Protecting Species through Legislation

Brei, Michael; Perez-Barahona, Agustin; Strobl, Eric

DOI:
10.1093/ajae/aaz025

Document Version
Peer reviewed version

Citation for published version (Harvard):
Brei, M, Perez-Barahona, A & Strobl, E 2020, 'Protecting Species through Legislation: The Case of Sea Turtles', *American Journal of Agricultural Economics*, vol. 102, no. 1, aaz025, pp. 300-328. https://doi.org/10.1093/ajae/aaz025

Protecting Species Through Legislation: The Case of Sea Turtles

Michael Brei
The University of the West Indies (Barbados) & Université Paris Ouest

Agustín Pérez-Barahona
THEMA, University of Cergy-Pontoise & École Polytechnique

Eric Strobl
University of Bern[a]

April 11, 2019

Abstract

We evaluate the effectiveness of legislation in reducing the negative impacts of beachfront lighting on sea turtle nesting activity, one of the main threats to the species. To this end we construct a time varying index of ordinance effectiveness across Florida counties and combine this with loggerhead nesting data to create a panel data set covering a 26 year period. Our econometric findings show that such legislation can significantly increase nesting activity, where current levels of protection result in an additional 34 per cent. Using our estimates within a calibrated population model we also demonstrate that legislation can reduce the time to the animals' extinction. Finally, we show that alternatively raising sea turtles in captivity under a head start program may be prohibitively expensive, especially when considering estimates of local willingness to pay for sea turtle preservation.

Keywords: species protection, legislation, sea turtles.
JEL: Q57

[a]Corresponding author at: Department of Economics and Oeschger Centre for Climate Change Research, University of Bern, Schanzeneckstrasse 1 Postfach 3001 Bern, Switzerland; e-mail: eastrobl13@gmail.com.

1 Introduction

Estimates suggest that up to 539 species have become extinct in the US over the past 200 years (Sedjo, 2008). Yet, while there had been a growing awareness of the extinction threat to a number of prominent species since the turn of the 19th century, prior to the Endangered Species Act in 1973 (ESA) no general protective legislation had been put in place.[1] The ESA potentially provides extensive protection for species listed, including protection of critical habitat, implementation of a recovery plan, restrictions on take and trade, authorization to make land purchase or exchanges for important habitat, and federal aid to State and Commonwealth conservation departments. As matter of fact, as of date over 2,400 species have made it on the ESA's endangered species list, and current annual expenditure on their conservation are over US$ 1.5 billion.[2] This begs the question as to how effective the ESA has been in terms of facilitating species recovery.

One has to recognize of course that in practise implementation of an effective species recovery plan is not necessarily straightforward since habitats are not always easily defined, threats are often multi-faceted, monitoring can be difficult, and implementation can be costly, both more generally and in terms of opportunity cost.[3,4,5] A representative case in point of these challenges, and the object of our study here, is the loggerhead sea turtle (*Caretta caretta*). More specifically, sea turtles are threatened by a number of factors, including entanglement in fishing gear, poaching and illegal trade of eggs, meat, and shells, ocean pollution, and coastal development[6], their population is widely believed to be decreasing at an alarming rate worldwide. A crucial part of the threat of coastal development to loggerheads is the presence of artificial lighting on their nesting beaches, where there has been considerable evidence that shows that artificial nighttime light de-

[1] Early calls for wildlife conservation emerged in the 1900s with the near extinction of the bison and the disappearance of the passenger pigeon. A number of specific legislation pieces followed, such as the Lacey Act of 1900, the Migratory Bird Conservation Act of 1929, and the Bald Eagle Protection Act of 1940. The first more comprehensive legislation passed was the Endangered Species Preservation Act of 1966, but while this act enabled the listing of native US animal species as endangered it provided very limited protection (see Roman, 2011).

[2] USFWS (2015).

[3] See, for instance, Boersma *et al.* (2001), Boyd (2014) and EF (2016). Kerkvliet and Langpap (2007) find no evidence of increased funding or habitat designation aiding species recovery. However, in a subsequent study, Langpap and Kerkvliet (2012) show that habitat conservation plans do have a significant impact on species recovery, as long as these are not multi-species plans.

[4] There is also some evidence that landowners preemptively destroy habitat to avoid potential land-use regulations; see, for example, Lueck and Michael (2003) and Zhang (2004).

[5] For instance, Borkovic and Nostbakken (2017) find that in Canada the oil leases that are regulated by species regulation lose 24% in value.

[6] See http://www.seeturtles.org/sea-turtles-threats/

ters sea turtle adults from nesting and disorients them (for instance, Witherington, 1992; and Johnson *et al.*, 1996). Moreover, artificial lighting also increases the mortality rate of sea turtle hatchlings because it interferes with hatchlings' ability to find their way from their nests on the beach to the sea (see, amongst others, Tuxbury and Salmon, 2005; and Lorne and Salmon, 2007). In this regard, Brei *et al.* (2016) find that coastal light pollution, through its effect on nesting activity, has substantially accelerated the potential extinction of sea turtles in the Caribbean.

In terms of legal protection loggerheads have been ESA listed and hence protected since 1978, with current annual expenditures of nearly US$ 9.5 million. Furthermore, in Florida, which hosts 90 per cent of nesting activity in the US, they enjoy additional protection under the 1995 Florida Marine Turtle Protection Act (MTPA).[7] These legislative pieces specifically prohibit, amongst other things, the "take" of loggerhead turtles, where "take" includes their harassment and harm. As confirmed by a ruling of a federal appellate court in 1998, artificial light on beaches during their nesting period falls under this definition of "take" and hence can be viewed as prohibited by both the ESA and the MTPA (Barshel *et al.*, 2014). Additionally, the Florida Department of Environmental Protection (DEP) has enacted rule 62B-55 F.A.C., setting forth a set of guidelines for local government regulations that control beachfront lighting to protect nesting females and hatching sea turtles.[8] However, importantly, the DEP sea turtle lighting rule does not require local governments to legally adopt the proposed guidelines. While presently most Florida coastal counties and municipalities have meanwhile adopted some form of beach lighting ordinances, their ordinances differ widely not only in terms of their legislative details but also in their effectiveness of implementation (Barshel *et al.*, 2014). Moreover, data on sea turtle disorientation collected by the U.S. Fish and Wildlife Service (FWS) since 1987 suggests that nesting turtle disorientation has remained constant.[9] This of course begs the question of how effective local legislation really is, particularly since there are other alternative strategies to aid recovery of the loggerhead population, such as captive rearing programs.

In this paper we specifically investigate the effectiveness of sea turtle lighting friendly (STFL) legislation in encouraging loggerhead sea turtle nesting in Florida, what impli-

[7] See USFWS (2015).

[8] Apart from loggerheads, there are four other sea turtle species in Florida, although their nesting activity is minimal compared to the former.

[9] Personal communication with Robbin Trindell, Florida's Imperiled Species Management Plan.

cations this has had for the total Florida sea turtle population, and what the monetary benefits of the implemented legislation have been. To this end we, in the spirit of Barshel *et al.* (2014), build a time-varying county level sea turtle friendly lighting ordinances index that takes the intricacies of the legislative pieces and their implications for sea turtle nesting into account. We combine this with annual local nesting activity and a rich set of controls across a sample of Florida coastal counties to create a 26 year panel data set, which we use to econometrically quantify the effectiveness of the ordinances. With this estimate in hand, we employ a calibrated population model for loggerheads to assess the effect of STFL ordinances on the evolution of their population over the long term. We find that the regulations can significantly delay loggerheads' extinction. Our framework also allows us to compute the replacement cost associated with the extinction of sea turtles and STFL ordinances.

