Noah's Ark in a Warming World: Climate Change, Biodiversity Loss and Public Adaptation Costs in the United States

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Abstract

Climate change poses a growing threat to biodiversity, but the welfare consequences of these changes are not well understood. Here we analyze data on the US Endangered Species Act and project increases in species listing and spending due to climate change. We show that higher endangerment is strongly associated with the probability of listing, but also find a large bias towards vertebrate species for both listing and spending. Unmitigated warming would cause the listing of an additional 690 species and committed spending of $21 billion by 2100. Several thousand more species would be critically imperiled by climate change but remain unlisted. Finally, we compare ESA spending with estimates of willingness to pay for conservation of 36 listed species. Aggregate WTP is larger than ESA spending for the vast majority of species even using conservative assumptions, and typically 1-2 orders of magnitude larger that direct ESA spending using less restrictive assumptions.

Introduction

Humans co-exist on Earth with immense, but still not fully understood, biological diversity – likely the richest assemblage of life to ever exist on the planet (Benton 1994). Human influence has, however, taken a toll on this diversity. Human beings appropriate an estimated 25% of the Earth's net primary productivity and 35% of its land area to support just a few dozen domesticated crop and livestock species, reducing the richness and abundance of natural systems (FAO 2016; Krausmann et al. 2013; Newbold et al. 2015). Almost 200 vertebrate species have gone extinct since 1900, a rate at least 8 times the background rate inferred from the fossil record (Ceballos et al. 2015).

Anthropogenic climate change poses a further threat to global biodiversity through a number of pathways – the climatological niche may become too small to support a viable population; limits to dispersal mean species may not be able to move with shifting climate envelopes; phenological changes could disrupt food-webs, particularly for migratory species; and wide-spread tree die-offs and wildfires linked to climate change could directly destroy habitat (Bellard et al. 2012). Climate change has already been implicated in the extinction of at least one species (Pounds et al. 1999; Thomas et al. 2004) and projections for the future are that unmitigated climate change over the 21st century will threaten somewhere between 10 and 15% of species with extinction (Urban 2015). The US Fish and Wildlife Service (USFWS) has already listed the polar bear and has proposed listing the North American wolverine, primarily due to the risk of habitat loss from climate change (Blumm and Marienfeld 2014). Increasing extinction risk has implications for current and future generations, but this important climate change cost is not well understood.

Estimating the costs of increasing extinction risk is challenging. Accounting for the value of a species' simple existence requires application of stated-preference nonmarket valuation methods. The most notable of these methods, contingent valuation, has been the subject of on-going debate for nearly thirty years (e.g. Haab, Interis, Petrolia, & Whitehead, 2013; Hausman, 2012; Kling, Phaneuf, & Zhao, 2012). This continued debate has left existence values often sidelined in the evaluation of conservation policies, despite the development of alternative choice experiment methods that address many of the original methodological concerns (Adamowicz et al. 1994; Hanley et al. 1998).

Despite large gaps in the data that economists would normally use in policy evaluation, it is an empirical fact that many governments around the world, particularly those in richer countries, devote resources to preventing extinctions (Waldron et al. 2013). Although separating consumptive use from non-consumptive and non-use values may be challenging, most wealthy countries have policies, laws, and spending that aim explicitly to prevent species extinctions, irrespective of any consumption value those species do or do not provide for people.

This fact raises both positive and normative questions for environmental economics. First, given that biodiversity preservation is a policy goal for many governments, economics can provide insight into the most cost-effective means of meeting those goals: how should a limited budget be prioritized so as to protect the largest diversity of species? Second, patterns of spending on biodiversity may reveal a signal, albeit a noisy signal filtered through a complex political process, of public preferences over different and competing priorities for species protection. Professor Weitzman tackled both these questions in an extensive research program over the 1990s.

In a series of papers, Weitzman developed a mathematical description of diversity based on pairwise dissimilarity, formally connecting it to the branching tree structure often used to describe evolutionary relationships and the genetic information content of extant species (Weitzman, 1992, 1998). Using data on crane species, he provided an illustration of how this approach could be applied to inform conservation priorities, highlighting the complex interactions between evolutionary relationships, probability of survival, and the effectiveness of conservation spending that determine the optimal allocation of resources (Weitzman, 1993). In the well-known "Noah's Ark Problem", Weitzman (1998) advanced the theory of conservation spending further, developing a simple and robust ranking for cost-effective spending that depends on evolutionary distinctiveness, utility value to humans, and the effectiveness of spending at increasing survival probability.

In two other papers, Weitzman, together with Andrew Metrick, analyzed what was then still new data on federal and state spending under the US Endangered Species Act (ESA) (Metrick and Weitzman 1998, 1996). In contrast to the line of work just described, these papers are primarily descriptive, an attempt to empirically distinguish the priorities implied by conservation decisions amid the "shopping list of objectives" typically used to justify it:

> *"Decisions about endangered species reflect the values, perceptions, and contradictions of the society that makes them. Thus … this paper addresses some very general issues about humankind's relation to nature and about our choices when confronted by competing and often unquantifiable objectives."* (Metrick & Weitzman, 1996, p. 1)

In these papers, Metrick and Weitzman developed several proxies of the variables that appear in the optimal allocation results in "Noah's Ark Problem", such as distinctiveness (measured as a species being the only member of its genus), utility value (measured using taxonomic group and body size), and degree of threat (measured using a scientific evaluation by The Nature Conservancy). They found evidence that both scientific considerations (i.e. degree of threat and biological distinctiveness) and variables measuring the utility value to humans play a roll in both the listing and spending decisions.

Other papers have since examined the effect of ESA spending on species recovery (Kerkvliet and Langpap 2007), the optimal allocation of ESA spending (Langpap and Kerkvliet 2010), and the costs of other aspects of endangered species protection such as controls over federal land use or restrictions on private development (Langpap et al. 2018; Ando 2001) and the political economy of lobbying over listing decisions (Ando 2003). However, few recent papers have revisited Metrick and Weitzman's original question of what social priorities seem to govern either the listing of species or the allocation of resources between species (Dawson & Shogren, 2001 and Kerkvliet & Langpap, 2007 being the exceptions).

Here we develop models of ESA listing and spending and combine them with projections of increasing extinction risk under climate change to provide a rough estimate of the change in the number of listed species, spending under the ESA, and species that will be imperiled by climate change but remain unlisted in order to bound the costs associated with climate-change driven biodiversity loss in the United States. These costs can be broken down into four parts, of which we explicitly quantify only the first:

1. Direct spending on the conservation of protected species under the ESA. These could include conservation activities such as research, census, habitat maintenance or transplantation, as well as land and habitat acquisition. These are the costs considered in this paper.
2. Direct spending on conservation is a form of public adaptation. Because public funds must be raised from distortionary taxes, each dollar of public spending produces additional effects that

lead the total welfare cost to be greater than $1. Barrage (2020) has demonstrated the importance of these effects for the welfare costs of climate change and suggests they increase the costs of direct public spending by 4 - 53%, depending on the distortionary effect of the revenue raising mechanism.

3. ESA listing entails protections that place limits on the use and development of private land. These restrictions, discussed more fully in Section 3, entail opportunity costs that have not been systematically quantified for all listed species. It is likely that these would increase with the number of listed species, but are not quantified here.
4. Our results suggest only a small fraction of species imperiled by climate change will be listed, meaning climate change will likely cause extinction of some species. The lost existence value from these extinctions cannot be quantified in this paper. Section 4 however provides a comparison between direct spending on particular endangered species and WTP estimates derived from stated preference studies, providing some indication of the order of magnitude of these costs relative to direct spending, at least for this small subset of species.

Beyond our assessment of climate change-driven costs, the paper extends the original analysis by Metrick and Weitzman in several ways. First, the quantity of spending data has increased substantially since the early 1990s. This results both from a longer (24 year) time series and a much larger set of listed species. This much longer dataset allows us to avoid relying on the pooled cross-sectional models used in Metrick and Weitzman (1996, 1998) and to estimate a model with species fixed-effect that controls for unobserved variation between species, comparing it to a random-effects model that allows estimates of important time-invariant characteristics. In addition, Metrick and Weitzman's (1996, 1998) analysis was limited only to vertebrate species, meaning it is not clear how their findings might generalize to the far more numerous set of non-vertebrate species. Our analysis also includes plants and invertebrates, increasing the number of species considered from 511 in Metrick and Weitzman (1996) to over 64,500.

Next, advances in genetics mean that it is now possible to directly quantify the evolutionary distinctiveness of a large number of species, a measure of the genetic distance of a particular species from its nearest relatives. This concept figures prominently in Weitzman's theoretical results on optimal conservation spending (Weitzman, 1992, 1993) but is proxied only very imperfectly by taxonomic measures that have been previously used to assess conservation spending. We include recentlyavailable measures of the species phylogenetic distinctiveness as a variable explaining listing decision to assess the degree to which this has been incorporated into conservation policy.

