcome when treated or when not treated, but not both, evaluation requires constructing an appropriate counterfactual. A variety of approaches have been developed to do this (see Lee 2005 or Imbens and Wooldridge 2009 for reviews).

The treatments we want to evaluate are HCPs, and the outcome is species recovery. Following Ferraro et al. (2007) we focus on changes in recovery status over time and define the outcome variable $\Delta Status_i$ as the change in $Status_{it}$ between 1998 and 2004. We chose 1998 as the initial year because the number of species with HCPs remains relatively low until 1996, and we expect that an additional period (two years) is necessary before any impacts are perceived in the data. We explore alternative outcome variables in section 5. The treatment variable is defined as $HCP_i = 1$ if species i has at least one HCP by 1998, and $HCP_i = 0$ otherwise.

[13] As an additional robustness test, we estimated the model using regional dummies as instruments instead of the number of previous HCPs in a region. These variables capture regional differences in the HCP approval process. The estimation results are consistent with those presented here. They are available from the authors.

Our objective is to measure the Average Treatment Effect (ATE), which gives the impact of HCPs on change in recovery status for the average listed species[14]. Unbiased estimation of the ATE relies on the conditional independence assumption (CIA). The CIA states that, conditional on the vector of covariates $\boldsymbol{X}$, the outcomes are independent of treatment. In our context this implies that, after controlling for the effect of species characteristics and other variables, whether a species has HCPs does not depend on changes in recovery. This requirement is likely satisfied because there is no idiosyncratic influence from species themselves, and hence there is no self-selection (Ferraro et al. 2007). We discuss sensitivity to violations of the CIA in section 5.

We control for a comprehensive list of species characteristics, ESA implementation variables, and other relevant factors by including all the variables described in section 3. To avoid bias it is necessary to control for pre-treatment covariates (Lee 2005). Hence, we redefine all time-variant control variables to account for timing of the HCP treatment. For example, to control for recovery planning the variable *Final Plan* is redefined as $Final\ Plan_i = 1$ if species i has a final recovery plan completed before species i's first HCP, or by 1998 if the species is not treated (no HCP by that year), and $Final\ Plan_i = 0$ otherwise.

We use three different approaches to estimate ATEs. The simplest approach is to use linear regression methods. The ATE can be estimated as the coefficient $\hat{\beta}_1$ for the treatment variable in the regression

$$\Delta\ Status_i = \beta_0 + \beta_1 HCP_i + \boldsymbol{\beta}_2{}' \boldsymbol{X}_i + \boldsymbol{\beta}_3{}' HCP_i \cdot \left(\boldsymbol{X}_i - \overline{\boldsymbol{X}} \right) + \nu_i \qquad (2)$$

where $\overline{\boldsymbol{X}}$ is the sample mean of the covariates. Alternatively, the ATE can be estimated as the coefficient $\hat{\gamma}_1$ for the treatment variable in the regression

[14] The average treatment on the treated (ATT), which would measure the change in recovery status for a species with at least one HCP if it did not have any HCPs, is not the interesting measure in our case, because it is more relevant to consider the hypothetical gain from HCPs for an average listed species.

$$\Delta\ Status_i = \gamma_0 + \gamma_1 HCP_i + \gamma_2 \hat{p}(\boldsymbol{X}_i) + \gamma_3 HCP_i \cdot \left[\hat{p}(\boldsymbol{X}_i) - \overline{p}\right] + u_i \tag{3}$$

where $\hat{p}(\boldsymbol{X}_i)$ = Est. Prob(HCP_i = 1|$\boldsymbol{X}_i$) is the estimated probability of treatment given the covariates, or propensity score, and $\overline{p}$ is the sample average of $\hat{p}(\boldsymbol{X}_i)$ (Wooldridge 2002; Imbens and Wooldridge 2009). We use the complete set of covariates to estimate the propensity score.

A potential limitation of this approach is that estimated ATEs may be biased if the linear approximation to the regression function is not accurate globally. Linear regression methods tend to be sensitive to specification if the averages of the covariates in the treated and untreated subsamples are very different; specifically, if the normalized difference $\Delta_x = \left(\overline{x}_1 - \overline{x}_0\right) \Big/ \sqrt{S_0^2 + S_1^2}$ for covariate x exceeds 0.25, where $\overline{x}_1$ and $\overline{x}_0$ are sample means for observations with and without treatment, respectively, and S_1^2 and S_0^2 are the corresponding sample variances (Imbens and Rubin 2010). In our case, $\Delta_x < 0.25$ for all but five covariates (*Nocturnal*, *Terrestrial*, *Riverine*, *High Priority*, and *Final Plan*), which we exclude from the model. Our results are robust to including these variables.

Our second approach is to use matching estimators. Matching refers to estimating ATEs by comparing outcomes only for individuals with the same or similar covariate values across the treated and untreated groups. That is, in our case a counterfactual is constructed by selecting species with and without HCPs that are sufficiently similar across the observed covariates $\boldsymbol{X}$, and comparing the change in their recovery status. The criterion for selecting a comparison group is determined by the choice of matching estimator. Following Ferraro et al. (2007), we use four different matching estimators: nearest-neighbor covariate matching with inverse variance matrix weighting, nearest-neighbor covariate matching with Mahalanobis weighting, nearest-neighbor propensity score matching, and Gaussian kernel propensity score matching. Covariate matching is done across the full set of control variables. We use five nearest neighbors with replacement for nearest-neighbor matching, but our results are generally robust to fewer and more neighbors. We use Abadie and Imbens' (2002) bias-corrected matching estimator to address the bias in matching estimators with finite samples when matching is not exact. The standard errors for nearest-neighbor estimators are computed using the heteroskedasticity-robust variance formula proposed in Abadie and Imbens (2006), and the standard errors for the kernel estimator are bootstrapped using 1,000 repetitions. For the propensity score estimators, we impose a common support and conduct balancing tests[15], which indicate that balance is achieved on all covariates.