There are already a number of studies that examine the effectiveness of species protection legislation in aiding species recovery.[10] The evidence in this regard is rather inconclusive. For instance, while Ferraro *et al.* (2007) show that unless the ESA listing is combined with substantial funds listed species are likely to further decline, they also find that listing followed by funding has a positive impact. Similarly, Kerkvliet and Langpap (2007) find no evidence of increased funding or habitat designation aiding species recovery, but a significantly negative impact on the probability that species are calssified as declining or extinct. While Gibbs and Currie (2012), in contrast, discover that the number of years listed, years with critical habitat designation, the amount of peer-reviewed scientific information, and funding did weakly increase recovery, these factors are only able to explain 13 per cent. Also, Langpap and Kerkvliet (2010) discover that spending increases the probability of being classified as stable and improving, and decreases the probability of being categorized as declining or extinct. Finally, in a follow up to their earlier study, Langpap and Kerkvliet (2012) discover some evidence that habitat conservation plans have a significant impact on species recovery, as long as these are not multi-species plans. Importantly, however, all of these earlier studies pool data on species as well as the legislation and recovery plans in their analysis. However, not only do species differ widely in the nature of their habitat and the threats thereto, but, as a perusal of current listed species shows, implemented recovery plans and legislation tend to be very species specific and intricate. The derived results are thus difficult to evaluate in terms of their effectiveness and their implications for policy. By examining a single

[10] See Langpap *et al.* (2018) for a comprehensive review

species and legislation in detail, as we do here, one is arguably able to infer much more precise findings and subsequent recommendations for species recovery. [11]

The remainder of the paper is organised as follows. Section 2 introduces the concept of STFL and the related regulations in Florida. We describe in Section 3 the construction of our database, including details regarding the construction of our measure of the effectiveness of STFL ordinances in Florida. Section 4 provides the econometric analysis of the paper. In Section 5 we study the effect of STFL ordinances on the loggerheads' population dynamics, and in Section 6 investigate the monetary cost of light pollution. Section 7 concludes.

2 Sea turtle friendly lighting (STFL)

Beaches are key for the survival of sea turtles. More precisely, although sea turtles spend only a very small proportion of their lifetime on beaches, these sites are fundamental to their reproductive phase, since females nest and hatchlings emerge on beaches. In addition, their nesting behaviour exhibits natal philopatry, *i.e.*, females are likely to only nest on their natal beach.[12]

Given that sea turtle nesting and the emergence of hatchlings occurs almost exclusively at night, artificial beach illumination can drastically disturb the normal nesting behaviour of adult females and hatchlings (see Raymond, 1984; Witherington and Martin, 1996; or Witherington and Frazer, 2003, amongst others). Importantly, adult turtles prefer to nest on unlit beaches. Moreover, nighttime illumination fosters direct human disturbance of the nesting activity, frequently resulting in the abandonment or improper completion of nesting.[13] Beach lighting interferes as well with female adults' ability to correctly interpret physical cues that allow them to return to the safety of the sea after nesting. This disorientation problem seems to be even more severe for the hatchlings (*e.g.*, Witherington and Martin, 1996), where the unnatural stimuli of the artificial illumination can disrupt hatchlings' instinctive sea-finding mechanisms, reducing their survival

[11] Arguably, our paper also contributes to the literature on the impact light polllution in general, in that we provide a framework with which to examine how legislation can be used to counter-act its effect, at least in terms of species.

[12] Sea turtles could in practise look for alternative nesting sites on neighbouring beaches if the original site is no longer suitable (Worth and Smith, 1976; Witherington and Martin, 1996). However, studies such as Brei *et al.* (2016) did not find no significant empirical evidence of this.

[13] Witherington and Martin (1996) also found that sea turtles discard their eggs in the ocean when they do not find an appropriate nesting beach.

probability due to exhaustion, dehydration, and predation (Bustard, 1967; Witherington and Martin, 1996).

Preventing nightlight pollution on nesting beaches thus arguably constitutes an important component of the preservation of sea turtles. As noted in the introduction, the Florida DEP explicitly recognizes this and has consequently set (non-mandatory) guidelines for local government regulations to control beachfront lighting. The main focus of these guidelines is the adoption of STFL. Moreover, it should be noted that the new lighting technologies can additionally minimize the need for human behaviour regulation, such as requiring residents to close their curtains or to turn off exterior lights during the nesting season.

The general principles of the DEP guidelines can be summarized as "Keep it Low, Keep it Long, and Keep it Shielded". "Keep it Low" refers to mounting the light fixtures low in order to minimize light trespass, using as well the lowest lumens output needed. Moreover, since sea turtles are very sensitive to the blue-light (short wavelength)[14], replacing the common blue lamps with long wavelength light sources (greater than 580 nm, *i.e.*, amber/red lamps) would significantly reduce sea turtles' disorientation. This additional principle is frequently known as "Keep it Long". Finally, it is also advised to fully shield ("Keep it Shielded") the lamps in order to eliminate point source light using, for instance, full cut-off fixtures (where no light is emitted above a 90-degree angle).

3 Data and summary statistics

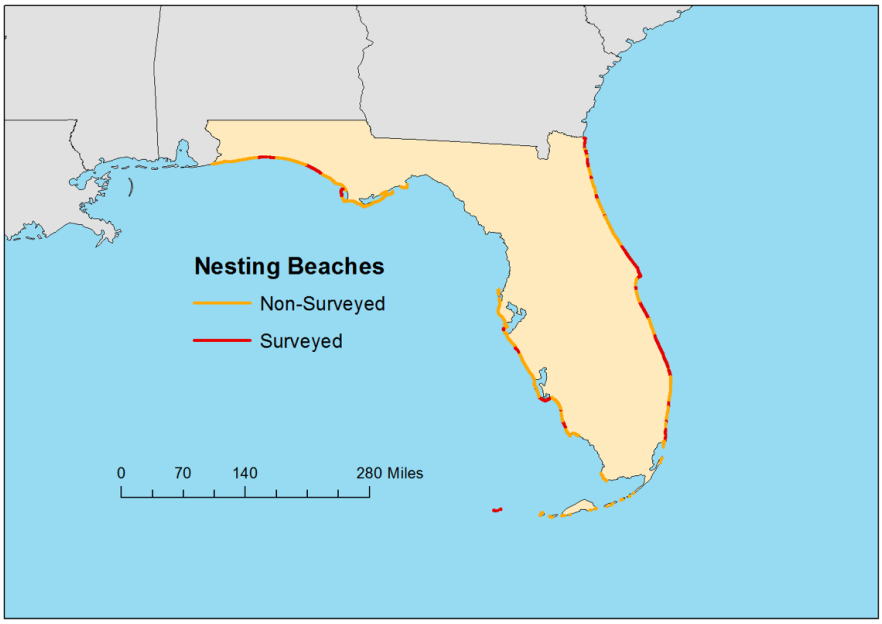
3.1 Loggerhead nesting data

There are currently two main loggerhead sea turtle nest-count surveys in Florida, namely the Statewide program and the Index program.[15] While the Statewide program is intended to be as complete as possible in terms of geographic coverage, it has not been consistent over time, adding beaches, changing boundaries, and changing the survey dates. In contrast, while not geographically exhaustive, the Index program has been constant in effort and coverage over time. More specifically, trained observers count and record nesting activity daily from the 15th of May to 31st of August, which represents most of the loggerhead nesting season. These data are used as the main source for statistical as-

[14]See, for instance, Witherington (1992).
[15]For a detail discussion about these surveys, see Witherington *et al.* (2009).

sessments of temporal trends in loggerhead nesting in Florida (Witherington *et al.* 2009). For our analysis here we hence rely on the nesting data from the beaches covered under the Index program for estimation purposes, but use the Statewide program beaches to make predictions regarding the population implications of the sea turtle lighting legislation on nesting activity for the wider Florida. The total number of nesting beaches for Florida in 2014 are depicted in Figure 1.[16] Of these 214 beaches, 33 are covered under the Index program, presented in red, and the rest under the Statewide program.

Figure 1: Nesting Beaches Surveyed under the Index Program



We use the annual Index nesting data for the period from 1989, its onset, until 2014, for the 25 years for which we have non-missing data.[17] Summary statistics of the number of nest counts of loggerheads for the Index beaches for our sample period are given in Table 1. As can be seen, the average annual number of nest counts per beach is 1,468, although with considerable variability.

[16]The shapefiles for the nesting beaches were obtained from the Florida Fish and Wildlife Conservation Commission.

[17]For the remaining 3 beaches, namely Siesta Key, Egmont Key, and Dry Tortugas, there were only a few data points and hence we dropped these.

Table 1: Descriptive Statistics

Variable	Mean	St.Dev.	Min.	Max.
NL	5.48	10.72	-0.11	66.11
NESTS	1468	3231	0	23712
SCORE	16	23	0	83
ROOMS [d=100m]	48	253	0	2490
ROOMS [d=200m]	212	927	0	7787
INCOME/CAP	40	12	15	80
NOURISHMENT	2	24	0	829
STORMS	0.63	1.52	0	12
DEMOCRATS	0.63	1.52	0	12

Notes: *NL* ≡ intensity of nightlights in 2014; *NEST* ≡ number of nests; *SCORE* ≡ legislation score; *ROOMS* ≡ number of rooms within *d* meters of the shoreline; *INCOME/CAP* ≡ county income per capita (2014 US dollars); *NOURISHMENT* ≡ average annual volume (cubic yards) of sand placed on nesting beaches; *STORMS* ≡ number of storms that affected a beach in a given year; *DEMOCRATS* ≡ share of registered democratic voters.