Finally, the paper begins to address the question of the benefits of ESA listing through a comparison of the WTP for conservation of endangered species with direct ESA spending on those species. Interpreting ESA listing as an adaptation to climate change implies that the benefits exceed the costs. Although comprehensive benefit cost analysis of ESA listings is not possible here, this review of values reported in the stated preference literature begins to bound the benefit-cost ratios associated with listing.

The paper proceeds as follows: the next section introduces the data sources used. Section 2 presents statistical modeling results for two models, the listing status and the spending decision. Section 3 combines findings from these regressions with prior estimates from the ecological literature of the threat climate change poses to North American biodiversity to estimate the increase in species listing and spending with climate change. Section 4 provides a review of WTP estimates for 36 listed species and compares these values with the observed ESA spending. Section 5 concludes.

Section 1: Data Sources

1. *Listing and spending*

Our primary dependent variables of interest are 1) listing a species as threatened or endangered and 2) the level of spending received by that species, conditional on listing. These two factors will jointly determine how the increased risk of extinction posed by climate change translates into higher fiscal burdens under the ESA. Although the ESA requires the listing of species at risk of extinction, capacity and budgetary constraints mean that the process is slow and there is a backlog of species awaiting consideration (Alexander 2010). In some cases, USFWS will make a "warranted but precluded" designation, explicitly acknowledging that the biological threat of a species requires listing but denying listing because other species are a higher priority (Alexander 2010). Therefore, it seems likely that which species do or do not receive listing protection may be influenced both by scientific assessments of endangerment as well as other social, political, and economic factors that determine priority.

US Fish and Wildlife Service (USFWS) funding, which constitutes about 15% of the reported expenditures, is distributed among the eight USFWS administrative regions based on the total number of species assigned to each region and the estimated recovery costs of those species. Regional offices then distribute funds to field offices, a process that may be heavily influenced by field offices' "long-standing arrangements to work with partners to recover specific species" (GAO 2005). Field offices use funds to implement recovery plans of species under the office's jurisdiction, based on priority rankings (described further below) and partnerships with outside organizations. Therefore, like the listing decision, the spending decision also likely reflects both scientific factors such as the level of endangerment or potential for recovery, as well as the utility value (i.e. popularity) of particular species.

Species-specific federal and state spending on endangered species recovery is reported every year by the USFWS and the National Marine Fisheries Service (NMFS). NMFS oversees a small set of coastal and marine species – they constitute 46 out of 1613 listed species in our listing regression, but are dropped from our spending analysis because of the lack of covariate data, particularly on range area. Expenditures, which includes both spending on recovery efforts and land acquisition, were extracted from these reports for the period 1993-2016. Expenditures were adjusted to real 2017 dollars using the Bureau of Labor Statistics' Consumer Price Index. Entities eligible for protection by the ESA include species, subspecies, and distinct population segments (DPS). When expenditures were reported separately for distinct population segments of the same species, expenditures were aggregated to the species or (where appropriate) sub-species level. This is necessary since ESA spending and recovery data for DPS were not reported separately for many listings prior to 2003 and other covariates (such as conservation status and n-gram data, described more fully below) are not available at the DPS level.

The species-specific spending documented in these reports includes all spending that may be reasonably attributed to an individual species. Reported expenditures capture a range of conservation activities such as research, census, habitat-maintenance, propagation, live-trapping, and transplantation, as well as land acquisition, employee salaries, listing, consultation and law enforcement costs where these can be attributed to individual species (USFWS 2016). Reports include spending by both federal and state agencies, with the vast majority (92%) attributable to federal agencies. Spending is disaggregated by land acquisition vs other conservation costs, with land acquisitions constituting 20% of total expenditures. General operational expenditures that cannot be attributed to a particular species are not reported and are not included in this analysis.

Table 1 and Figure 1a give some summary figures from the spending data. Figure 1a shows median annual spending by taxon across the 24 years in the dataset. Large differences across groups of species are apparent in the raw data, in particular an order-of-magnitude difference between the vertebrate species (birds, mammals, reptiles, amphibians and fish) and the plants, invertebrates, and fungi. Table 1 shows the 25 species that receive the most spending and highlights the skewed distribution of spending among species, with a large emphasis on salmonid species, which are unusual among endangered species in having a large recreational and commercial value.

Species	Total Spending 1993-2016 (Millions 2017 $)	Percent of Total Expenditures
Chinook salmon[a]	4543	18
Steelhead[a]	3581	14
Coho salmon[a]	899	3
Sockeye salmon[a]	777	3
Bull Trout	748	3
Pallid sturgeon	735	3
Steller sea lion[a]	730	3
Red-cockaded woodpecker	583	2
Desert tortoise[b]	482	2
Chum salmon[a]	381	1
Northern spotted owl	381	1
Wood stork	381	1
Bald eagle[b]	354	1
Southwestern willow flycatcher	323	1
Coastal California gnatcatcher	293	1
Piping Plover[b]	278	1
Indiana bat	262	1
Razorback sucker	260	1
North Atlantic Right Whale[a]	239	<1
Louisiana black bear[b]	237	<1
West Indian Manatee	233	<1
Rio Grande Silvery Minnow	232	<1
Grizzly bear	224	<1
Colorado pikeminnow	219	<1
White sturgeon	203	<1

Table 1: Table of species with 25 largest total expenditures 1993 to 2016.

Notes: [a]Dropped from spending regressions due to NMFS listing missing USFWS covariates such as range area. [b]Dropped from spending regressions due to inaccurate range information (i.e. reported range corresponds exactly to state boundaries).

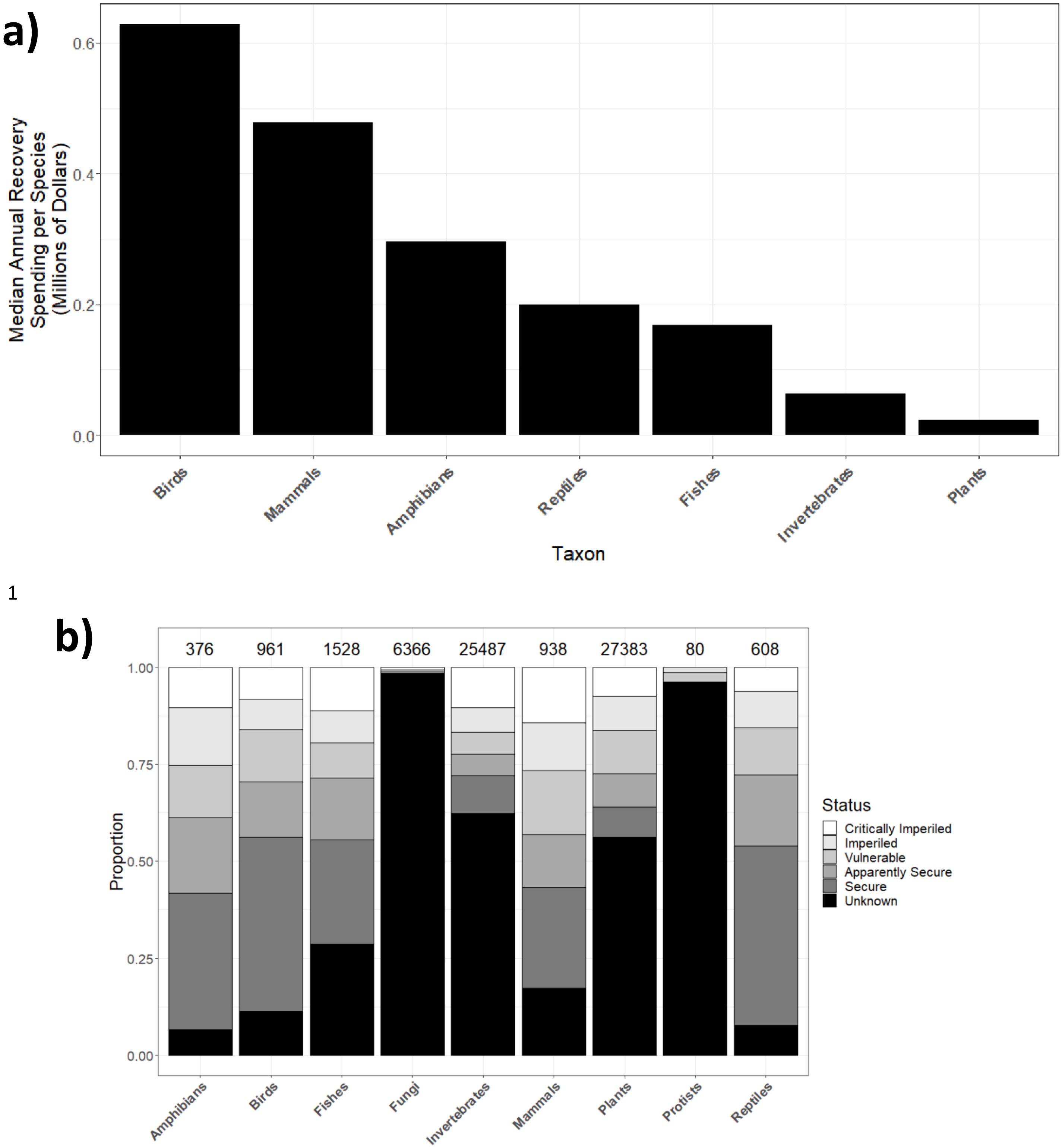

Figure 1: a) Annual spending by taxonomic grouping for the median species within each group. (Mean values are substantially higher for birds and fish because of the highly concentrated spending on a few species, as shown in Table 1). b) Distribution of assessed conservation status for 64,589 North American species and sub-species in the NatureServe dataset. Numbers above bars give the total number of species within each group.