Our final approach is to use a difference-in-differences (DID) estimator. In the simplest setting for this method, outcomes are observed for individuals in two periods. There are no treated individuals in the first period, and only a subset of all observed individuals are treated in the second period. The remaining individuals are never observed to receive treatment. The average change in outcome over time in the non-treated individuals is subtracted from the change over time in the treated individuals. This double differencing removes bias from comparisons over time in the treated group that could reflect time trends unrelated to the treatment, as well as bias in second period comparisons between the treatment and control groups that could result from permanent differences between those groups.

In our data, only eight out of 176 species have an HCP in 1990, and fifty five out of 213 species have an HCP in 2004. Hence, if we leave out the eight species that have a plan in 1990 we can use this simple DID approach to estimate the average treatment effect of HCPs.

[15] Common support (or overlap) ensures that $0 < p(\boldsymbol{X}_i) < 1$, i.e. that observations with the same covariates have a positive probability of being treated and not treated. Balancing means that observations with the same propensity score have the same distribution of observable characteristics independently of treatment. That is, for a given propensity score exposure to treatment is random and hence treated and control units should on average be observationally identical.

Assuming conditional independence given lagged outcomes, it is possible to estimate the ATE as the coefficient $\hat{\delta}_1$ in the regression

$$\Delta Status_{i,1990\text{-}2004} = \delta_0 + \delta_1 HCP_{i,2004} + \delta_2 Status_{i,1990} + \delta_3 \Delta \boldsymbol{X}_{i,1990\text{-}2004} + \eta_i \qquad (4)$$

where $\Delta Status_{i,1990\text{-}2004}$ is the change in recovery status between 1990 and 2004, $HCP_{i,2004} = 1$ if species i has an HCP by 2004 and $HCP_{i,2004} = 0$ otherwise, $Status_{i,1990}$ is status in 1990, and $\Delta \boldsymbol{X}_{i,1990\text{-}2004}$ are changes in covariates between 1990 and 2004 (Imbens and Wooldridge 2009).

The main advantages of the treatment approaches are the flexibility they provide in constructing counterfactuals and comparing outcomes for species with and without HCPs, and the fact that they do not rely on strong distributional assumptions. A potential disadvantage is that these models do not take full advantage of the panel nature of the data[16].

By using both an ordered dependent variable model and a variety of treatment evaluation models we exploit two distinct approaches that rely on different assumptions for identification and use the available data in different ways. This comprehensive estimation strategy mitigates the weaknesses of each approach while taking advantage of their strengths, thus generating confidence in the resulting estimates.

5. Effects of HCPs

5.1 Results

In this section we present and interpret our estimation results. Table 3 shows the estimated coefficients for the IV conditional random effects ordered probit. The first part of table 4 shows the corresponding marginal effects of the estimated probability of having an HCP on the probabilities that a species is classified as Extinct, Declining, Stable, or Improving. The second part of table 4 shows average treatment effects estimated using the treatment evaluation models.

[16] The DID approach addresses this concern to some extent by considering the change in recovery status over the entire study period, but it still does not use the information available for all periods in the same way that the panel model does.

The estimates presented in table 4 indicate that HCPs have a positive effect on endangered species recovery. The marginal effects show that species with an HCP are less likely to be classified as Extinct or Declining and more likely to be classified as Stable or Improving. For the average species, a one percentage point increase in the probability of having an HCP lowers the probability that it is classified as Extinct by 5.7 percentage points, and the probability that it is Declining by 43.5 percentage points. The probability that it is classified as Stable increases by 16.5 percentage points, and the probability that it is Improving goes up by 32.6 percentage points.[17] Alternatively, the expected recovery score increases by close to one point: the predicted score for the average species without an HCP is 1.55 (between Declining and Stable), whereas with an HCP it is 2.42 (between Stable and Improving).

The estimated average treatment effects in the bottom part of table 4 are all positive and statistically significant, confirming the ordered probit results. The estimated ATEs for the OLS and matching models indicate that, based on the change in recovery score between 1998 and 2004, the recovery status for a species with an HCP would be between 0.39 points and 0.68 points higher than for a species without an HCP. For the DID estimator the estimated ATE indicates that, based on the change in recovery score between 1990 and 2004, the recovery status for a species with an HCP would be 0.297 points higher than for a species without an HCP. Overall, the estimated ATEs suggest that HCPs will increase a species' recovery score between close to one third and two thirds of a point. The average species that does not have an HCP has a recovery score of 1.55, whereas with an HCP the recovery score is between 1.85 and 2.22. Thus, the magnitude of these effects is somewhat smaller, but qualitatively consistent with those derived with the CRE ordered probit approach.

[17] Relative to the mean probability that a species has an HCP (17%), an additional percentage point represents a 6% increase.

5.2 Sensitivity

5.2.1 Sensitivity: Discrete Choice Models – Alternative Specifications

We use a variety of alternative estimation approaches to check the robustness of the results obtained with the IV CRE ordered probit. We estimate an IV RE ordered probit without conditional effects and a linear fixed effects IV model. We also redefine the dependent variable $Status_{it}$ as a dichotomous variable, $Status\ 2_{it} = 1$ if species *i* is classified as Improving (or Recovered) and $Status\ 2_{it} = 0$ otherwise, and use it to estimate a fixed effects ordered logit model (Ferrer-i-Carbonel and Frijters 2004)[18]. The estimated coefficients for the HCP variable, which are presented in table 5, are positive and significant for all estimation approaches, confirming the result of a positive impact on species recovery. The magnitude of the effects of HCPs for the IV random effects ordered probit and linear fixed effects IV models are consistent with those shown in table 4 (the magnitude of the effects corresponding to the fixed effects ordered logit model cannot be compared because it is based on a dichotomous dependent variable).

In addition to using different estimation approaches, we use the IV CRE ordered probit approach with alternative model specifications. We estimate a specification that leaves out cumulative spending given its potential endogeneity, as well as a specification that includes an indicator variable for critical habitat designation, which is potentially endogenous and thus needs to be instrumented. Finally, we also estimate the model using only the regions in which HCPs are most common. The results presented here are robust to these alternative specifications.