3.2 STFL ordinance measure

While the DEP enacted the legislative rule "Model Lightning Ordinance for Marine Turtle Protection" in 1993, it did not make it mandatory for local governments to adopt the model. Rather sea turtle friendly lighting has been regulated at the county and/or municipality level only. In order to identify all ordinances relevant to sea turtle friendly lighting we started with the list of ordinances compiled by the Florida Fish and Wild Conservation Commission (FWC)[18], but completed these with an exhaustive search of county and municipality level legislation.[19] This resulted in a total of 92 coastal ordinances on sea turtles nesting activity, including their implementation date. The first of these was adopted in 1986.

Important for our analysis, following the STFL principles outlined in Section 2, Barshel *et al.* (2014) introduced a method to measure the strength of local ordinances regarding the regulation of beach light pollution. It is based on an approach called Con-

[18]http://myfwc.com/conservation/you-conserve/lighting/ordinances/
[19]County and Municipality legislation can be found on library.municode.com

tent Analysis, which aims at systematically quantifying the information included in texts such as media messages or legal documents (*e.g.*, Krippendorff, 2013). To this end the authors evaluate to what extent an ordinance sets appropriate beach lighting conditions for turtle nesting on two fronts. Firstly, they asses the photic habitat conditions provided by the legislation using 17 statements termed the STFL Principles Component. Secondly, they consider whether the ordinance includes appropriate legal devices in order to ensure the implementation of these conditions in terms of a further 9 statements, labeled Implementation Component. For comprehensive lists of both sets of statements see Appendix A. For instance, the statement "Exterior artificial light for existing development must be long wavelength (*i.e.*, 580 nm or greater)" (item 5 of STFL Principles Component) objectively defines the wavelength for exterior illumination, while the statement "Is a provision made for a compliance inspection during the nesting season?" (item 1 of Implementation Component) requires the provision of appropriate means to monitor the installation of this favorable type of beach lighting.

To rate each ordinance in terms of its effectiveness of ensuring sea turtle friendly lighting we follow the approach by Barshel *et al.* (2014) closely. More precisely, an ordinance is evaluated on two specific scales. Regarding the Principles Component statements, the scale takes four possible values: $0 \equiv$ concept not mentioned; $1 \equiv$ concept addressed but vague; $2 \equiv$ concept addressed but less stringent (wording provides loopholes); and $3 \equiv$ addressed with the same strength. With respect to the Implementation statements, the scale assigns 1 if the concept is addressed or 0 otherwise. For each group of statements the scores were summed and normalized to have a maximum value of 50, and then these two sums were added to provide a score that ranges between 0 and 100 for each ordinance. For each of the nesting beaches we determined which municipality and county they were located in and assigned to them the highest score of the two for each year, taking changes over time into account in doing so. In cases where beaches crossed county or municipality borders we weighted the scores according to the length of the beach located within them.

Figure 2 depicts the distribution of the legislation score corresponding to each nesting beach at the end of our sampling period, 2014. Accordingly, there is clearly substantial spatial heterogeneity with regard to sea turtle friendly legislation. From Table 1 one can see that the average ordinance effective proxy, $SCORE$, in the Index beaches is somewhat smaller (16) than the total sample (42), but both have considerable variation. The maximum score observed over our sample period is 83.

Histogram description

Figure 2: Score of Nesting Beaches

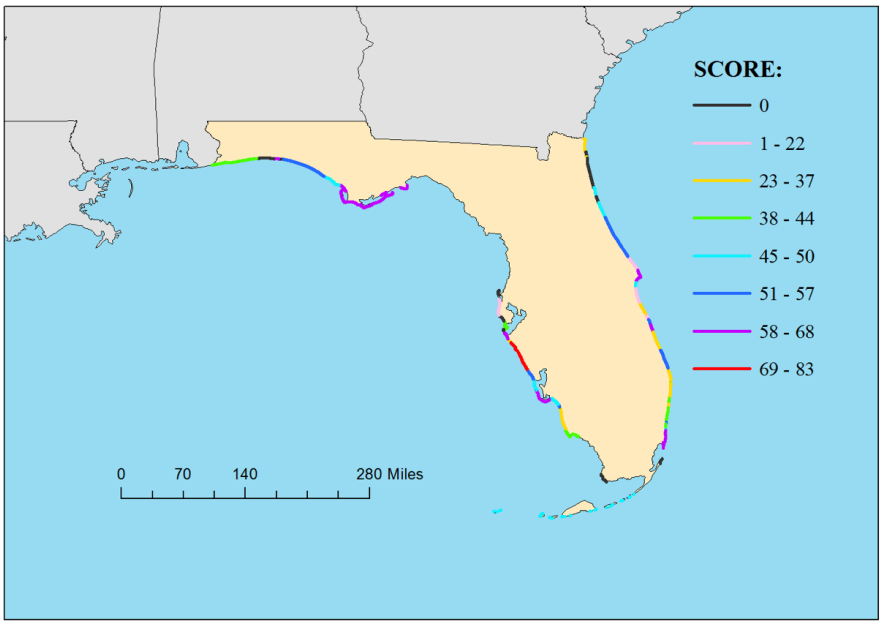
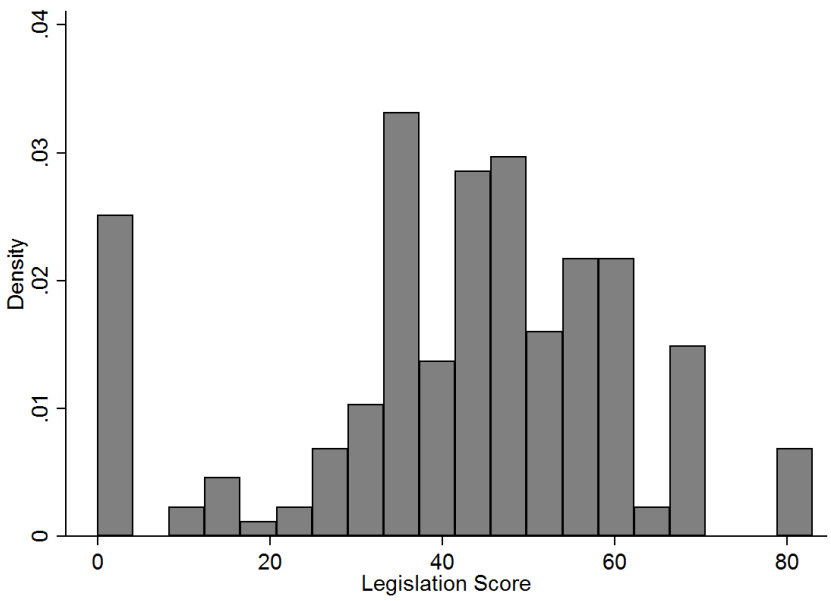


Figure 3: Distribution of *SCORE*



3.2.1 Verification of the ordinance score proxy

Given that the creation of our ordinance score variable is based to some extent on a subjective evaluation of the legislation involved, it is important to verify that it actually

captures what it is supposed to, namely the variation in the effectiveness in ensuring STFL during nesting season. To this end Anderson *et al.* (2013) identified a number of beaches in Florida with and without lighting ordinances and compared lighting intensity in months during the nesting season to the months outside of the nesting season using nighttime luminosity data derived from satellite imagery. Their analysis, employing difference in means tests, suggested that almost all beaches with ordinances in their sample seemed to be in compliance with the legislation in the sense that nightlight intensity was significantly lower during nesting season. We extend their approach here to allow for ordinance effictiveness, rather than just incidence, and econometrically test any differences across beaches using the following specification:

$$NL_{imt} = \alpha + \eta SCORE_{imt} + \beta SCORE_{imt} \times MAYNOV_i + + \gamma MAYNOV + \zeta X + \pi_m + \lambda_t + \mu_i + e_{imt}, \tag{1}$$