2. *Other USFWS Covariates of Listed Species*

Since 1983, the USFWS has assigned priority numbers, intended to guide funding allocations, to each species. The priority number is a composite of three factors: degree of threat, potential for recovery, and taxonomic distinctiveness. These three factors are aggregated lexicographically into a single priority number that takes a value between 1 and 18, with the lexicographic ordering first prioritizing the degree of threat, followed by the potential for recovery, and lastly the taxonomic distinctiveness. In addition, USFWS assigns a "conflict" code, indicating whether or not the USFWS has determined that protection of the species conflicts with economic development. These priority numbers do change over time, but such changes are uncommon; 72% of species have a single priority number throughout the 21 year record. Historical priority numbers were obtained by species recovery reports, which are reported biennially and extrapolated to intervening years.

Additional information on listing and delisting dates, current federal listing status, lead USFWS administrative regions, species occurrence in USFWS administrative regions, and geographic range were retrieved from the USFWS's Environmental Conservation Online System. Species ranges offered by the USFWS represent the geographic area where a species is known or suspected to occur and omit historical areas of occurrence. Ranges that were not defined by the USFWS at a resolution finer than a US state level were excluded from analysis as possibly not providing an accurate assessment of the true species range. Data for distinct population segments of the same species were aggregated by taking the minimum priority number rank, the earliest listing date, and the mode for listing classifications (i.e. endangered or threatened) and lead regions across populations of the same species.

3. *Endangerment of Listed and Non-Listed Species*

The conservation status and taxonomic information for listed and non-listed species was obtained from the US non-profit NatureServe. Nature Serve tracks 53,417 full species and 11, 172 subspecies and varieties found within North America. Conservation status ranks are assigned on a five-point scale of imperilment ranging from "secure" to "critically imperiled" for extant species, with additional categories of "possibly extinct", "presumed extinct", and "unknown". When a species included a range of imperilment categories, reflecting uncertainty in a species' conservation status, we took the most conservative (i.e. most imperiled) conservation rank. Although NatureServe periodically revises its assessment of species, we have access only to the most recent assessment for each species. The most recent assessments were conducted between 1985 and 2019, with the vast majority occurring between 1996 and 2005 (mean date is 2003).

The details of the NatureServe ranking strategy are complex, but broadly speaking it is a weighted average of three components. The most important is rarity, defined by both population size and range extent. The other two components are the anthropogenic threat to populations and the short-term (10-50 year) and long-term (~200 year) trends in population or extent (Faber-Landgendoen et al. 2012). A breakdown of conservation status by different taxonomic categories is shown in Figure 1b. An important aspect of the dataset revealed clearly in Figure 1b, is the large number of species with unknown status Sixty percent of species in the dataset have an unknown status. These are almost entirely invertebrates (41%), plants (40%), and fungi (16%). This unknown status is not simply missing information, but informative in the sense that the lack of scientific attention itself reveals this set of species to be not highly prioritized in the conservation process.

4. *Google N-grams for Listed and Non-Listed Species*

In their original studies, Metrick and Weitzman emphasized both the importance and difficulty of measuring the utility value of species to humans, which they proxied using taxonomic groupings (i.e. mammal, bird, amphibian, reptile, and fish) and average body length (1996, 1998). In this analysis, we preserve the taxonomic groupings but add a proxy for the utility value of species using the frequency that species' names appear in books published in the English language. The benefits of this measure are that it integrates over many different ways in which the existence of a particular species may provide utility – a species may be culturally significant, may be of scientific interest either because of unique biological characteristics or the role it plays within an ecosystem, or may simply be cute, interesting, beautiful, or majestic. Some of these characteristics map only imperfectly onto body length, but would all likely influence the frequency that the species is written about. An additional benefit of this data is that it varies over time, allowing us to estimate the within-species effect of changing prominence on spending.

The Google Books N-gram Viewer provides the frequency of particular words or phrases within a corpus of over28 million books in 8 languages (Google 2020). Google Book's English 2019 corpus consists of over 16 million books and 1.9 trillion words published in the English language between 1470 and 2019 (Michel et al. 2011). To obtain n-gram frequencies for species names in Google Book's English 2019 corpus, case insensitive searches were performed for all scientific and common names in the NatureServe and ESA datasets for the years between 1800 and 2016. When a name failed to return valid data between the years 1800 and 2016 (Google Books N-gram Viewer only offers frequencies for words and phrases that occur in at least 40 books), the name was assigned a frequency of zero across all years. Some limitations of the n-gram measure are that it captures only English-language media and includes only material published in books, excluding some popular media such as magazines, websites or newspapers.

We treat scientific and common name n-grams separately because common names pose particular challenges. Species may be known vernacularly by multiple names or may lack a common name altogether. To help account for generic common names (e.g. “blackberry” for *Rubus ostryifolius*; “a millipede” for *Trigenotyla blacki*) and common names with additional meanings or uses (e.g. “small blue” for *Philotiella speciose*; “British soldier” for *Cladonia cristatella*), we discarded common name n-grams with a standardized frequency 10 times greater than the standardized scientific name n-gram frequency for a given species.

N-gram frequencies were aggregated to the species level (for the listing decision) and the species-by-year level (for the spending decision). Species level n-gram frequencies are the average from 1950 to the 2016, except for listed species. For listed species, we take the average frequency from 1950 to 10 years before the listing decision, to avoid any chance that publications generated by the listing decision itself contribute to the n-gram value. The species-by-year n-gram frequency used in the spending regression is the lagged 5 year rolling mean, which smooths out idiosyncratic year-to-year variation and allows for a delayed effect on spending patterns. Because raw n-gram frequencies are extremely small, all values are standardized before analysis.

5. Evolutionary Distinctiveness

We used two variables to represent species uniqueness. The first one, ‘Evolutionary Distinctiveness’ (ED) describes species’ relative contribution to the total evolutionary history, or phylogenetic diversity (PD). For a group of species comprising the extant descendants of a common ancestor (clade), PD of the clade is the sum of all branch lengths of the phylogenic tree, measured in millions of years. ED is the “fair proportion” (Hartmann 2013) of the total PD assigned to an individual species in that clade, with the length of each branch of the phylogeny divided equally amongst all species to which it is ancestral.

We used published ED scores for mammals (Gumbs et al. 2018), amphibians (Isaac et al. 2012; Safi et al. 2013; Gumbs et al. 2018), birds (Jetz et al. 2014; Gumbs et al. 2018), reptiles (Gumbs et al. 2018), and plants (Potter 2018), which were obtained from the EDGE (Evolutionary Distinctive and Globally

Endangered) of Existence program (Zoological Society of London, 2008) Overall, we collected ED score for 244 out of 309 amphibians (79%), 649 out of 768 birds (85%), 374 out of 439 mammals (85%), 328 out of 342 reptiles (96%), and 319 out of 19,092 plants (1.7%) in our list. No ED scores were available yet for fish, fungi, invertebrates, or protists in our study. Because the limited data availability for plants could imply a strong selection effect, we omit them from the regression that includes the ED score, considering only mammals, birds, amphibians and reptiles.

The second metric to describe species uniqueness is the number of species within a genus, a variable originally used in Weitzman's studies (1992, 1993). Compared to ED, data for this metric covers signficantly more species in our list. Decisions about assigining species to Linnean ranks above the species level are subjective and genera size can vary simply based on the tendency of particular taxonomists towards either lumping or splitting (Darwin 1857; Laurin 2010). However, because genera have historically been defined based on observable characteristics of species, it may be that this measure actually captures apsects of distinctiveness that are more salient to humans. The data on the number of species in a genus was obtained from two databases: the Integrated Taxonomic Information Service (ITIS) and NCBI (National Center for Biotechonology Information) taxonmy database, using '*taxize*' package in R.

The noisiness of genera size as a proxy for genetic distance is shown in Supplementary Figure 1. Although the relationship is in the expected direction (species in smaller genera tend to be more evolutionarily distinct), it explains only a very small fraction of the variance (R^2=3.6%), suggesting genus size is only a very imperfect measure of the genetic distinctiveness of a particular species.

Section 2: Model and Results

1. *Listing Status*

We model a species' listing status at the time of its NatureServe assessment as a binary outcome using a logistic regression. Although in situations with random treatment assignment and fixed-effects the linear probability model is sometimes preferred (Lancaster 2000), these conditions do not apply here. In our case the unconditional probability of listing is low, about 2.5%. In addition, because a goal of the analysis is estimating the effects of climate change, which requires predicting out of sample, we use a model that returns strictly positive predicted probabilities.