[18] This approach relies on species-specific fixed effects, as well as species-specific thresholds. Ferrer-i-Carbonel and Frijters (2004) show that this reformulation allows Chamberlain's method to be used and removes the species-specific effects and thresholds from the likelihood function.

5.2.2 Sensitivity: Ordered Dependent Variable Models - Potential Violations of Exclusion Restrictions

There are no direct tests of the exclusion restrictions underlying our IV approach. Hence, we take three indirect approaches to assess whether these restrictions hold in our model and to evaluate the robustness of the results to potential violations of these restrictions. First, we use the dichotomous variable *Status* 2_{it} to estimate a bivariate probit. The advantage of this approach is that it does not require excluded instruments for identification, and thus allows us to check for sensitivity to potential violations of the exclusion restrictions. The results, which are included in table 5, confirm that the effect of HCPs on recovery is positive and significant.

Second, we use a linear version of the recovery model to obtain two-stage least squares estimates. Although this methodology ignores the discrete nature of the data, it allows us to conduct a Sargan test for overidentifying restrictions (Cameron and Trivedi 2005). The test indicates that we cannot reject the null hypothesis that the overidentifying instruments are uncorrelated with the errors of the recovery model ($p = 0.58$).

Finally, we rely on an approach developed in Ginther (2000), who uses Manski (1994) bounds on the conditional expectation of the outcome variable to examine whether exclusion restriction assumptions are consistent with the data[19]. Specifically, let y_1 and y_0 be the outcome when treated and when not treated, respectively (in our case recovery for species with and without HCPs), let x represent observable covariates, and z represent the exclusion restriction (the instrument). Assessing whether the exclusion restriction assumptions are consistent with the data involves estimating the exclusion restriction bounds on $E(y_0 \mid x)$ and $E(y_1 \mid x)$[20]. The null

[19] We thank an anonymous referee for suggesting the use of Manski bounds.
[20] The bounds are estimated using

$$\sup_z \left[E(y_1 \mid (x,z), HCP=1) P(HCP=1 \mid (x,z)) + K_L P(HCP=0 \mid (x,z)) \right] \le E(y_1 \mid x) \le$$

hypothesis that the exclusion restriction assumptions are consistent with the data can be tested using a bootstrap procedure and calculating confidence intervals for the bounds on these conditional expectations. If the upper bound exceeds the lower bound, the null hypothesis cannot be rejected. To implement this test, we use 500 bootstrap samples to estimate 95% confidence intervals for the upper and lower bounds of $E(y_0 \mid x)$ and $E(y_1 \mid x)$ for exclusion restrictions based on our two instruments: number of previous HCPs in the region and the no surprises indicator. We condition on the species' body length and on cumulative spending.[21] The results of this test are shown in an appendix available on JEEM's online repository of supplemental material (http://www.aere.org/journals/). The 95% confidence intervals for the lower and upper bounds do not overlap and the upper bounds always exceed the lower bounds. Hence, we cannot reject the null hypothesis that the exclusion restriction assumptions are consistent with the data.

To summarize, we find plausible evidence that our instruments adequately satisfy the exclusion restrictions. Furthermore, our key result is robust to an estimation method that does not require excluded instruments for identification. Hence, we expect any remaining bias from imperfectly satisfied exclusion restrictions to be small on average.

5.2.3 Sensitivity: Treatment Effect Models

We also conduct sensitivity checks for the results obtained with the treatment effect models. First, to address a potential concern about the credibility of counterfactuals constructed by matching species from different taxonomic categories, such as a mammal and a fish, we use the nearest-neighbor covariate and propensity score matching estimators and require an exact match

---

$\inf_z \left[E(y_1 \mid (x,z), HCP=1) P(HCP=1 \mid (x,z)) + K_U P(HCP=0 \mid (x,z)) \right]$, and similarly for y_0, where K_L and K_U are the lower and upper bounds on the outcome variable (the recovery measure).

[21] Nonparametric estimation methods have the disadvantage of being limited in the number of possible conditioning variables (Ginther 2000).

on taxonomy (Ferraro et al. 2007). We also require the match to be exact on the 1998 FWS baseline score, as well as on both taxonomy and the baseline score.

Second, we test whether using an alternative outcome variable affects the results. We estimate the treatment models using the recovery status in 2004 as the outcome variable instead of the change in status between 1998 and 2004. The status in 2004 reflects the end-result, up to that point in time, of the recovery process, including HCP preparation. The results are presented in table 6. Average treatment effects of HCPs remain positive and significant when we impose exact matching on taxonomy and baseline recovery score and, with the exception of the kernel matching estimator, when the outcome variable is recovery status in 2004.[22] ATEs also remain positive and significant when using data only from the regions in which HCPs are most common.

Finally, all the results for the treatment effect models are conditional on observed covariates (the CIA assumption). If there are unobservable factors that could affect both the likelihood of HCPs and species recovery, these results would be biased. Although the CIA cannot be tested, it is possible to assess the sensitivity of our results to violations of this assumption. The results of this sensitivity analysis, presented in detail in the online appendix, suggest that our results are robust to reasonable failures of the CIA.

5.2.4 Sensitivity: Alternative Recovery Measure

To address potential concerns about our use of the FWS status as a recovery measure (National Research Council 1995; Ferraro et al 2007), we check the robustness of our main result to using a different measure of species recovery. We estimate an IV ordered probit, as well as the OLS and matching models, using the International Union for Conservation of Nature's Red List status

[22] The nearest-neighbor matching results with the 2004 recovery status outcome are not as robust as the original results to changes in the number of neighbors used for matching.

for 2004 as a dependent variable.[23] Marginal effects and ATEs, which are presented in the online appendix, suggest that our main results are unlikely to be driven by the choice of the FWS score as a measure of recovery.[24] The estimated effects confirm the evidence of a positive impact of HCPs on species recovery. HCPs decrease the probability that species are classified as Critically Endangered or Endangered, and increase the probability that they are classified as Near Threatened or Least Concern. The ATEs are all positive and significant.