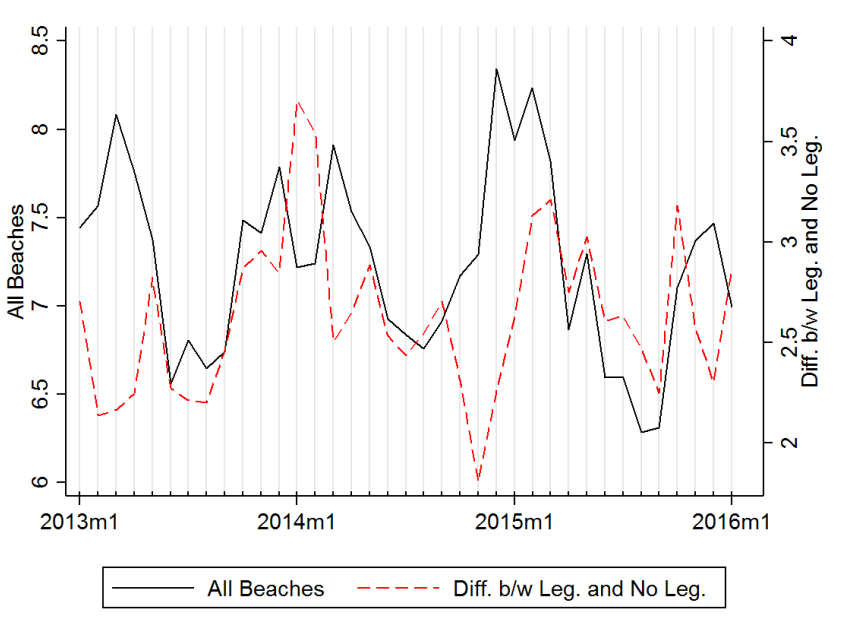
where NL is the intensity of nightlights on beach i in month m of year t, $SCORE$ is the nesting friendly legislation score, $MAYNOV$ is an indicator variable capturing the joint effect of the May through November months, *i.e.*, the official nesting season when ordinances are generally in effect, X is a vector of control variables, π is a vector of monthly indicator variables controlling for monthly differences in nighttime brightness independent of lighting ordinance effectiveness , λ is a vector of yearly indicator variables capturing yearly effects common to all beaches, μ is a set of beach specific indicator variables capturing time invariant beach specific factors, and e is an error term. Given the likely serial and spatial correlation of the data we calculate Driscoll-Kray (1998) standard errors for (1).

To proxy local nightlight intensity, NL, we, as in Anderson *et al.* (2013), use the Visible Infrared Imaging Radiometer Suite (VIIRS) monthly nightlight imagery collected and processed from the Suomi National Polar-Orbiting Partnership (SNPP) satellite since April 2012. These processed data provide measures of nightlight intensity across the globe at roughly 1h30 in the morning at a resolution of roughly 500m.[20] We use images over the 3 year period 2013-2015 in order to have three complete annual cycles. To capture the monthly nighttime brightness on all nesting beaches, both Index and non-Index, we took the nesting beach polylines in Figure 1, created 250m buffers around these, extracted the nightlight grid cells from the monthly VIIRS, and averaged these for each

[20]The fact that light intensity is only measured at 1h30 in the morning possibly means that using it as a proxy may ential missing some of the variation in nightlights if some areas reduce nightlights after a certain hour regardless of legislation.

month for each beach. As can be seen from 1, the average (pseudo) radiance value varies considerably across beaches. We also depict the evolution of the average monthly nesting beach nightlight intensity over our sample period in Figure 4, along with the difference in this between beaches with and without STFL lighting ordinances. Accordingly, there is clearly seasonal variation in nighttime brightness of the beaches. More specifically, the brightest period is around March, after which the level of brightness falls until around September when it noticeably rises again. Arguably there are potentially two factors underlying these trends. Firstly, this cyclical pattern is certainly due in part to the nature of the tourist season in Florida. For instance, in the sub-tropical south of Florida the peak season roughly spans from mid-December until mid-April. In terms of estimating (1) one should note that the monthly indicators π should control for this seasonality across years. A second reason is that sea turtle nesting friendly lighting legislation during nesting season existent on some beaches may also play a role. This is, however, not discernible by comparing the trends in beaches with and those without any legislation, as the red line in Figure 4 shows, suggesting that the extent of legislation, rather than the incidence, may instead play a greater role.

Figure 4: Monthly Nightlight Intensity on Nesting Beaches



In terms of estimating (1) we first start of with modeling μ as a beach specific random effect, and include, in addition to monthly and year dummies, the number of hotel rooms in 2013, average annual number of storms since 1989, average annual beach nourishment

since 1989, and average income per capita in 2013 as controls in X.[21] As can be seen in Table 2, $SCORE$ only has an effect on beach nightlight intensity during the official nesting season, May to November. More specifically, beaches with greater STFL legislation display lower light pollution during this period. This effect is, as shown in the second column, only linear.

We next experiment with modeling μ as a time invariant fixed effect. One should note that in this regard, that while there were considerable changes of legislation over the sample period of our nesting data, there were no changes over the three year period of the nightlight data, 2013-2016, so that $SCORE$ itself is actually time invariant for these years. Its effect is thus absorbed by the beach fixed effects. The results of estimating (1) are depicted in the third and fourth column of Table 2. Accordingly, one can decisively reject the null hypotheses that time invariant beach specific effects have no influence on differences in nightlight intensity across beaches. Again, we find a negative and statistically significant coefficient on the $SCORE \times MAYNOV$ interaction term, but no non-linear effect. Taking the coefficient at face value, the estimate suggests that a one point increase in $SCORE$ reduces nightlight intensity on a nesting beach by 0.1 per cent on the average lit beach. If we take our average (max) legislatively protected nesting beach relative to those that have no legislation in place, then the relative reduction in nightlight intensity on the average beach would be 5.2 (12) per cent. Overall, while our lack of other monthly time varying controls does not allow us to decisively conclude that the relationship between nightlight light intensity and lighting ordinances is causal, it is certainly suggestive of this and gives us some confidence that our proxy is a reasonable measure of ordinance effectiveness.

[21] Unfortunately we do not have monthly data for these controls.

Table 2: Nightlight Regression

	(1)	(2)	(3)	(4)
$SCORE$	0.1075	0.0034		
	(0.0940)	(0.2985)		
$SCORE \times MAYNOV$	-0.0105**	0.0084	-0.0105**	0.0084
	(0.0041)	(0.0123)	(0.0042)	(0.0125)
$MAYNOV$	-0.1278	-0.3018	-0.0606	-0.2345
	(0.2722)	(0.2924)	(0.2307)	(0.2548)
$SCORE^2$		0.0015		
		(0.0041)		
$SCORE^2 \times MAYNOV$		-0.0003		-0.0003
		(0.0002)		(0.0002)
Model	RE	RE	FE	FE
Observations	1188	1188	1188	1188
(within) R^2	0.087	0.089	0.063	0.065
$F(\mu_i \neq 0)$	—	—	2130***	2124***

Notes: (a) Estimator: Fixed Effects (FE) Linear Estimator; (b) All time invariant factors are purged from the Equation (1) via the FE estimator; (c) Driscol-Kraay (1998) standard errors in parentheses; (d) (within) R^2 is the percentage of within beach explained variation; (e) ***, **, and * indicate 1, 5, and 10 per cent significance levels, respectively; (f) $F(\mu_i \neq 0)$ is an F-test of the beach specific effects μ being jointly equal to zero.

3.3 Other determinants of nesting activity

Given the likely non-random nature of the location of sea turtle lighting friendly legislation, it is of course of crucial importance to ensure that in our empirical estimation we capture all determinants of sea turtle nesting that may be correlated with the implementation of the regulatory framework across time and space. Our strategy to this end was to extensively survey both the economic and non-economic literature for factors that have been shown to affect or are correlated with nesting, and create indicators of these.

3.3.1 Hotels

Florida is well known for its attractive beaches. In this regard, guest accommodation near the beachfront is often particularly valued and unsurprisingly Florida has seen a surge in hotel construction over the last few decades. However, hotels constructed close to the waterfront may, apart from causing lighting at night, also have other adverse impacts on sea turtle nesting activity that one would want to control for. For one they may

cause coastal squeeze and thus reduce the natural habitat of sea turtles.[22] Additionally, the inherent tourist activity surrounding hotels may act as a direct disturbance to sea turtle nesting; see Davenport and Davenport (2006). Our data source for hotels within proximity of a beach is the SRT Share hotels database. More specifically, the SRT lists all hotels, existing and closed, since 1987, including the exact starting year, number of rooms[23], and latitude and longitude of location, amongst other things. It thus allows us to create a time varying measure of hotel capacity for each nesting beach within a chosen threshold of proximity. We as a benchmark consider hotels within 100 meters of the shoreline, but also experiment with setting this threshold further away. As can be seen from Table 1, across our nesting beach sample there is, on average, a hotel room capacity of about 48 rooms, but with considerable variation.