The binary listing variable is modeled as a function of taxon (specifically amphibian, bird, mammal, fish, fungi, invertebrate, mammal, plant, protist or reptile), the NatureServe conservation assessment status, standardized n-grams for both the scientific and common name of the species, and two different measures of taxonomic uniqueness. To ensure the assessed conservation status corresponds to the listing status at the time of assessment, we code species as “unlisted” if they have never been listed, were delisted before the year of the conservation assessment, or were listed more than 10 years after the assessment.

To capture the evolutionary uniqueness of a species, we first include the number of species within the genus. This is a very rough proxy for genetic distinctiveness (Supplementary Figure 1), but is available for almost all species. In a second regression we also report the more precise measure of evolutionary distinctiveness, which directly measures phylogenetic uniqueness of species but is only available for a subset of species. Because this genetic data is only available for birds, mammals, amphibians and reptiles, the second regression (“Few Taxa”) is limited to just those species.

Standard errors in all regressions are estimated using 250 clustered bootstraps, clustering at the family level (the taxonomic level above genus), which allows for correlation of the residuals between related species. There are 1933 families in the dataset and the median family has 5 species.

		Dependent variable: Listed	
		All Taxa, No Ev. Dist	Few Taxa, inc Ev. Dist
Taxa:	Amphibians	-0.607 (3.326)	-1.137 (5.059)
(dropped=Mammals)	Birds	0.419 (0.393)	-0.450 (0.373)
	Fish	0.057 (0.330)	
	Fungi	-15.615*** (0.552)	
	Invertebrates	-2.441*** (0.475)	
	Plants	-0.951*** (0.311)	
	Protists	-15.524*** (0.707)	
	Reptiles	0.066 (0.443)	-0.466 (0.494)
Conservation Status:	Imperiled	-1.363*** (0.125)	-0.335 (0.318)
(dropped=Critically	Vulnerable	-2.985*** (0.167)	-1.923*** (0.483)
Imperiled)	Apparently Secure	-4.557*** (0.361)	-3.093** (1.227)
	Secure	-7.201 (5.179)	-5.613 (6.159)
	Extinct	-2.106*** (0.545)	-0.206
	Probably Extinct	-1.264*** (0.267)	0.435 (4.127)
	Unknown	-6.003*** (0.344)	-3.189 (4.378)
Other Covariates:	Common N-gram	0.642 (0.436)	0.447 (0.579)
	Scientific N-gram	0.021 (0.219)	0.059 (0.192)
	Genus Size (logged)	-0.174*** (0.046)	-0.262 (0.161)
	Evol. Dist. (logged)		0.020 (0.275)
	Observations	53,688	1,501
	Log Likelihood	-3,158.321	-241.391
	Akaike Inf. Crit.	6,354.641	512.782
	Note:		*p$<$0.1; **p$<$0.05; ***p$<$0.01

Table 2: Results of Listing Decision Regression. Dependent variable is a binary variable indicating whether the species is listed under the ESA or not at the time of conservation assessment, and is modeled using a logistic regression. Status gives the conservation status of the species as assessed by NatureServe, with lower numbers corresponding to higher endangerment. Standard errors are block bootstrapped, clustering at the family level.

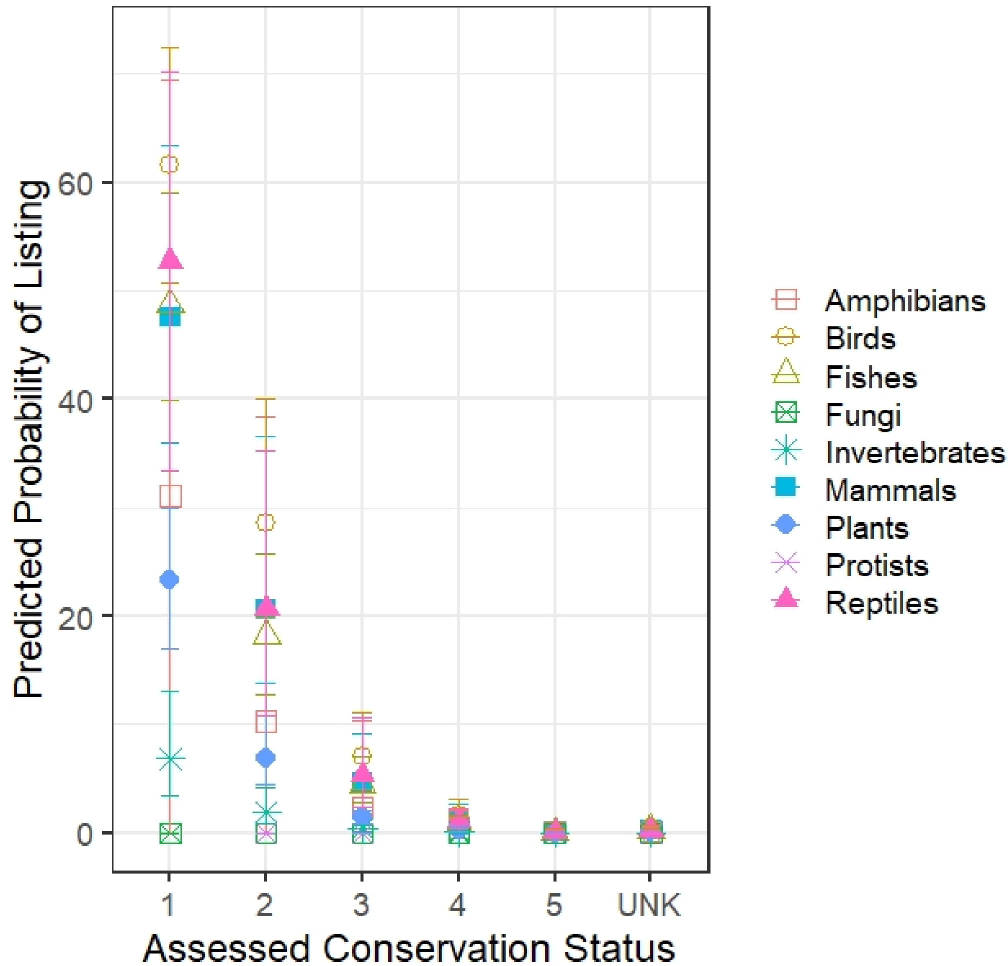

Figure 2: Probability of a species appearing on the ESA list at the time of conservation assessment, implied by coefficients in Table 2 (Column 1) for each taxonomic group and NatureServe conservation rank. Error bars give the 95% confidence interval. Probabilities are estimated at the median value of model covariates (n-gram value and genera size) for each taxon by status combination.

Results are shown in the first column of Table 2. Like Metrick and Weitzman (1996, 1998), we find strong evidence that the probability of listing changes with conservation status. The probability of a species appearing on the ESA list at the time of assessment decreases monotonically from the most (status=1, critically imperiled) to least (status=5, secure) endangered. Coefficients in Table 2 give the difference in listing probability compared to critically imperiled status. Table S1 gives the difference in regression coefficients between different conservation status levels and shows these differences are almost all significantly different from each other at the 5% level.

However, also like Metrick and Weitzman, we find evidence that factors associated with species' utility value to humans affect listing. Although differences in listing probability do not differ significantly between vertebrate species (i.e. mammals, birds, fish, amphibians and reptiles), plants and invertebrates are much less likely to be listed. However, we do not see an effect of either scientific or common name n-grams on listing probability.

The two variables related to the distinctiveness of a species show an interesting pattern. In the full model including all taxa, species in smaller genera are more likely to be listed. Increasing the number of species in the genus by 10% decreases the probability of listing by 0.42 percentage points at the population mean, roughly a 16% decrease relative to the average listing probability of 2.5%. This effect is largely driven by plants and invertebrates, which have both larger and more variable genera size. Column 2 however, shows there is no evidence that evolutionary distinctiveness – the unique evolutionary history content contained in the "library" of a particular species (Metrick and Weitzman 1998) – plays a role in the listing decision. Because this genetic data is only available for birds, mammals, amphibians and reptiles, the regression in the second column of Table 2 ("Few Taxa") is limited to just those taxa groups.

Figure 2 converts the coefficients for the full model in Table 1 into a probability of listing for each taxonomic group and NatureServe conservation status rank. It shows the strong preference for vertebrate species, particularly mammals, over plants or invertebrates apparent in the listing decision, as well as the steep decline in listing probability with improvements in the assessed conservation status.

2. The Spending Decision

The logarithm of spending by USFWS on a particular species in a particular year is regressed on a set of variables that capture the taxonomic group, the listing status (specifically endangered, threatened, or extinct/probably extinct), the spatial extent of the species (both its range and the number of USFWS regions in which it occurs), the notability of the species (measured by the n-gram 5-year rolling mean), the number of species in the genus, number of years since listing, and a set of variables capturing USFWS' own prioritization. The USFWS covariates include the three components of the priority number (degree of threat, species taxonomic rarity, and recovery potential) and an indicator variable capturing USFWS' assessment of whether conservation of the species conflicts with economic development. Standard errors are clustered at the family level in all regressions.