6. Characteristics of HCPs

Our analysis so far has considered the general effect of HCPs on species recovery. However, these effects need not be homogeneous, as species can have diverse sets of HCPs with different attributes. In this section, we use the information available on two key HCP characteristics to examine whether these characteristics have an effect on species recovery.

One important HCP characteristic is its size, or the total area it covers. *A priori*, we may expect plans covering larger areas to provide additional conservation benefits. To examine whether larger plans make a greater contribution to species' recovery, we construct a variable that measures the total acreage covered by a species' HCPs. We use this variable as an additional regressor in the CRE ordered probit, and also include squared total size to allow for nonlinear effects.[25] For the treatment models, we define two alternative treatment variables. A species is treated with small HCPs if the maximum acreage of its plans up until 1998 is less than the

[23] The Red List score is also a discrete, ordered ranking of species' status. The categories, and the numerical value we assign them, are: Extinct (0), Critically Endangered (1), Endangered (2), Vulnerable (3), Near Threatened (4), and Least Concern (5). We are not able to use a panel or consider changes in the Red List status because Red List scores have not been constructed consistently over time.

[24] Only a fraction of the species in our data has updated Red List scores for 2004, and hence we lose a considerable number of observations. Thus, these estimates should be interpreted with caution.

[25] Since many species are included in more than one plan, and species that have more plans are also likely to have more total HCP acreage, we also estimated a model with average acreage of a species' HCPs as an alternative measure of size, as well as a model that includes both total acreage covered by a species' HCPs and the species' total number of HCPs. The results are consistent with the ones presented here.

median acreage of all plans up to that year. A species is treated with large HCPs if the maximum acreage is bigger than the median. We use 2004 as the reference year for the DID models.

A second important characteristic is the total number of species included in a plan. The FWS promotes multi-species HCPs as a way of implementing an ecosystem approach to conservation. The implicit expectation is that plans covering a larger number of species will be more effective in promoting recovery. Landowners may be more likely to pursue multispecies plans for two main reasons. If their property could host more than one listed species, then preparing a single plan covering multiple species may be less costly than preparing several individual HCPs. Additionally, if the property provides habitat for other species that are not yet listed, but which may be in the future, then preparing an HCP that includes these species can provide additional certainty that future listings will not result in further restrictions.

To examine whether HCPs that cover more additional species have greater impacts on recovery than HCPs covering fewer species, we construct a variable that measures the average number of additional species covered by a species' plans. We include this variable in the CRE ordered probit model, along with squared number of additional species. As above, we define two additional treatment variables for the treatment effects models. A species has HCPs with few additional species if the average number of additional species covered by its plans up until 1998 is less than the median for the sample up to that year. A species has HCPs with many additional species if the average number of additional species is larger than the median. As before, we use 2004 as the reference year for the DID models. The results are presented in table 7.

The marginal effects for the CRE ordered probit indicate that larger HCPs decrease the probability that a species is classified as Extinct or Declining, and increase the probability that it is classified as Stable or Improving. Increasing the acreage of a species' plans by 100,000 acres would lower the probability that a species is classified as Extinct by 0.04 percentage points and the probability that it is Declining by 0.32 percentage points. An additional hundred thousand acres covered by HCPs increase the probability that a species is classified as Stable by 0.15 percentage points, and the probability that it is Improving by 0.21 percentage points[26]. The results for the treatment evaluation models generally support this conclusion. ATEs are larger for big HCPs than for small HCPs, although the estimates for the DID model are not statistically significant. ATEs for the matching models suggest that big HCPs can increase the recovery score by as much as 0.87 points, whereas small HCPs can increase it by at most 0.35 points.

The results for number of additional species included in plans are mixed. The marginal effects for the ordered probit are not statistically significant, which suggests that plans covering more species do not yield additional benefits in terms of promoting recovery. The ATEs for the OLS models and the nearest neighbor propensity score matching model support this result. On the other hand, for the remaining models ATEs for plans with more species are larger than ATEs for plans with fewer species, suggesting a positive impact of species covered on recovery. Hence, we do not find consistent support for the hypothesis that plans covering more species have a bigger impact on species recovery. Note, however, that this is conditional on the positive effect of having an HCP. That is, this result suggests that, although having HCPs is beneficial for recovery, we cannot conclude that adding species to an HCP provides *additional* benefits.

This lack of strong evidence for the effectiveness of multi-species plans may seem counterintuitive. A multi-species approach makes biological sense because it might increase the likelihood of creating an effective reserve system, and more comprehensive HCPs are generally expected to be more effective in preserving biological resources (Beatley 1994). In fact, a

---

[26] The mean acreage covered by a species HCPs in 2004 was almost 600,000 acres, so an additional 100,000 acres represents an increase of 17%.

guiding principle of the FWS' HCP handbook is to encourage regional and multi-species plans and to assist such efforts to the maximum extent practicable (FWS 1996). However, our mixed result agrees with the literature examining multi-species HCPs. Smallwood et al. (1998) argue that often multi-species plans intended to provide coverage for many species actually focus on just one. Boersma et al. (2001) and Taylor et al. (2005) find that species in multi-species recovery plans are less likely to show improving trends in status than species in single-species plans. Rahn et al. (2006) argue that, while a comprehensive planning approach may seem reasonable, it carries the risk that the needs of particular species may be overlooked, as many multi-species HCPs ignore the importance of species-specific conservation actions. Furthermore, the conservation gains of adding species may be illusory if the species added are not effectively provided for in the plan. They find that many plans are excessively broad, covering species for which they provide no localized scientific information. They conclude that multi-species planning will not always guarantee effective conservation.

7. Discussion and Conclusions

Our results indicate that, on average, HCPs have positive effects on endangered species recovery. From 1990 to 2004, species with plans are more likely to show improvement in recovery status and less likely to be declining or classified as extinct than species without an HCP. Additionally, average treatment effects indicate that recovery scores are between one-third and two-thirds of a point higher for species with HCPs. Our analysis also indicates that the positive impacts on recovery are larger when a species has plans encompassing more acreage. In contrast, we find mixed evidence about the effect of HCPs which cover relatively more species.