3.3.2 Income per capita

The wealth of the local community of a beach may also play a role in nesting activity in that wealthier communities are more likely to be environmentally friendly and hence supportive of sea turtle friendly lighting. For instance, Yin *et al.* (2010) showed that income played an important positive role for the valuation of sea turtle conservation in Asia. To capture this aspect we use the most disaggregated measure of local income available for Florida, *i.e.*, county level local pesonal income from the BEA Regional Economics Accounts. We normalize this series by county population and deflate it to be in '000s of 2014 US dollars. Each nesting beach is assigned the income per capita series of the county that it is located in. For those beaches that stretch across more than one county we use a beach length weighted average of the county level deflated per capita data. Table 1 shows that the average income in the counties containing the sample nesting beaches is about $US 40,000, with some beaches located in counties with an income double this value.

3.3.3 Beach nourishment

Beach nourishment, the human replacement of lost sand on beaches, which is a common practise in Florida, may also have an effect on sea turtle nesting. For instance, Rumbold *et al.* (2001) find that in the first season after a beach nourishment project was implemented

[22]See, for instance, Mazaris *et al.* (2009) for an analysis of coastal squeeze on sea turtles.

[23]For a few hotels there was no information with regard to the number of rooms. For these we estimated the number of rooms by using the mean number of rooms of hotels by type of operation (chain management, franchise, or independent), type of location (airport, interstate, resort, small metro/town, suburban, or urban), type of price (budget, economy, luxury, mid-price, or upscale), class (economy, luxury, midscale, upper midscale, upper upscale, or upscale), and whether it had a restaurant.

on Palm Beach (Florida) the number of loggerhead nests fell significantly, with a smaller effect in the second season. To take account of beach nourishment we construct annual time series of beach nourishment projects using information from the Strategic Beach Management Plan (SBMP) reports. More specifically, the SBMP reports, one for each of Florida's 7 regions[24], serves as an inventory of Florida's strategic beach management areas located on the Atlantic Ocean, Gulf of Mexico, and Straits of Florida. Importantly, these reports contain for each beach a detailed account of beach nourishment projects over time, including the year, the volume of sand used, and the location of the segment(s) of beach treated in terms of the nearest starting and nearest ending beach range monument.[25] We allocated projects to the nesting beaches if their segments fell within the stretch of a beach. In the case where the segment(s) was not completely contained within a nesting beach, we allocated the proportion of volume equivalent to the proportion of segment of the project intersected by the beach. Table 1 shows that the average annual volume of sand placed on nesting beaches is about 2 million cubic yards.

3.3.4 Storms

Another factor potentially affecting sea turtle nesting are storms causing beach erosion. In this regard, Houtan and Bass (2007) used monitoring surveys over a ten year period from the Dry Tortugas National Park to show that the incidence of tropical cyclones decreased the number of loggerheads substantially. In order to capture the potential role of storms we use the SBMP reports which list for each region all known damaging storms as well as the beaches affected. To take account of their effect we simply use the total count of storms, both tropical storms and easterlies, that affected a beach in a given year. The summary statistics in Table 1 reveal that on average a beach is affected by roughly one storm per year, but that some have been substantially more affected than others.

[24]These regions are the Northeast Atlantic Coast, Central Atlantic Coast, Southeast Atlantic Coast, Florida Keys, Southwest Gulf Coast, Big Bend Gulf Coast, and Panhandle Gulf Coast.

[25]The range monuments are FDEP beach reference points, spaced offshore approximately 1,000 feet apart, and are typically known as "R" stations.

4 Econometric analysis

4.1 Nesting activity regression

In order to investigate the impact of sea turtle friendly lighting legislation on loggerhead nesting activity we estimate the following:

$$NESTS_{it} = \alpha + \sum_{j=1}^{n} \beta_j SCORE_{it}^j + \lambda C_{it} + \pi_t + \mu_i + e_{it}, \tag{2}$$

where $NESTS$ are the number of loggerhead nests found on Index beach i in year t, and $SCORE$ is our legislation scoring variable, possibly including higher order terms. The vector π are a set of yearly indicator variables capturing year specific shocks affecting all nesting beaches, μ are time invariant beach specific effects, and e is the error term.

The dependent variable in (2), $NESTS$, is by nature a count variable and hence standard linear regression methods are not appropriate. We thus instead use a count variable estimation model. In this regard, the two most common choices are the Poisson and Negative Binomial model, where the latter is preferred if there is over-dispersion. As Table 1 shows, this is indeed true in our case and we therefore employ a negative binomial count model. In those specification where we take account of the beach specific time invariant effects, μ, we run the fixed effects version of this model; see Cameron and Trivedi (2013).[26]

Our aim is to identify the causal effect of $SCORE$ on nesting activity. Clearly, given that there a number of environmental and economic factors that are likely to affect $NESTS$ but are also correlated with $SCORE$, simply regressing $NESTS$ on $SCORE$ is likely to fail to do so. Incorporating beach fixed effects μ will take account of any potentially problematic time invariant unobservables, such as unobserved beach characteristics, that are beneficial to nesting but also may attract tourist activity. To also control for beach specific time varying factors that may affect both, our vector C consists of our other control variables, which, as outlined earlier, are the known relevant determinants of nesting as identified from the existing literature. More specifically, we include the number of hotel rooms, $ROOMS$, county level income per capita, $INCOME/CAP$, the volume of sand placed under beach nourishment proejcts, $NOURISHMENT$, and the

[26] One should note that the fixed effects in Cameron and Trivedi (2013) estimator are conditional fixed effects and thus not strictly equivalent to the inclusion of panel unit dummies, which would produce inconsistent estimates; see Allison and Waterman (2002). For convenience sake we nevertheless simply refer to these as fixed effects.

number of storms, $STORMS$. Arguably, with this rich set of controls we are likely to be capturing all time varying beach specific factors relevant to sea turtle nesting activity that may also be related to the extent of sea friendly lighting legislation at a beach.

One may also worry about possible simultaneity of the passing of ordinances, or changes therein, and nesting activity. In particular, if nest counts at a beach indicate a potential fall in nesting activity for that season a ordinance may be passed to counteract such a decrease. There are a number of reasons why in reality this is unlikely to be a concern. Firstly, nest counts are only collated at the end of the season. Secondly, passing an ordinance can be a tedious and long process.[27] For instance, at the municipality level the typical procedure involves drafting a proposal that needs to be submitted to the city council, which may then consult on the content through a special committee. Often there is also a public hearing, which may result in further proposed changes to the legislation. Finally, after potentially several revisions and repetitions of this procedure, the ordinance must be voted on by the council and possibly the city major, and then takes effect at a specified later date. This whole process is likely to take some time, and almost certainly would not be completed within a nesting season.

4.2 Regression results

The results of estimating variations of (2) are provided in Table 3. In the first column we ran a negative binomial regression of $NESTS$ on $SCORE$ without controlling for any covariates apart from yearly indicator variables. As can be seen, the coefficient on the legislative variable is positive and significant, indicating that sea turtle protective legislation has acted to increase nesting activity of loggerheads. When we subsequently allowed for beach specific time invariant differences by using the fixed negative binomial estimator in the second column, the coefficient was reduced by nearly 90 per cent. This indicates that there are important time invariant differences across beaches that are positively correlated with both legislation and nesting activity and not taking account of these would severely upwardly bias the estimated effect of legislation on nesting. We thus for further estimations proceeded to rely on the fixed effects negative binomial model.

In the third column of Table 3 we include our vector of controls, C, as outlined above. As can be seen, their inclusion only slightly changes the coefficient on $SCORE$ after controlling for beach time invariant effects. In terms of their role in affecting nesting activity, we find that more hotel rooms located near the beach lowers the number of

[27] See http://www.statescape.com/resources/local/ordinance-process.aspx

loggerhead nests, whereas beaches located in richer counties tend to have more nesting activity. In contrast, neither beach nourishment projects nor the number of storms have significantly affected loggerhead nesting. To investigate whether hotels further than 100 meters from the shoreline might still have an impact on nesting we increased the threshold d to 200 meters, where the results of including this alternative proxy are depicted in the fourth column. However, the lower threshold renders the coefficient on $ROOMS$ insignificant. We also explored whether the fact that we have not found any significant role for beach nourishment projects and storms is because we were assuming only a contemporaneous effect by including their lagged values in the fifth and sixth column. However, as can be seen, this does not change our conclusion regarding their lack of importance in discouraging loggerheads to nest on a beach. Since re-migration, when it occurs, is about every 2 years [28], we also included up to three lags of dependent variable as additional controls in the seventh column, but this also does not change the significance on $SCORE$.