The inclusion of fixed-effects at the species level helps control for unobserved factors that may affect spending, but limits the set of variables that can be identified. We navigate this trade-off by presenting two models. A correlated random effects model ("full model") estimates the effect of all variables, including time-invariant species-level characteristics. Species-level unobservables are modeled through a random species effect, nested within families to allow correlation between related species. The fixed-effect model ("SpeciesFE") removes all time-invariant variation between species. This potentially improves confidence in the causal effects estimated, but limits the effects that can be estimated only to those that change over time within a species, including USFWS prioritization, ESA listing status, and n-grams. Both models include year fixed-effects that control for common variation over time.

Table 3 gives results of the spending regressions. We find evidence for a range of factors influencing patterns of spending. Firstly, vertebrates receive, on average, more funding than plants or invertebrates, though there is no significant difference among vertebrate groups. Secondly, there is no indication that listing status (i.e. threatened vs endangered) influences spending. It is possible that spending might influence conservation status although several prior studies have look at the relationship between cumulative lagged spending and species recovery, as measured in the biennial species recovery reports with mixed results (Kerkvliet and Langpap 2007; Miller et al. 2002; Gibbs and Currie 2012).

Like previous papers (Metrick and Weitzman 1996; Langpap and Kerkvliet 2010), we find that the "conflict" indicator variable is both large and statistically significant in the full model, meaning that species where conservation has been identified as being in conflict with economic development tend to receive significantly more spending. Unlike much previous work, we also find effects of other elements of USFWS prioritization on spending (c.f. Simon et al. 1995). The USFWS ranks the importance of the priority number criteria in the order threat, then recovery potential, then rarity and we see some evidence of that in our estimates. The effect of a 1 unit change in threat level (on a 3 point scale) is around a 15% increase in spending (statistically significant in both models). For recovery potential that falls to around 10% (significant in the full model but not the species fixed-effect model) and to a statistically-insignificant 3% for species rarity.

Several variables new to this analysis have both large and statistically significant effects on spending. Widely distributed species, measured both by range area and the number of USFWS regions in which

		Dependent variable: Annual Spending (logged)	
		Correlated Random Effects Model	*Ordinary Least Squares*
		Full Model	Species FEs
Taxon:	Birds	0.858 (0.522)	
(dropped=Amphibians)	Fish	-0.274 (0.531)	
	Invertebrates	-1.685*** (0.501)	
	Mammals	0.254 (0.534)	
	Plants	-1.841*** (0.491)	
	Reptiles	-0.413 (0.598)	
Status:	Threatened	0.044 (0.083)	0.108 (0.180)
(dropped=Endangered)	Extinct	-1.008*** (0.206)	
Priority Number:	Threat	0.161*** (0.038)	0.145** (0.068)
	Rarity	0.047 (0.061)	0.034 (0.163)
	Potential	0.092** (0.043)	0.115 (0.071)
	Conflict	0.265*** (0.048)	0.155 (0.105)
Geographic Factors:	FWS Regions	0.792*** (0.078)	
	Range Area (logged)	0.148*** (0.017)	
Other Covariates:	Scientific N-gram	0.087** (0.042)	-0.028 (0.064)
	Common N-gram	0.067** (0.033)	0.123** (0.048)
	Years Listed (logged)	-0.016 (0.024)	-0.029 (0.075)
	Genus Size (logged)	-0.043 (0.027)	
Random Effects		Species Nested in Family	NA
Fixed Effects		Year	Species, Year
	Observations	20,011	22,995
	R^2 (full model)		0.724
	R^2(projected model)		0.002
	Log Likelihood	-33,734.490	
	Akaike Inf. Crit.	67,558.980	
	Note:		*p<0.1; **p<0.05; ***p<0.01

Table 3: Results of Spending Decision Regression: Dependent variable is logged species by year spending. Spending data spans 1993-2016. The R^2 of the projected model gives the fraction of variance explained by the regressors after removing the fixed effects. Standard errors are clustered at the family level.

the species occurs are associated with higher spending. We also find an association between species' prominence in English-language books and spending. These effects are empirically small though – typical within-species variation in the scientific n-gram would lead to changes in spending on the order of 0.5%, or about \$200 for the median species. We do not find any systematic effect of years since listing or phylogenetic distinctiveness (i.e. genus size).

Using the full models, we calculate an expected present cost of 100 years of managing listed species under the ESA. The cost (C) of managing a listed species in taxonomic group g is calculated via simulations from the full random effects model as follows:

$$(1) \qquad C_g = \sum_{t=0}^{100} \frac{1}{N_g} \sum_{n \in g} \left(e^{\left(\hat{\beta}_0 + \hat{\beta}_g + \hat{\boldsymbol{\beta}} \overline{\boldsymbol{X}}_g + \hat{\beta}_t \log(t) + \gamma_n + \varepsilon\right)} \right) * DF_t$$

Where the β terms are draws from the multi-variate distribution of model coefficients, $\overline{\boldsymbol{X}}_g$ is the group-specific average of model covariates, γ_n is the estimated random effect for species n in group g, N_g is the total number of species in taxonomic group g, and ε is a draw from the residual distribution. The mean and distribution of C_g is calculated based on 1000 samples of the estimated parameter and residual distribution.

DF_t is the discount factor in year t, calculated using the declining discount rate schedule given in Weitzman (2001). There are a number of reasons why a declining discount rate may be appropriate for long-term projects such as species preservation (Arrow et al., 2013; Arrow et al., 2014). In "Gamma Discounting", Weitzman points out that under uncertainty, the relative importance of the lower region of the discount rate distribution grows over time, an effect that can be approximated by a declining discount rate. The schedule he provides, calibrated to a survey of 2,160 economists, starts at 4% per year and declines steadily to 1% after 75 years. To account for the possibility of future delisting, we estimate the average annual probability of delisting from our sample at 0.09% (resulting from 24 recoveries and 7 extinctions) and add this to the annual discount rate.

Results are given in Table 4. It is important to note that these costs represent only direct public spending on listed species and assume stability of policy priorities and public preferences into the future. There are also the opportunity costs of restrictions on land use and other activities that accompanies listings that are not bounded here, but could be large. These are discussed further in Section 3.2.

	Central Estimate	Lower Bound (2.5%)	Upper Bound (97.5%)
Amphibians	39.5	32.2	50.2
Birds	107.6	99.3	117.6
Fish	57.9	52.6	64.7
Invertebrates	8.4	7.9	9.1
Mammals	74.1	68.3	80.6
Plants	3.5	3.3	3.6
Reptiles	61.1	52.0	72.8

Table 4: Present costs of direct spending on listed species for 100 years following listing (million $) with 95% confidence interval

What, in sum, do our findings reveal about the listing and spending decisions under the ESA compared to Metrick and Weitzman (1998, 1996)? Most significantly, while the previous papers showed some evidence for preferences in both listing and spending for some vertebrate groups over others, we do not find evidence for prioritization among vertebrate groups, but do find large preferences for vertebrate species over the much larger groups of plants and invertebrates that were not considered in previous papers. We also find associations between USFWS prioritization and spending, particularly the large effect of the "economic conflict" variable, but also an important role for the "degree of threat" indicator, the most important factor in the lexicographic prioritization scheme used by USFWS. We also document an important role of species distribution, both absolute range area and the number of FWS regions it covers in determining spending. Finally, the null findings are also notable. We do not find an effect of species n-grams on listing probability and only a very small effect on spending. We also do not see evidence that species representing particularly distinctive evolutionary histories – the most unique of the genetic libraries described by Weitzman in the Noah's Ark Problem (1998) – are prioritized for ESA listing.

Section 3: Change in Direct ESA Expenditures Implied by Climate Change

Climate change will affect species everywhere and, for some subset of species, will pose an existential threat. Rising temperatures could therefore imply a growing fiscal and regulatory burden for species protections in the US. The magnitude of this burden has not previously been constrained, but the results shown in Tables 2 and 3 can be used for a rough calculation of climate change costs, assuming priorities for species protection persist into the future. This assumption is large, but can be at least partly justified by the observation that analyses of ESA spending data conducted almost 25 years apart reveal similar patterns, such as higher spending on species in conflict with economic development (Metrick and Weitzman 1998, 1996).

There is now a large literature in ecology estimating the number of species at risk of extinction at different levels of warming. The vast majority of this literature uses the climatic range of the current distribution of species and examines how this area will expand or shrink with warming. Some studies incorporate limits on the rate of species movement, but they typically exclude factors such as species interactions or landscape dispersal barriers, or ecologically important but species-specific factors such as breeding grounds, all of which may modulate the impact of climate change (Urban 2015). Species that lose more than a threshold area of their range are projected to be at risk of extinction. A 2015 meta-analysis of 131 published studies showed that species at risk of extinction increases from approximately 3% at one degree of warming to almost 20% at 5 degrees (Urban 2015).