These results have important implications for endangered species policy. First, evidence of the positive effects of plans suggests that the FWS and NMFS should further encourage private parties to develop HCPs, particularly for listed species that currently do not have a plan. Preparation of an HCP can be a lengthy and cumbersome, and therefore costly, process. Efforts to reduce the burden on applicants by expediting and streamlining this process, as well as expansion of the FWS' Habitat Conservation Planning grants program may well pay off in terms of improved recovery outcomes. Additionally, these results suggest that HCP-type agreements could be valuable in more general policy contexts in which land use, property rights, and conservation are in conflict. This could provide an additional policy tool, which could be combined with the more commonly debated option of compensation (Innes et al. 1998).

Second, our results suggest that species benefit from inclusion in spatially larger plans. Third, our results indicate that the FWS' policy of encouraging multispecies HCPs may be misguided, as there is no strong evidence that plans with more additional species have larger positive impacts on recovery. Multispecies plans, although appealing from a biological perspective, may not be as successful as smaller plans if monitoring and conservation resources have to be spread more thinly and all species on the plan are not effectively provided for. Multispecies plans could be even more time-consuming and expensive to prepare and administer than single species plans. Hence, more comprehensive plans should be encouraged only to the extent that FWS and NMFS can ensure that the needs of individual species are not overlooked.

These findings should not be interpreted to mean that common criticisms of the HCP policy and individual plans are unfounded. Our results say something about the impacts of HCPs on a representative species, not about particular plans or the conservation planning process. However, they do suggest that, despite their potential flaws and the lack of a legal mandate for promoting recovery, HCPs can have a positive impact on species recovery and that, overall, endangered species' chances of survival are greater with an HCP than without.

References

Abadie, A., and G. Imbens. 2002. "Simple and Bias-Corrected Matching Estimators." Technical report, Department of Economics, University of California, Berkeley.

Abadie, A., and G. Imbens. 2002. "Large Sample Properties of Matching Estimators for Average Treatment Effects." *Econometrica* 74: 235 – 267.

Bean, M.J., and D.S. Wilcove. 1997. "The Private Land Problem" *Conservation Biology* 11: 1-2.

Beatley, T. 1994. *Habitat Conservation Planning. Endangered Species and Urban Growth.* Austin: University of Texas Press.

Bingham, B.B., and B.R. Noon. 1997. "Mitigation of Habitat "Take": Application to Habitat Conservation Planning." *Conservation Biology* **11**(1): 127-139.

Blume, L., D. Rubinfeld, and P. Shapiro. 1984. "The Taking of Land: When Should Compensation Be Paid?" *Quarterly Journal of Economics* 99(1): 71-92.

Boersma, P.D., P. Kareiva, W.F. Fagan, J.A. Clark, and J.M. Hoekstra. 2001. "How Good Are Endangered Species Recovery Plans? *BioScience* **51**: 643–649.

Brussard, P., G. Meffe, D. Murphy., B. Noon, R.F. Noss, J. Quinn, K. Ralls, M. Soulé, and R. Tracy. 1997. "A Statement on Proposed Private Land Initiatives and Reauthorization of the Endangered Species Act from the Meeting of Scientists at Stanford University." In R.F. Noss, M.A. O'Connell, and D.D. Murphy, *The Science of Conservation Planning*. Washington, DC: Island Press.

Cameron, C.A. and P.K. Trivedi. 2005. *Microeconometrics. Methods and Applications*. New York: Cambridge University Press.

Cash, D.W. 2001. "Beyond Cute and Fuzzy: Science and Politics in the U.S. Endangered Species Act." In J. Shogren and J. Tschirhart, eds. *Protecting Endangered Species in the United States: Biological Needs, Political Realities, and Economic Choices*. Cambridge: Cambridge University Press.

Cash, D.W., J. R. DeShazo, A. Metrick, S. Shapiro, T. Schatzki, and M. Weitzman. 1998. *Database on the Economics and Management of Endangered Species* (*DEMES*). Harvard University: Department of Economics.

Chamberlain, G. 1980. "Analysis of Covariance with Qualitative Data." *Review of Economic Studies* 47: 225-238.

DeShazo, J.R. and J. Freeman. 2006. "Congressional Politics." In D.D. Goble, J.M. Scott, and F.W. Davis, eds. *The Endangered Species Act at Thirty. Renewing the Conservation Promise.* Vol. 1. Washington, DC: Island Press, pp. 68-74.

Ferraro, P.J., C. McIntosh, and M. Ospina. 2007. "The Effectiveness of the US Endangered Species Act: An Econometric Analysis Using Matching Methods." *Journal of Environmental Economics and Management* **54**: 245-261.

Ferrer-i-Carbonell, A. and P. Frijters. 2004. "How Important is Methodology for the Estimates of the Determinants of Happiness?" *The Economic Journal* 114: 641-659.

Ginther, D.K. 2000. "Alternative Estimates of the Effect of Schoolings on Earnings." *The Review of Economics and Statistics* 82(1): 103-116.

Harding, E.K., E.E. Crone, B.D. Elderd, J.M. Hoekstra, A.J. McKerrow, J.D. Perrine, J. Regetz, L. J. Rissler, A.G. Stanley, E.L. Walters, and NCEAS Habitat Conservation Plan Working Group. 2001. "The Science Foundations of Habitat Conservation Plans: a Quantitative Assessment." *Conservation Biology* **15**(2): 488-500.

Honnold, D., J.A. Jackson, and S. Lowry. 1997. "Habitat Conservation Plans and the Protection of Habitat: Reply to Bean and Wilcove." *Conservation Biology* **11**(2): 297-299.

Hood, L.C. 1998. *Frayed Safety Nets. Conservation Planning Under the Endangered Species Act.* Washington, DC: Defenders of Wildlife.