[28] See Bjornda and Meylan (1983).

Table 3: Nests Regression

	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
$SCORE$	0.04270**	0.00478**	0.00481**	0.00479**	0.00482**	0.00478**	0.00892***	0.02040**
	(0.00524)	(0.00099)	(0.00101)	(0.00111)	(0.00101)	(0.00101)	(0.00121)	(0.00374)
$SCORE^2$								-0.00029**
								(6.62e-05)
$ROOMS[d = 100m]$			-0.00223**		-0.00223**	0.00222**	-0.00229***	-0.00207**
			(0.000705)		(0.000695)	(0.000707)	(0.00062)	(0.000741)
$INCOME/CAP$			0.00888*	0.00885*	0.00906**	0.00869*	0.00982	0.00667
			(0.00354)	(0.00358)	(0.00351)	(0.00356)	(0.00370)	(0.00351)
$NOURISHMENT$			-0.00233	-0.00288	-0.00240	-0.00234	-0.00213	-0.00217
			(0.00202)	(0.00201)	(0.00204)	(0.00202)	(0.00171)	(0.00199)
$STORMS$			-0.00337			-0.00329	0.00191	-0.00153
			(0.0122)			(0.0122)	(0.01124)	(0.0119)
$ROOMS[d = 200m]$			-1.89e-05					
			(3.70e-05)					
$NOURISHMENT(t-1)$					-0.000465			
					(0.00173)			
$STORMS(t-1)$						-0.00606		
						(0.0114)		
$NESTS(t-1)$							0.00002***	
							(9.75e-06)	
$NESTS(t-2)$							0.00002	
							(0.00001)	
$NESTS(t-3)$							0.00002	
							(9.93e-06)	
Observations	780	780	780	780	780	780	690	780
Beaches	33	33	33	33	33	33	33	33
Log Likelihood	-5962	-4221	-4214	-4216	-4214	-4214	-3632	-4205
χ^2-Test	66.39***	342.96***	356.66***	353.01***	366.66***	365.67***	611.19	389.96***

Notes: (a) Robust standard errors in parentheses; (b) ***, **, and * indicate 1, 5, and 10 per cent significant levels; (c) Dependent variable is number of loggerhead nests; (d) All regressions include yearly indicator variables; (e) Column (1) is a standard negative binomial estimator, while columns (2) through (8) include time invariant beach specific effects.

Thus far we have assumed that an increase in our legislative score can only have a linear impact on nesting activity. Feasibly at higher levels of effectiveness additional refinement of the code may have less of an impact than where there are few legislative provisions to ensure sea turtle friendly lighting. In the final column of Table 3 we hence included the squared value of $SCORE$ as an additional control. Accordingly, while the linear term remains significant, the quantitative size of the coefficient increases nearly fourfold. At the same time, however, the squared value of $SCORE$ is significantly negative. Taking together this suggests that there is am inverted u-shaped relationship between $SCORE$ and $NESTS$, so that a higher legislative score will encourage nesting activity but at a decreasing rate.[29]

4.3 Robustness Checks

[29]One may want to note that, although not reported here, we also experimented with further higher order terms but these turned out to be insignificant.

Table 4: Robustness Checks

	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
$SCORE$	0.01161***	0.01089***	0.04453***	0.03621***				
	(0.00143)	(0.00176)	(0.00518)	(0.01074)				
$SCORE^2$			-0.0060***	-0.00048**				
			(0.0009)	(0.00019)				
$FIRST$					-0.40874			
					(0.24571)			
$SCORE * FIRST$					0.04211***			
					(0.01305)			
$SCORE^2 * FIRST$					-0.00053***			
					(0.00016)			
$SCORE25$						0.37751***		
						(0.06373)		
$SCORE50$						0.42175***		
						(0.08889)		
$SCORE100$						0.13164**		
						(0.05526)		
$PSCORE$							0.01406***	
							(0.00291)	
$ISCORE$							0.02622***	
							(0.00672)	
$PSCORE * ISCORE$							-0.00114***	
							(0.00026)	
Observations	600	600	600	600	780	780	780	
Beaches	33	33	33	33	33	33	33	
Log Likelihood/R^2	-3168	-3157	-3146	-3142	-4203	-4201	-4202	
χ^2/F-Test	427.83***	476.69***	500.05***	521.15***	398.19***	415.41***	396.07***	

Notes: (a) Robust standard errors in parentheses; (b) ***, **, and * indicate 1, 5, and 10 per cent significant levels; (c) Dependent variable is number of loggerhead nests; (d) All regressions include yearly indicator variables; (e) Column (1) is a standard negative binomial estimator, while columns (2) through (8) include time invariant beach specific effects.

We argued earlier that since we included proxies for all possible other determinants of nesting, as found in the existing literature, our specification in (2) is unlikely to suffer from omitted variable bias. Moreover, given the lengthy process of passing legislation there is also unlikely to be a simultaneity issue. To further verify this we additionally experimented with instrumenting $SCORE$. More specifically, a plausible instrument might be the political composition of the area around a beach, under the argument that local voters may pressure local legislators to pass STFL more, or less, according to their political affiliation, but that this composition is unlikely to have any direct effect on sea turtle nesting. To proxy such local political composition we use data on voter registration by county and by party available from the Florida State Department, and construct from this the share of democratic party registered voters related to each beach. Since this data is only available from 1995 onwards, we first re-ran our specification in (2) allowing for a linear effect of $SCORE$ on this reduced sample. As can be seen from the first column in 4, while the coefficient is somewhat higher than in the full sample, it remains statistically significant. Since (2) is a non-linear model, standard 2SLS methods would be inappropriate in order to instrument $SCORE$. Rather we, as suggested by Woolridge (2005), use a control function approach where the endogenous variable is regressed on the instrument and all other control variables and the predicted error term is then included in the second stage as an additional variable. One should note in this regard that for the first stage the share of democratic registered voters was a highly significant (negative) predictor of STFL, with a t-statistic of 8.51, and thus a relevant instrument. Importantly, instrumenting for $SCORE$ in (2) only marginally changes its coefficient, as shown in the second column. As a matter of fact, a z-test (statistic of 0.32) does not suggest that one should reject the null hypothesis that instrumented and non-instrumented coefficients are the same. In the third and fourth column we show similarly for the 1995 to 2013 sample the specification allowing for a non-linear effect of $SCORE$ non-instrumented and instrumented, respectively. [30] Again, the coefficients on $SCORE$ and $SCORE^2$ hardly change when we instrument for them, where this lack of difference is further confirmed by a z-test.[31]

Next we explored whether it is the incidence of having legislation, rather than the extent of legislation, that affects loggerhead nesting activity. To this end we created

[30] This consisted of instrumenting $SCORE$ and $SCORE^2$ with the share of demographic voters and its value squared. The F-statistics on these predictors for the first stage of the levels and squared equations were 44.53 and 29.51, respectively.

[31] The z-statistic was 0.70 and 0.59 for $SCORE$ and $SCORE^2$, respectively.