For this paper, we re-estimated the temperature-extinction relationship using the subset of 191 estimates from 32 separate studies focused on North America in the Urban (2015) dataset. Each estimate is one result from an ecological study giving the fraction of species estimated to be at risk of extinction at a particular level of warming. A single paper may report multiple estimates that differ in terms of the level of warming, the set of species considered (e.g. birds vs mammals vs invertebrates), the dispersal model used (assumptions about how species might move as the climate shifts), and the threshold range contraction defining when a species is defined as at risk of extinction. Within the 191 estimates, 98 (51%) assume universal dispersion (i.e. no geographic or speed constraints on movement to a new area), 25 (13%) assume some species-specific rate of movement limiting dispersion, 21 (11%) assume species can only move into areas contiguous to their current range, and the remaining 47 (25%) assume no species movement.

Using the North American studies, we estimate the regression:

$$Ext = \beta_1 \Delta T + \beta_2 \Delta T * Threshold + \varepsilon$$

Where Ext is the proportion of species estimated to be at risk of extinction and ΔT is the rise in temperature since the pre-industrial period. *Threshold* is the proportion of range area lost before the species is “counted” as functionally extinct – many studies use a 100% threshold (i.e the climatic range ceases to exist), but others vary between 80% and 100%. We allow this choice to modify the temperature-extinction relationship. Standard errors are clustered at the author level, allowing for correlation between estimates from the same study, as well as between different studies by the same author. The regression is weighted by the number of species each study examined to estimate the fraction of species at risk of extinction.

Results are given in Table 5 and show the expected increase in extinction risk with warming, as well as the effect of the threshold variable modifying this relationship in the expected direction. We test, but do not find evidence for, both a quadratic effect of temperature change (p=0.71) as well as heterogenous effects of warming on plants, vertebrates, and invertebrates (a Wald test of the joint significance of taxa-temperature interaction terms fails to reject the null hypothesis, p=0.29).

	Parameter	**P-Value**
ΔT	0.380	0.004
$\Delta T * Threshold$	-0.377	0.005

Table 5: Results of regression of fraction of species at risk of extinction on warming level

Supplementary Figure 2 displays the estimated effect and uncertainty range for a threshold value of 90%. We conservatively chose a 90% value to represent the fact that species are of conservation concern long before they are on the brink of extinction; the IUCN criteria for critically-endangered, for example, includes a reduction in population greater than or equal to 80%. We find the fraction of species at risk of extinction in North America increases from 4% at one degree of warming to 20% at 5 degrees, a close quantitative match to the global average values reported by Urban (2015).

Combining the expected effect of warming on extinction risk, the probability of listing conditional on conservation status, and the average level of direct spending conditional on listing, it is possible to generate a rough, ballpark estimate the fiscal burden in terms of direct spending on the ESA implied by climate change, assuming future public preferences over endangered species protection resemble those of the last 25 years. There are two main challenges in making this estimate. Firstly, the mapping from "risk of extinction" as defined in ecological studies of climate impacts, to the NatureServe conservation status is unclear. We proceed under the working assumption that the species threatened with extinction by climate change will have a conservation assessment of 1 ("critically imperiled"). This means that, by construction climate change has no effect on the probability of listing for species already critically imperiled. Range contractions caused by climate change for these already-imperiled species might substantially increase the probability of listing. It is also possible though, in theory, that climate change could lead to range expansions for some of these species, lowering the probability of listing. In either case, we are not able to resolve this effect given the coarse nature of conservation status information for non-listed species.

A second ambiguity concerns the future status of the large number of species with unknown status (61% of the NatureServe dataset). The vast majority (98%) of these species are plants, invertebrates or fungi. Given the demonstrated lack of attention and scientific knowledge about these species, it is unclear whether any existential threat posed by climate change would be sufficient to propel them onto the Endangered Species List. Accordingly, our preferred results err conservatively by including only the set of species with a current conservation assessment as potentially-listable, though we also present results that include the full set of species for comparison. Details on the simulations used to estimate these effects is given in the Supplementary Appendix.

Figure 3a shows the expected increase in the number of critically-imperiled species due to climate change, as well as the subset of those that would be expected to be listed, excluding any species where status is currently unknown. The linear increase with warming follows from the estimated linear effect of warming on extinction risk (Supplementary Figure 2). The strong selection at the listing stage is also apparent – on average less than 20% of the species critically imperiled by climate change would be listed. The relative representation of different taxonomic categories among listed species depends both on their absolute prevalence (there are far more invertebrates and plants than mammals or birds) as well as the differential probability of listing (invertebrates are far less likely to be listed for instance, Figure 2). A warming of 2° would increase the number of species listed by 280 in expectation, increasing to 690 at 5° of warming. An additional 1100 (2°) to 2800 (5°) species, mostly plants and invertebrates, will be critically imperiled but remain unlisted. Including the additional 38,430 species with unknown conservation status increases the expected number of additional listings to 610 and 1540 under 2° and 5° respectively. Depending on greenhouse gas emissions over the 21st century, 2° might be reached any time between 2050 and 2100 and 5° might be reached by 2100 under a high emissions trajectory (Collins et al. 2013).

Figure 3b gives the total expected increase in protection costs with warming, both including and excluding species with unknown conservation status. Error bars give the 95% confidence interval accounting for four sources of uncertainty: the effect of warming on extinction risk, which species are affected by climate change, uncertainty in the probability of listing, and uncertainty in spending, conditional on listing. Our preferred estimate that excludes species with currently unknown conservation status implies a steady increase in committed direct ESA spending with warming, reaching

\$4.3 billion at 2° and \$21.2 billion at 5°. Although including unknown species into the analysis more than doubles the expected number of listed species, spending is only ~40% greater because the majority of these species are plants and invertebrates that receive relatively few resources (Table 3).

How large are these values? Certainly not large compared either to the total costs of climate change or the discretionary spending of the US federal government. But relative to historic spending on endangered species, these represent a substantial additional burden. For the 1613 listed species that appear in our dataset, the present cost of 100 years of spending totals just under \$40 billion (based on the taxa averages given in Table 2). Unmitigated climate change therefore represents an increase of somewhere between 50 and 75% of all the resources committed to direct spending on endangered species protection since the beginning of the ESA in the late 1960s. This ignores any effect of climate change on the listing of species that are already highly imperiled, any interactions of climate change with other threats to species such as habitat destruction or invasive species, or the possibility that climate change might increase average spending on listed species. For these reasons, the estimate could be considered a lower bound.