Ichino, A., F. Mealli, and T. Nannicini. 2008. "From Temporary Help Jobs to Permanent Employment: What Can We learn From Matching Estimators and Their Sensitivity?" *Journal of Applied Econometrics* 23: 305 – 327.

Imbens, G.W., and D. B. Rubin. Forthcoming. *Causal Inference in Statistics and the Social Sciences*. Cambridge and New York: Cambridge University Press.

Imbens, G.W., and J.M. Wooldridge. 2009. "Recent Developments in the Econometrics of Program Evaluation." *Journal of Economic Literature* 47(1): 5–86.

Innes, R. 1997. "Takings, Compensation, and Equal Treatment for Owners of Developed an Undeveloped Property." *The Journal Of Law and Economics* 40(2): 403 – 432.

Innes, R. 2000. "The Economics of Takings and Compensation When Land and its Public Use Value are in Private Hands." *Land Economics* **76**: 195-212.

Innes, R., S. Polasky and J. Tschirhart. 1998. "Takings, Compensation and Endangered Species Protection on Private Land." *Journal of Economic Perspectives* 12(3): 35-52.

Kareiva, P., S. Andelman, D. Doak, B. Elderd, M. Groom, J. Hoekstra, L. Hood, F. James, J. Lamoreux, G. LeBuhn, C. McCulloch, J. Regetz, L. Savage, M. Ruckelshaus, D. Skelly, H. Wilbur, K. Zamudio, and NCEAS HCP Working Group. 1998. "Using Science in Habitat Conservation Plans." National Center for Ecological Analysis and Synthesis, Santa Barbara, CA, and American Institute of Biological Sciences, Washington, DC.

Kerkvliet, J. and C. Langpap. 2007. "Learning From Endangered and Threatened Species Recovery Programs: A Case Study Using U.S. Endangered Species Act Recovery Scores." *Ecological Economics* 63: 499 – 510.

Kostyack, J. 1998. "Surprise!" *The Environmental Forum*. March-April 1998: 19-28.

Langpap, C. 2006. "Conservation of Endangered Species: Can Incentives Work for Private Landowners?" *Ecological Economics* **57**: 558-572.

Langpap, C., and J. Wu. 2004. "Voluntary Conservation of Endangered Species: When Does No Regulatory Assurance Mean No Conservation?" *Journal of Environmental Economics and Management* **47**(3): 435-457.

Langpap, C., and J. Kerkvliet. 2010. "Allocating Conservation Resources under the Endangered Species Act." *American Journal of Agricultural Economics* 92(1): 110-124.

League of Conservation Voters. 1990-2004. *National Environmental Scorecard*. http://www.lcv.org/scorecard/.

Lee, M. 2005. *Micro-Econometrics for Policy, Program, and Treatment Effects*. Oxford and New York: Oxford University Press.

Lehmkuhl, J., C. Loggers, J. Creighton. 2004. "Wildlife Considerations for Small Diameter Timber Harvesting." In D.M. Baumgartner, L.R. Johnson, and E.J. DePuit, Eds. *Small Diameter Timber: Resource Management, Manufacturing, and Markets*. Washington State University Cooperative Extension, Pullman. MISC0509.

List, J. A., M. Margolis, and D.E. Osgood. 2006. "Is the Endangered Species Act Endangering Species?" NBER Working Paper No. 12777.

Male, T.D. and Bean, M.J. 2005. "Measuring Progress in U.S. Endangered Species Conservation." *Ecology Letters* **8**: 986 – 992.

Manski, C.F. 1994. "The Selection Problem." In C. Sims, Ed., *Advances in Econometrics*. New York, Cambridge University Press.

Michael, J., and D. Lueck. 2003. "Preemptive Habitat Destruction Under the Endangered Species Act." *Journal of Law and Economics* **46**: 27-60.

Moser, D.E. 2000. "Habitat Conservation Plans under the U.S. Endangered Species Act: the Legal Perspective." *Environmental Management* **26**(Suppl. 1): S7-S13.

Nannicini, T. 2007. "Simulation-Based Sensitivity Analysis for Matching Estimators." *The Stata Journal* 7(3): 334 – 350.

National Research Council. 1995. *Science and the Endangered Species Act.* Washington, DC: National Academy Press.

NatureServe. 2007. NatureServe Explorer. http://www.natureserve.org/explorer/.

Polasky, S., and H. Doremus. 1998. "When the Truth Hurts: Endangered Species Policy on Private Land with Imperfect Information." *Journal of Environmental Economics and Management* 35(1): 22- 47.

Rahn, M.E., H. Doremus, and J. Diffendorfer. 2006. "Species Coverage in Multispecies Habitat Conservation Plans: Where's the Science?" *BioScience* **56**(7): 613-619.

Ruhl, J.B. 1999. "How to Kill Endangered Species, Legally: The Nuts and Bolts of Endangered Species Act HCP Permits for Real Estate Development." *The Environmental Lawyer* **5**: 345 – 405.

Shilling, F. 1997. "Do Habitat Conservation Plans Protect Endangered Species?" *Science* **276**(5319): 1662-1663.

Smallwood, K., B. Wilcox, R. Leidy, R., and K. Yarris. 1998. "Indicators Assessment for Habitat Conservation Plan of Yolo County, California, USA." *Environmental Management* **22**: 947–958.

Taylor, M.F.J., K. F. Suckling, and J. R. Rachlinski. 2005. "The Effectiveness of the Endangered Species Act: A Quantitative Analysis." *BioScience* 55: 360-367.

U.S. Dept. of Agriculture. 1990-2004. "Agricultural Land Values and Cash Rents: Annual Summary, Economics, Statistics, and Market Information System."

U.S. Fish and Wildlife Service. 1996. "Habitat Conservation Planning and Incidental Take Permit Processing Handbook." http://www.fws.gov/endangered/hcp/hcpbook.html

U.S. Fish and Wildlife Service. 1990a-2004a. *Report to Congress on the Recovery Program for Threatened and Endangered Species.* Washington, DC.