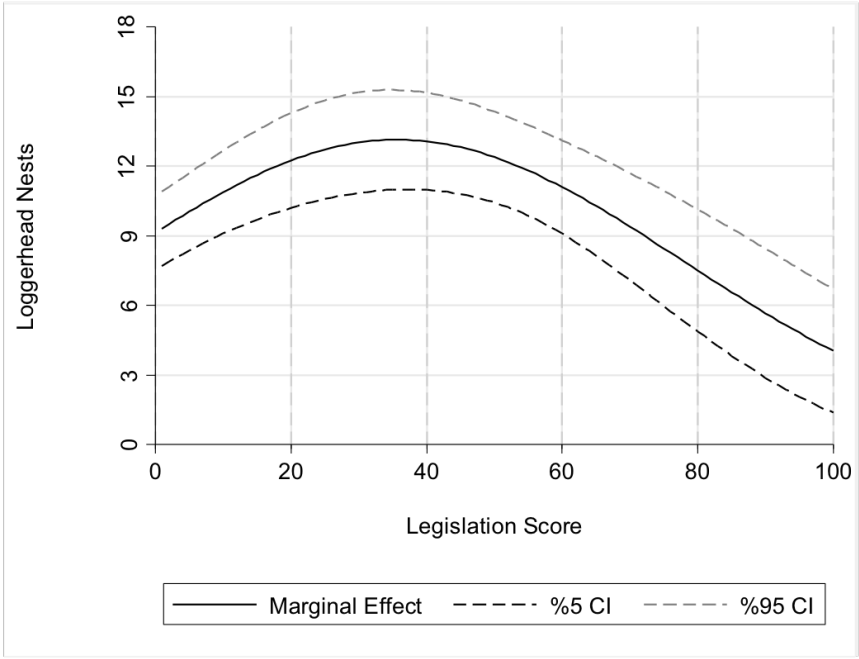
a dummy variable, $FIRST$, that takes on the value of one when $SCORE > 0$ and zero otherwise, and include it as well as its interactions with $SCORE$ and $SCORE^2$ in our benchmark specification. As can be seen, from the fifth column in Table 4, it is the extent of the legislation that is driving the positive impact on nesting. We also experimented with using a piecewise polynomial rather than a quadratic term to capture non-linearity in the legislation-nesting relationship, with knots chosen at 25, 50, and 100. The estimated coefficients on these, depicted in the sixth column, similarly that $SCORE$ has a decreasing impact on nesting, especially once it passes a value of 50.

One may recall that we constructed $SCORE$ as a simple sum of points given to the Principal Components and Implementation statements. In other words, we assume that both aspects have a similar and cumulative impact opn nesting. To further investigate this we split $SCORE$ into these two factors, $PSCORE$ and $ISCORE$, respectively, and included these as well as their interaction term in (2). As can be seen from the final column in 4, $ISCORE$ has a 85 per cent higher per point impact than $PSCORE$, but, as the interaction term suggests, the two factors act partially as substitutes. This may explain the inverted u-shaped relationship we find when we use our total legislation score.

4.4 Marginal Effect

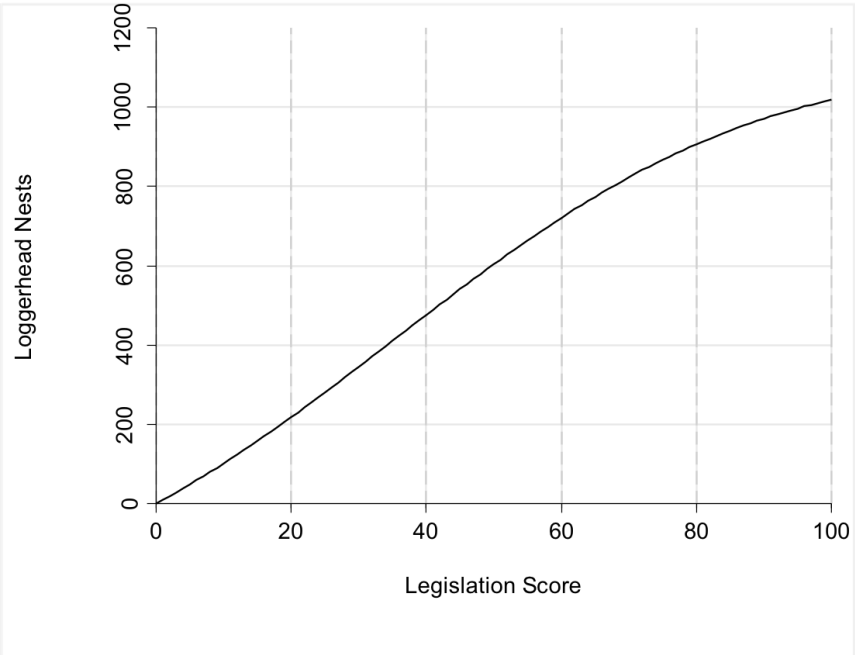
We can now use our coefficients to determine the quantitative impact of ordinance effectiveness on loggerhead nesting. One should note in this regard that there are two sources of non-linearity as a result of our econometric specification: the non-linear nature of the negative binomial model and the non-linear effect of $SCORE$. Using the estimated coefficients from column (7) of Table 3 and setting the value of all other co-variates at their mean, we depict the implied marginal impact of $SCORE$ across its range along with 95 per cent confidence intervals in Figure 5. As can be seen, moving from a beach not covered by any sea turtle lighting related legislation to one with a score of 1 increases the number of nests by 9. This marginal effect increases as one moves across the range of possible values to reach a peak at 36 to then start falling. At the end of the spectrum, increasing the score from 99 to 100 results in an additional 4 nests.

Figure 5: Marginal Effect of Legislation



In Figure 6 we plot the full range of cumulative effects by summing the marginal effects across the range of possible values of $SCORE$. Accordingly, the additional nests to be gained for a beach with no legislation in place of one with the highest possible score of a 100 is 1,119 nests. Comparing this to the mean, this suggest that the average beach would experience a 76 per cent increase to go from no legislation to adopting an ordinance with maximum effectiviness. If we consider the mean score of all nesting beaches in Florida, *i.e.*, 42, then our results suggest that on average an additional 502 nests annually on a beach can be attributed to the current legislation implemented in Florida, *i.e.*, about 34 per cent. One may also note that the turning point of inverted u-shape relationship is beyond the maximum score of 100.

Figure 6: Cumulative Effect of Legislation



5 Sea turtle population dynamics

In the previous section we quantified the effect of STFL ordinances on nesting activity. However, importantly in terms of species preservation, impacts at the nesting stage will also feed into the population dynamics of turtles. We thus now quantify the effect of coastal ordinances on sea turtles by means of introducing our estimates into a dynamic model of their population. In this regard it is well-known that the life cycle of sea turtles is comprised of a series of development stages (see, for instance, Heppell *et al.*, 2003). We thus employ the stage-structured population model of Crouse *et al.* (1987) and Crowder *et al.* (1994), in which the individual females are grouped by stage. Each stage is characterized by its annual reproduction and survival rates, as well as by the number of years that a turtle stays in that stage.

5.1 Population model

Our model consists of five stages of sea turtle development: (1) eggs/hatchlings, (2) small juveniles, (3) large juveniles, (4) subadults, and (5) adults. We define the stage distribution vector at time $t \geq 0$ as

$$x_t \equiv (x_{1t}, x_{2t}, x_{3t}, x_{4t}, x_{5t}), \tag{3}$$

where x_{it} is the number of female sea turtles in stage i at time t for $i = 1, \ldots, 5$. By means of a five-stage Leslie matrix L, we obtain the population distribution at time $t+1$ as

$$x'_{t+1} = Lx'_t, \tag{4}$$

where x' denotes the transpose of vector x. Hence, starting from a given initial stage distribution x_0, we get the evolution of the population by recursively applying (4).

Let P_i denote the percentage of females in stage i that survive but remain in that stage, G_i be the percentage of females in stage i that survive and progress to the next stage, and F_i be the number of hatchlings per year produced by a sea turtle in stage i (i.e., annual fecundity). Therefore, the number of hatchlings produced by each stage class at time $t+1$ is given by

$$x_{1t+1} = F_1 x_{1t} + F_2 x_{2t} + F_3 x_{3t} + F_4 x_{4t} + F_5 x_{5t}. \tag{5}$$

Moreover, the number of females present in the subsequent stage j, for $j = 2, \ldots, 5$, is

$$x_{j_{t+1}} = G_{j-1} x_{j-1_t} + P_j x_{j_t}. \tag{6}$$

Taking (5) and (6), the matrix L in (4) is

$$L \equiv \begin{bmatrix} F_1 & F_2 & F_3 & F_4 & F_5 \\ G_1 & P_2 & 0 & 0 & 0 \\ 0 & G_2 & P_3 & 0 & 0 \\ 0 & 0 & G_3 & P_4 & 0 \\ 0 & 0 & 0 & G_4 & P_5 \end{bmatrix}.$$

The fecundity rates F_i are typically directly obtained from actual data on sea turtles. However, G_i and P_i need to be calculated from information about the duration (d_i) and the yearly survival rate (σ_i) of each stage i. Let us first determine the percentage of sea turtles from stage i that grow into stage $i+1$ as

$$\gamma_i = \begin{cases} \frac{(1-\sigma_i)\sigma_i^{d_i-1}}{1-\sigma_i^{d_i}} & \text{if} \quad \sigma_i \neq 1 \\ \frac{1}{d_i} & \text{if} \quad \sigma_i = 1. \end{cases} \tag{7}$$