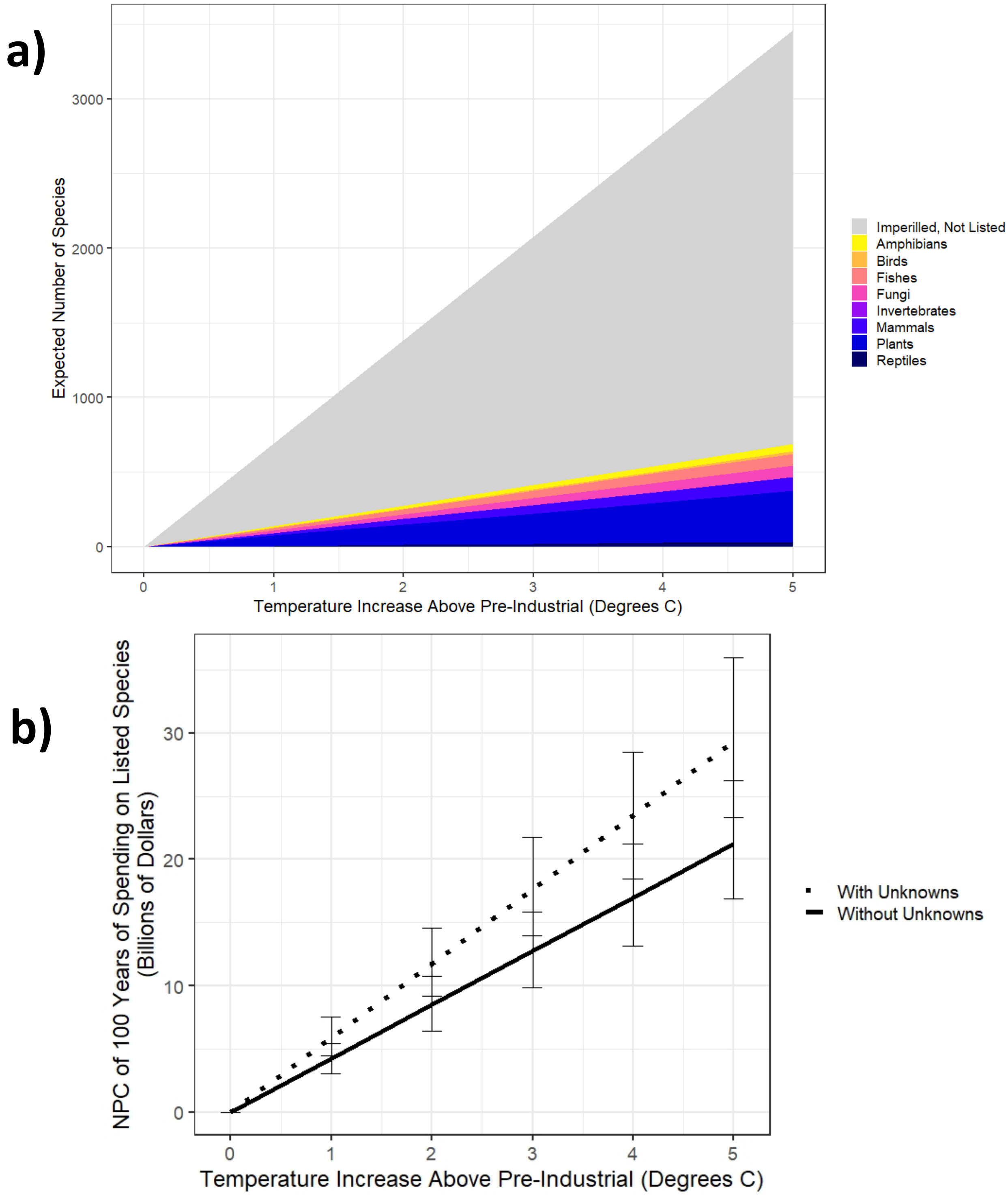

Figure 3: a) Expected increase in number of species that are critically imperiled due to climate change with warming, as well as the expected sub-set that would be listed (in colors). Includes only species with currently known conservation status as potentially listable b) Net present cost of committed direct spending implied by climate-change-induced listings, both including and excluding the set of species with unknown conservation status. Error bars show the 95% confidence interval based on 1000 simulations.

Other Listing Costs

Direct conservation expenditures are only one part of the social cost of species conservation. The ESA protects listed species in two main ways. Firstly, the agency designates critical habitat, which requires federal agencies to consult with USFWS before taking decisions that could threaten listed species (Section 7; Langpap et al., 2018). In some states, critical habitat designation also triggers additional state and local review of land use permitting (Auffhammer et al. 2020). Not all species have critical habitat designated - as of 2020 it applied to about 850 out of over 1600 listed species. Secondly, the "no take" provision of the ESA (Section 9) prohibits any action that harms endangered species, interpreted to include indirect harm through habitat modification (Langpap et al. 2018). This section, which applies to all listed species, can directly limit private land development and otherwise restrict economic activity.

The total costs of these provisions as a whole is not well established (Plantinga et al. 2014). It is clear that for some species, no take and critical habitat provisions impose large costs on particular land owners or sectors (Auffhammer et al. 2020; Frank 2020; Sunding and Terhorst 2014). Melstrom (2021) estimates negative effects on the order of 4% from critical habitat listing on the profits and value of dryland agriculture, with no effects on irrigated farmland. Other have shown that the aggregate welfare consequences of these restrictions can depend sensitively on the specifics of land, labor, and product markets, as well as the interaction with other local or state restrictions on land use (Sunding and Terhorst 2014; Quigley and Swoboda 2007; Murray and Wear 1998). The ESA contains very few provisions that allow for cost-effective management of species protection (Plantinga et al. 2014) and there is evidence both for highly heterogeneous protection costs among landowners (Sunding and Terhorst 2014) and steeply increasing marginal costs at high levels of protection (Langpap et al. 2018). Taken together, these suggest that the costs from no-take and critical habitat constraints, compared to more cost-effective management, may be substantial, at least in certain contexts.

The threat that climate change poses to biodiversity therefore has implications not just for direct spending on conservation, but also for the opportunity costs that protection of endangered species requires. These costs are at present unquantifiable, but should be understood as increasing the direct costs shown in Figure 3b by an unknown but potentially large multiplier. The total adaptation costs for species protection in this second- or even third-best regulatory setting therefore come from 1) the increase in direct conservation spending estimated in the previous section, 2) the welfare costs of raising these funds from distortionary taxes and 3) welfare loss from constraints on economic activity required for the protection of species that would not have been endangered in the absence of climate change.

The last category may well be the largest, but given the limited evidence on the costs of ESA restrictions, it cannot currently be estimated. Instead it joins the long list of “known unknowns” in the enumeration of climate change costs (Pindyck 2013).

Section 4: The Benefits of Species Protection and Residual Climate Change Damages

Interpreting the increase in listing and associated social costs of species protection with climate change as adaptive requires that the benefits of listing protections for these species exceed the social costs. A precise determination of net benefits of ESA species listing is impossible for a number of reasons, not least that the total social cost of listing is unknown, as discussed in the previous section. Here instead we attempt to roughly bound this question through a review of stated-preference estimates (i.e., contingent valuation and choice experiments) of household-level willingness to pay (WTP) for species protection.

We identify 48 estimates from 28 studies for 36 different listed species (details for each estimate in Supplementary Table 2). These 36 species are not a random sample of listed species. Mammals, birds, and fish are vastly over-represented compared to plants (1), invertebrates (3), and amphibians (0). Moreover, there is a focus on well-known species such as the gray wolf or sea turtles and on species with high-profile conflicts around ESA protections, specifically the red-cockaded woodpecker and Northern spotted-owl. The studies consider the value of a number of different outcomes related to species conservation, which may not be mutually exclusive, from habitat conservation (13/48 species), including critical habitat designation (5/48), to population changes, including avoided losses (17/48) as well as population gains (26/48). The studies quantify the value of these outcomes via contingent valuation (30/48) or choice experiments (18/48), using household WTP as the metric of choice.

Studies report WTP in terms of either a one-time payment (6 estimates) or an annual payment (42 estimates, details in Supplementary Table 2). Given the preponderance of annual WTP estimates, we compare annual aggregate WTP values to average annual ESA expenditures for each species. For studies reporting a one-time payment, we convert these amounts into equivalent annual payments assuming a 20 year time horizon and a 3% discount rate. Although these one-time payment studies do not always

Species	References	Avg Annual ESA expenditures (Millions of 2017$s)	Avg Annual Agg. WTP (Millions of 2017$s) Full Population	Avg Annual Agg. WTP (Millions of 2017$s) Restricted Population	Avg Ratio WTP/ ESA Expenditures Full Population	Avg Ratio WTP/ ESA Expenditures Restricted Population
Birds						
Bald eagle	Boyle & Bishop (1987), Stevens et al. (1991)	$17.7	$6,073	$2,268	343.2	128.2
Northern spotted owl	Rubin et al. (1991), Hagan et al. (1992)	$15.9	$18,216	$1,347	1146.2	84.7
Whooping crane	Bowker & Stoll (1988)	$6.2	$11,958	$3,338	1934.3	540.0
Mexican spotted owl	Loomis & Ekstrand (1997), Giraud et al. (1999)	$4.2	$8,726	$834	2054.2	196.3
Fishes						
Chinook salmon	Olsen et al. (1991), Loomis (1996), Wallmo & Lew (2012)	$189.3	$4,634	$141	24.5	0.7
Southern CA steelhead	Wallmo & Lew (2016)	$188.5	$5,628	$518	29.9	2.7
Coho salmon	Olsen et al. (1991), Bell et al. (2003), Wallmo & Lew (2016), Lewis et al. (2019)	$44.9	$466	$236	10.4	5.3
Sockeye salmon	Olsen et al. (1991)	$32.4	$581	$373	18.0	11.5
Chum salmon	Loomis (1996)	$21.2	$5,119	$945	241.6	44.6
Atlantic salmon	Stevens et al. (1991)	$10.2	$377	$13	37.1	1.3
Silvery minnow	Berrens et al. (1996)	$10.1	$638	$42	63.3	4.2
Colorado squawfish	Cummings et al. (1994)	$9.1	$481	$169	52.7	18.6
Shortnose sturgeon	Kotchen & Reiling (2000)	$2.4	$931	$78	381.1	32.0
Smalltooth sawfish	Wallmo & Lew (2012)	$0.9	$5,799	$225	6476.6	250.9
Invertebrates						
Black abalone	Wallmo & Lew (2016)	$1.4	$5,588	$736	3922.5	516.6
Elkhorn coral	Wallmo & Lew (2016)	$1.1	$5,441	$31	4891.8	28.3
Riverside fairy shrimp	Stanley (2005)	$0.5	$524	$23	1135.1	50.7
Mammals						
Steller sea lion	Giraud et al. (2002), Lew (2010)	$30.4	$9,232	-$130	303.7	-4.3
North Atlantic right whale	Wallmo & Lew (2012)	$9.9	$7,591	$209	763.0	21.0
Manatee	Solomon et al. (2004)	$9.7	$670	$103	68.9	10.6
Gray wolf	Chambers & Whitehead (2003)	$8.4	$647	$270	76.7	32.1
Northern Rocky Mountain wolf	Duffield (1991)	$8.4	$6,986	$581	829.3	68.9
Peregrine falcon	Kotchen & Reiling (2000)	$6.8	$436	$2,587	64.5	382.5
Hawaiian monk seal	Samples & Hollyer (1990), Wallmo & Lew (2012)	$3.4	$5,067	$906	1471.3	263.0
Humpback whale	Wallmo & Lew (2016)	$3.2	$14,917	$2,710	4705.6	854.8
Southern resident killer whale	Wallmo & Lew (2016)	$2.4	$6,882	$234	2812.4	95.7
Cook Inlet Beluga Whale	Lew (2019)	$2.4	$5	$5	2.2	2.2
Blue whale	Hageman (1985)	$1.0	$4,056	$820	4039.3	817.0
North Pacific right whale	Wallmo & Lew (2012)	$1.0	$7,899	$150	8033.9	152.7
California sea otters	Hageman (1985)	$0.9	$573	$531	671.1	622.0
Bighorn sheep	King et al. (1988)	$0.4	$524	$2	1188.6	3.9
Gray whale	Hageman (1985), Loomis & Larson (1994)	$0.3	$441	$107	1299.6	313.9
Reptiles						
Loggerhead sea turtle	Whitehead (1992), Wallmo & Lew (2012)	$8.3	$4,045	$614	489.1	74.2
Leatherback sea turtle	Wallmo & Lew (2012)	$6.1	$7,283	$814	1201.1	134.3
Hawksbill sea turtle	Wallmo & Lew (2016)	$4.0	$9,994	$1,426	2469.2	352.3
Plants						
Johnson's seagrass	Wallmo & Lew (2016)	$1.3	$6,597	$6,051.6	5028.4	4612.4
Average for all birds		$11.0	$11,244	$1,947	1369.5	237.3
Average for all fishes		$50.9	$2,465	$274	733.5	37.2
Average for all invertebrates		$1.0	$3,851	$264	3316.5	198.5
Average for all mammals		$5.9	$4,395	$606	1233.2	242.4
Average for all reptiles		$6.1	$7,108	$951	1386.5	186.9
Average for all plants		$1.3	$6,597	$6,052	5028.4	4612.4