U.S. Fish and Wildlife Service. 1989-2004b. *Federal and State Endangered and Threatened Species Expenditures*. Washington, DC.

U.S. Fish and Wildlife Service. 2012. *Conservation Plans and Agreements Database.* http://ecos.fws.gov/conserv_plans/public.jsp. Accessed February 21, 2011.

Watchman, L.H., M. Groom, and J. D. Perrine. 2001. "Science and Uncertainty in Habitat Conservation Planning." *American Scientist* 89(4): 351-359.

Wooldridge, J. M. 2002. *Econometric Analysis of Cross Section and Panel Data*. Cambridge: The MIT Press.

Table 1. Sample Means

Variable	1990	1992	1994	1996	1998	2000	2002	2004	HCPs[b]	No HCPs[b]
Number of Species in Sample	176	187	204	214	217	220	225	213		
FWS Status	1.67	1.72	1.71	1.69	1.75	1.74	1.68	1.66	1.58	1.73
HCP	0.05	0.05	0.11	0.17	0.21	0.23	0.25	0.26	-	-
Total HCPs[a]	1.75	1.80	2.41	6.13	6.47	8.30	12.32	13.73	-	-
HCP Size (1000s acres)[a]	25.76	60.52	39.52	46.75	108.12	691.51	600.97	585.10	-	-
Other Species HCP[a]	1.83	1.86	5.82	25.40	18.25	20.05	18.71	22.96	-	-
Cumulative Spending ($ millions)	0.51	1.66	2.37	3.03	3.63	4.42	5.91	7.78	10.32	2.11
Final Plan	0.44	0.44	0.52	0.50	0.55	0.48	0.66	0.54	0.47	0.53
Mammal	0.21	0.20	0.18	0.17	0.18	0.17	0.19	0.19	0.30	0.16
Amphibian	0.03	0.03	0.03	0.05	0.04	0.05	0.06	0.06	0.10	0.04
Bird	0.32	0.28	0.30	0.28	0.27	0.28	0.28	0.27	0.32	0.28
Reptile	0.09	0.10	0.10	0.10	0.12	0.11	0.09	0.09	0.21	0.08
Body Length (cm)	40.11	42.41	40.24	40.46	38.53	38.89	39.73	40.31	44.76	39.58
Land Values ($/acre)	1,155	1,153	1,448	1,515	1,597	1,756	2,083	2,363	1,555	2,156
Distinct	0.36	0.36	0.38	0.35	0.36	0.34	0.34	0.34	0.33	0.36
Conflict	0.38	0.42	0.41	0.42	0.42	0.42	0.44	0.45	0.73	0.36
League Cons. Voters	20.05	13.12	21.28	17.72	29.80	27.40	38.10	29.73	36.98	21.68
Population Growth	0.04	0.04	0.03	0.03	0.03	0.06	0.03	0.01	0.03	0.03

[a] Mean over all HCPs for a species conditional on the species having a plan (*HCP* = 1)
[b] Mean over all years.

Table 2. First-Stage HCP and Cumulative Spending Models

	HCP		*Cumulative Spending*	
Variable	*Coefficient*	*Std. Error*	*Coefficient*	*Std. Error*
Constant	-0.049	0.037	-8.708***	1.084
Past HCPs	0.002***	0.0004	0.013	0.013
No Surprises Policy	0.092***	0.027	-0.752	0.783
League Cons. Voters (Modified)	0.002***	0.0002	0.018***	0.007
Final Plan	-0.030**	0.015	-0.992**	0.421
Mammal	-0.048	0.041	-2.910**	1.193
Amphibian	0.053	0.043	-2.845**	1.265
Bird	-0.041	0.040	2.162*	1.160
Reptile	0.133***	0.041	-3.161***	1.188
Body Length (1000)	-0.860***	0.336	54.025***	9.732
Body Length^2 ($\times$ 1000)	-0.0001	0.0009	-0.126***	0.027
Distinct	-0.025	0.017	-0.308	0.484
Conflict	0.167***	0.016	3.760***	0.456
Non-migratory	0.041**	0.017	1.190**	0.501
Locally migratory	0.063***	0.022	5.079***	0.628
Distance migratory	0.079**	0.036	1.410	1.040
Diurnal	0.021	0.023	1.117*	0.665
Nocturnal	0.114***	0.026	2.604***	0.763
Carnivore	0.051*	0.027	3.197***	0.776
Herbivore	0.016	0.019	2.104***	0.538
Terrestrial Habitat	0.086***	0.027	2.017***	0.783
Subterranean Habitat	-0.116***	0.042	2.091*	1.209
Estuarine Habitat	0.021	0.030	0.683	0.860
Lacustrine Habitat	-0.015	0.022	0.141	0.647
Palustrine Habitat	-0.046**	0.018	-1.191**	0.533
Marine Habitat	0.040	0.045	3.308**	1.299
Riverine Habitat	-0.122***	0.030	1.861**	0.867
Special Habitat Needs	0.043**	0.018	1.519***	0.511
Trend	0.002	0.007	0.920***	0.197
Observations	2092		2086	
F-Stat. for Instruments (*p*-value)	19.14	(0.000)	2.93	(0.033)
Adj. R^2	0.27		0.26	
p value - LR Test [a]	0.000		0.000	

Note: *, **, *** indicate parameter significance at α = 10%, 5%, and 1% respectively
[a] *p* - value for likelihood ratio test of joint significance of regressors.

Table 3. Recovery Model – IV CRE Ordered Probit Coefficient Estimates

Dependent Variable: *Status*	*Coefficients*	*Standard Errors*
HCP	2.711***	0.775
Cumulative Spending	0.150**	0.069
Final Plan	0.162*	0.088
Mammal	0.416	0.303
Amphibian	-1.364***	0.310
Bird	-0.356	0.278
Reptile	-1.260***	0.315
Body Length	0.002	0.003
Body Length ^{2}	0.004 E-03	0.007 E-03
Distinct	-0.371***	0.112
Conflict	-0.837***	0.196
Non-migratory	-0.137	0.140
Locally migratory	-0.345	0.242
Distance migratory	-0.192	0.276
Carnivore	-0.369	0.229
Herbivore	-0.086	0.144
Diurnal	0.087	0.146
Nocturnal	-0.834***	0.219
Terrestrial Habitat	0.248	0.193
Subterranean Habitat	1.505***	0.321
Estuarine Habitat	0.304	0.228
Lacustrine Habitat	0.042	0.145
Palustrine Habitat	0.367***	0.120
Marine Habitat	0.115	0.312
River Habitat	0.398*	0.238
Special Habitat Needs	-0.257*	0.136
Mean HCP	-0.586***	0.194
Mean Cumulative Spending	0.007	0.007
Trend	-0.150***	0.037
Observations	1675	
Log Likelihood	-1357.630	

Note: *, **, *** indicate parameter significance at α = 10%, 5%, and 1% respectively. Standard errors in parentheses.

Table 4. Effects of HCPs on Species Recovery

Estimator	Effects on Recovery
IV CRE Ordered Probit	*Marginal Effects*
Prob (Status = Extinct)	- 0.057***
	(0.019)
Prob (Status = Declining)	- 0.435***
	(0.131)
Prob (Status = Stable)	0.165***
	(0.021)
Prob (Status = Improving)	0.326***
	(0.096)
Treatment Evaluation Models	*Average Treatment Effects*
OLS on covariates	0.683**
	(0.328)
OLS on propensity score	0.409**
	(0.200)
Nearest-neighbor covariate matching (*inverse variance*)	0.389** (0.187)
Nearest-neighbor covariate matching (*Mahalanobis*)	0.456** (0.229)
Nearest-neighbor propensity score matching	0.508***
	(0.183)
Kernel propensity score matching (*Gaussian*)	0.534*** (0.176)
Difference in Differences	0.297**
	(0.144)

Note: *, **, *** indicate parameter significance at α = 10%, 5%, and 1% respectively.
Standard errors in parentheses.
Number of Observations: IV CRE Ordered Probit: 1,675; OLS and Matching Estimators: 175; DID: 130.

Table 5. Sensitivity Analysis for Ordered Dependent Variable Model

	IV RE Ordered Probit	*Linear IV FE*	*IV FE Ordered Logit*	*Bivariate Probit*
HCP	2.623***	3.397*	9.373***	1.009***
	(0.787)	(1.774)	(2.804)	(0.348)
Observations	1675	1675	472	1675
Log Likelihood	-1361.367		-180.902	-1124.590

Note: *, **, *** indicate parameter significance at α = 10%, 5%, and 1% respectively

Table 6. Sensitivity Analysis for Treatment Evaluation Models

Estimator	Average Treatment Effects			
	Exact Matching on Taxonomy	*Exact Matching on Baseline*	*Exact Matching on Baseline and Taxonomy*	*Outcome Variable: 2004 Recovery Status*
OLS on covariates				0.564***
				(0.175)
OLS on propensity score				0.295*
				(0.174)
Nearest-neighbor covariate matching (inverse variance)	0.467**	0.270*	0.324*	0.304**
	(0.222)	(0.151)	(0.177)	(0.124)
Nearest-neighbor covariate matching (Mahalanobis)	0.477**	0.286*	0.330*	0.326***
	(0.242)	(0.152)	(0.181)	(0.125)
Nearest-neighbor propensity score matching	0.525*	0.297*	0.350**	0.350**
	(0.269)	(0.163)	(0.163)	(0.176)
Kernel propensity score matching (Gaussian)				0.252
				(0.198)
Observations	175	175	175	219

Note: *, **, *** indicate parameter significance at α = 10%, 5%, and 1% respectively.
Standard errors in parentheses.

Table 7. Effects of HCP Size and Number of Species on Recovery

Estimator	Effects on Recovery			
IV CRE Ordered Probit	*Marginal Effects-Size*		*Marginal Effects-Number of Species*	
Prob (Status = Extinct)	-0.004**		0.36E-04	
	(0.002)		(2.56E-04)	
Prob (Status = Declining)	-0.032**		2.91E-04	
	(0.015)		(0.002)	
Prob (Status = Stable)	0.015***		-1.35E-04	
	(0.005)		(6.69E-04)	
Prob (Status = Improving)	0.021**		-1.92E-04	
	(0.010)		(0.001)	
Treatment Evaluation Models	*Average Treatment Effects*			
	Small HCPs	*Big HCPs*	*Few Additional Species*	*Many Additional Species*
OLS on covariates	0.380*	1.745**	0.585	0.276
	(0.225)	(0.767)	(0.357)	(0.317)
OLS on propensity score	0.304*	0.575	0.358	0.528
	(0.186)	(0.311)	(0.238)	(0.339)
Nearest-neighbor covariate matching (*inverse variance*)	0.282*	0.867***	0.248*	0.933***
	(0.165)	(0.274)	(0.145)	(0.308)
Nearest-neighbor covariate matching (*Mahalanobis*)	0.350*	0.749***	0.294*	0.752**
	(0.186)	(0.253)	(0.177)	(0.330)
Nearest-neighbor propensity score matching	0.333*	0.670**	0.538**	0.451
	(0.176)	(0.341)	(0.230)	(0.382)
Kernel propensity score matching (*Gaussian*)	0.327*	0.704**	0.525*	0.593*
	(0.179)	(0.347)	(0.279)	(0.338)
Difference in Differences	0.265	0.317	0.199	0.418**
	(0.190)	(0.201)	(0.185)	(0.194)

Note: *, **, *** indicate parameter significance at α = 10%, 5%, and 1% respectively.
Standard errors in parentheses.
Number of Observations:
IV CRE Ordered Probit: 1,655
OLS and Matching Estimators: 154 (Small HCPs), 164 (Big HCPs), 156 (Few Other Species), 160 (Many Other Species)
DID: 116 (Small HCPs), 112 (Big HCPs), 118 (Few Other Species), 109 (Many Other Species)

Figure 1. Number of Habitat Conservation Plans per Year

In separate TIF file.