Table 6: Comparison of aggregate WTP for conservation of 36 listed species from stated-preference estimates with average annual direct ESA spending over the 1993-2016 period. Specifics of each estimate are given in Supplementary Table 2. Most studies report annual WTP. Six estimates are one time payments, which are converted into an annual equivalent amount assuming a 3% discount rate and 20 year time horizon. Aggregate WTP values depend on the population the values are aggregated over and so two population-definitions are reported: one based on the study sample area ("Full Population") and one based on a restricted population limited to the species range areas ("Restricted Population")

specify a time horizon, those that do tend to reference periods of between 10 and 20 year. For studies with annual WTP estimates, this approach assumes preference stability for species over time. The relevant literature suggests reasonably stable preferences over the medium-run, with less stability over longer time frames (Skourtos et al. 2010).

Total aggregated WTP values are sensitive to the population of households considered as holding existence values. In our preferred estimate, we assign the same WTP value across all households considered in the original study (e.g. the whole US for nationwide samples, individual states for studies conducted only with local or state residents, "Full Population" in Table 6). IIn a sensitivity analysis ("Restricted Population" in Table 6), we also calculate WTP over a smaller population, limited to people living within the USFWS species range (for terrestrial species) or within coastal counties in coastal states adjacent to its range (for marine species). We compare this aggregate WTP to average annual ESA spending on each species over the 1993-2016 period in Table 6.

The final two columns in Table 6 show the ratio of direct ESA spending to aggregate WTP values for particular species. For the vast majority of species, WTP is between one and two orders of magnitude larger than direct spending levels. The major exception is the salmonids, which stand out for receiving a very large fraction of spending on endangered species (Table 1). Salmon are somewhat unusual among endangered species in that they have high recreational and commercial use values, as well as unique spiritual and cultural value to indigenous peoples of the Pacific Northwest, perhaps explaining the disproportionate level of spending they receive relative to the WTP measures in Table 6.

For the vast majority of species, aggregate WTP is at least an order of magnitude larger than direct spending levels, using the preferred population estimates ("Full Population" columns). Even a much more restricted definition of population gives ratios greater than 1 for ~90% of species and WTP to spending ratios on the order of 10 to 100 times for many species ("Restricted Population" columns") [1].

[1] Note that the negative values for the Steller sea lion in the restricted sample are taken from results in the sample of local residents who were faced with a situation in which the change in population being valued was related to

Given the paucity of evidence on the total social costs of listing, as well as the difficulty of comparing the exact good being valued in WTP estimates with listing protections, this should not be interpreted as proving the benefits of listing exceed the costs. But it does give some sense of how large the indirect costs of listing would have to be before exceeding WTP. The unmeasured opportunity costs of species protection would have to be between one and three orders of magnitude larger than the direct ESA spending in order to exceed WTP. Alternate assumptions around the distribution of property rights over extant species would imply the use of willingness to accept measures, which are often higher than WTP (Hanemann 1991; Tunçel and Hammitt 2014).

Total climate change costs arise both from the costs of adaptation and the climate damages remaining after adaptation (Cropper and Oates 1992). Given the historical evidence, it is not clear that all or even a majority of species threatened with extinction by climate change would be protected by listing, and it is possible that some of these species may go extinct, implying residual damages may be substantial. The welfare estimates in Table 6 cannot be extrapolated to estimate these costs as the set of species likely to be critically imperiled by climate change but remain unlisted are disproportionately plants and invertebrates, of which there are almost no WTP estimates. Other residual welfare losses that are even harder to bound come from declines in species abundance or the extinction of local populations without full species extinction, as well as the possibility that listed species may be at higher risk of extinction due to climate change.

Section 5: Conclusions

This paper has estimated how revealed public priorities for conservation spending will interact with a changing climate over the 21st century. We find these priorities appear to be fairly stable, at least over the medium term, as we document somewhat similar patterns to those identified by Metrick and Weitzman (1996, 1998) 25 years ago, particularly the prioritization of species that are more imperiled and in conflict with economic development. By expanding the analysis to include the large groups of plants and invertebrates, we also document a strong preference, in terms of both listing and spending, in favor of vertebrate species.

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the closure of local fishing grounds. This closure would be associated with negative income shocks for local residents, offering an explanation for the negative WTP values.

Based on these patterns, and the estimated effect of climate change on US biodiversity, we estimate unmitigated climate change could increase committed direct spending under the ESA by 50-75% by the end of the century. Further costs will arise from restrictions on land use associated with the expected 700 additional listings. Several thousand species, mostly plants and invertebrates, will be critically imperiled by climate change but remain unlisted. The lost existence value associated with these species is difficult to estimate: while our survey of the WTP literature suggests these values are large, at least compared to direct ESA spending, the vast majority of these studies focus on vertebrate species, so the generalizability to plant and invertebrate species is unclear. For the limited subset of species included in valuation studies though, we find WTP for conservation greatly exceeds direct ESA spending, meaning listings of these species are welfare improving unless the indirect costs of listing are orders of magnitude greater than the direct expenditures.

Weitzman published extensively on both the biodiversity and climate change problems. His work on climate change focuses on the role that very low probability but high consequence climate damages can play in driving cost-benefit results (Weitzman, 2009; Weitzman, 2012). In his writing on the issue, Weitzman emphasizes the importance of deep structural uncertainties in the costs of climate change, and urges researchers not to sideline these concerns in pursuit of a false sense of precision or objectivity:

> "*The economics of fat-tailed catastrophes raises difficult conceptual issues that cause the analysis to appear less scientifically conclusive and more contentiously subjective than what comes out of an empirical CBA of more usual thin-tailed situations. But if this is the way things are with fat tails, then this is the way things are, and it is an inconvenient truth to be lived with rather than a fact to be evaded just because it looks less scientifically objective in cost-benefit applications*." (Weitzman, 2009, p.18)

The welfare consequences of ecosystem disruption and decline caused by unmitigated climate change are highly uncertain and potentially large, with some evidence that WTP for species' existence may be fat tailed (see Conte and Kelly 2021 for a review). Difficulty of measurement, compounding uncertainties in the climate, ecological, and economic estimates, and structural uncertainty in how these goods enter into the utility function (Bastien-Olvera & Moore, 2021; Drupp & Hänsel, 2020; Sterner & Persson, 2008; Weitzman, 2012b) mean these costs are currently very poorly constrained in climate damage estimates.

While the implications for direct-spending commitments on conservation can be estimated, this only highlights the large unknowns in the ecological costs of climate change, particularly the opportunity costs associated with species protection and the lost welfare from residual ecological damages. The ESA case also serves as a reminder that public climate change adaptation more generally will occur within an existing framework of laws and regulations, many of which are inefficient and distortionary, or which interact in complex ways with pre-existing market distortions. Climate change will interact with and in some cases exacerbate these distortions meaning a full accounting of climate change costs should include the additional deadweight loss in these second-best settings.

Weitzman's writing on biodiversity came at a time of growing public concern over the global loss of species from deforestation and land-cover change, resulting in the Convention on Biological Diversity, signed in 1992 at the Rio Earth Summit at the same time as the Framework Convention on Climate Change. Global progress on both issues in the almost 30 years since has been decidedly mixed, with the result that the two will become increasingly intertwined over the future as climate change interacts with existing pressures from habitat loss, pollution, and invasive species to drive ecosystem change. Understanding the welfare costs of these changes poses large empirical and theoretical challenges. But this understanding is also essential if environmental economics is to better inform the collective social response to these problems.

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