Consequently, the percentage of turtles in stage i that remain in that stage is $1 - \gamma_i$. We

can then obtain G_i and P_i as

$$G_i = \gamma_i \sigma_i, \tag{8}$$

$$P_i = (1 - \gamma_i)\sigma_i. \tag{9}$$

The solution of (4) for all $t \geq 0$ is $x'_t = \sum_{i=1}^{5} c_i v'_{\lambda_i} \lambda_i^t$, where v_{λ_i} denotes the eigenvector corresponding to the eigenvalue λ_i of L, and c_i are constants determined by the initial stage population distribution. Considering the specific characteristics of loggerheads in Florida, we will show later on that $|\lambda_1| > |\lambda_j|$ for $j = 2, \ldots, 5$. Then, for a large t the solution takes the form

$$x_t{}' \simeq c_1 v'_{\lambda_1} \lambda_1^t, \tag{10}$$

where λ_1 is frequently called the dominant eigenvalue. Therefore, the long-run proportion of the population in stage i is given by

$$\xi_i = \frac{v_{\lambda_1 i}}{\sum_{k=1}^{5} v_{\lambda_1 k}}, \tag{11}$$

where $v_{\lambda_1 k}$ is the kth coordinate of the eigenvector v_{λ_1}.

In order to compute the matrix L we need data about the duration (d_i), the annual fecundity rate (F_i) and the yearly survival rate (σ_i) of turtles in each stage i. Crowder *et al.* (1994) provide this information for loggerheads in Florida (see Appendix B). Appendix C describes the properties of the corresponding Leslie matrix, mainly focusing on the eigenvalues and the long-run stage distribution (11). With respect to the initial distribution x_0, we use available current estimates of the number of adult loggerhead females in Florida. With the share of turtles in each stage, we can then establish the number of individuals per stage and, consequently, the initial vector x_0.

As in Crowder *et al.* (1994), we suppose that the distribution of individuals among stages is stable. We showed above (Equation 10) that one can reasonably assume this when the population is near to its long-run equilibrium. In this regard, provided that λ_1 is the dominant eigenvalue of L (see first row of Table A.2 in Appendix C), the model predicts that the percentage of female loggerheads in each stage (Table A.3 in Appendix C) will be 21.72 (stage 1), 67.65 (stage 2), 9.76 (stage 3), 0.61 (stage 4), and 0.26 (stage 5). Moreover, since $|\lambda_i| < 1$ for $i = 1, \ldots, 5$, sea turtles characterized by the model parameters will face the risk of extinction in the long-run.[32] Indeed, as noted in the

[32]Notice that, for $|\lambda_i| < 1$ for $i = 1, \ldots, 5$, the number of individuals asymptotically converges to zero.

introduction, loggerheads in Florida are currently listed as threatened according to the US Endangered Species Act (Witherington *et al.*, 2009).

Richards *et al.* (2011) estimate that the population of adult loggerhead females in the western North Atlantic is about 38,334 individuals, most of them stemming from Florida (the largest subpopulation). Considering this value together with the percentages of individuals per stage stated above, we can calibrate x_0 (see last column of Table A.3 in Appendix C). Note that in our simulations we set $x_{10} = 3,528,180$ following the estimated production of loggerhead hatchlings per year in Florida reported by Brost *et al.* (2015). This number is fairly close to our estimate of 3,159,771 loggerhead hatchlings.

With our calibrated model in hand, we can give numerical projections of the population of female loggerheads in Florida (per stage and in total) starting from the initial distribution x_0 until a time horizon $t = T > 0$. In particular, by considering a large enough T, we can quantify the years to extinction. In the context of our model, this can be defined as the number of years for less than one individual to remain.[33] We first consider the benchmark scenario of no ordinances, which is described by the L matrix corresponding to Table A.1 in Appendix B. We then compare it with the population dynamics under different strength levels of STFL ordinances.

In order to quantify the ordinances' effect on the population dynamics we incorporate the estimated effect of the $SCORE$ variable into the population model. Since our earlier results show how legislation raises loggerheads' nesting activity, the implementation of STFL ordinances will increase the annual fecundity rate F_i in the matrix L. In order to modify this parameter we compute the increase in hatchlings per year due to the adoption of ordinances at a strength level S. Let us denote by $NEST_{avg}$ the average number of nests per beach. If the ordinances are set at a strength level S, the estimated cumulative marginal effect of the legislation on nesting activity, $CSCORE_S$, will be as depicted in Figure 6. The average percentage point increase in nests $\tau(S)$ is then given by

$$\tau(S) = \frac{CSCORE_S/r}{NEST_{avg} + CSCORE_S/r} 100, \tag{12}$$

where r represents the remigration interval (in years) of loggerheads. One should observe that, as in Brei *et al.* (2016), we are working at the individual sea turtle level. Hence, since

[33] As it is standard in the sea turtles' biology literature, the population models focus on female dynamics. Therefore, extinction occurs when the last female disappears.

each turtle does not nest every year, we adjust the accumulative effect of the ordinances by dividing $CSCORE_S$ by the remigration interval r. In this regard Bjornda *et al.* (1983) show evidence of a remigration interval of loggerheads in Florida of 2 years, and Phillips *et al.* (2014) observe that this remigration interval has not changed over time in Florida. We thus consider $r = 2$ for our simulations. Moreover, with no empirical evidence available, we simply assume that the percentage increase in nests will result in the same percentage increase in eggs per female sea turtle. Hence, considering (12), we finally modify the annual fecundity as $\tilde{F}_i = [1 + \tau(S)/100]F_i$.

5.2 Out of Sample Prediction

Our econometric analysis is based only on the Index Beaches in Florida since surveying efforts for these have been consistent over time. However, we would like to use our results to make predictions for the total Florida loggerhead population, i.e., also for nesting activity in non-Index beaches. It is thus important to demonstrate that the estimated legislation-nesting relationship is representative for these beaches as well. To this end we have access to nesting data from the Statewide program for the years 2008 to 2014, under which both Index and non-Index were surveyed, but surveying efforts across time were not necessarily consistent. Sample statistics by beach category are shown in Table 5. As can be seen, while the number of hotel rooms within 100m of a beach, income per capita, and storm activity are not statistically different between the two groups of beaches, there are considerable differences in terms of nightlight intensity, legislation score, beach nourishment, and number of nests.[34]

To examine whether the differences in the mean characteristics across beach groups also translates into a different relationship between legislation and nesting activity, we re-estimated (2) using the Statewide program nesting data, but separately for Index and non-Index beaches in Table 6. As can be seen, for both samples there is a significant inverted u-shaped relationship between legislation and nesting activity. If we compare the coefficients on the Index beach sample to those in Table 3 one finds that they are good bit larger. This might be in part due to the difference in monitoring technique between the two surveys, as well as the more recent sample period of the Statewide program.[35]

[34]The larger nesting activity in Index beaches is not surprising, since these are in part chosen to be capture much of the nesting activity in Florida.

[35]As a matter of fact, in reducing the sample of the Index beach survey data that we used earlier to the same 2008 to 2014 period, the estimated coefficients from (2) on $SCORE$ $SCORE^2$ were 0.045 and -0.0005, respectively.

Table 5: Index vs. Non-Index Beach Comparison

Sample	Index Beaches		Non-Index Beaches		Difference
Variable	Mean	St.Dev.	Mean	St.Dev.	t-stat
NL	5.48	10.71	8.62	10.55	13.11***
NESTS	1295.62	2422.36	176.25	505.65	11.95***
SCORE	34.29	21.79	40.21	21.26	3.18**
ROOMS [d=100m]	92.16	432.45	77.25	213.18	0.63
INCOME/CAP	50.44	12.52	50.64	13.82	0.17
NOURISHMENT	1.91	11.16	0.322	4.51	3.02***
STORMS	0.68	1.04	0.73	1.09	0.56

Notes: *NL* $\equiv$ intensity of nightlights; *NEST* $\equiv$ number of nests; *SCORE* $\equiv$ legislation score; *ROOMS* $\equiv$ number of rooms within d meters of the shoreline; *INCOME/CAP* $\equiv$ county income per capita ('000s); *NOURISHMENT* $\equiv$ average annual volume (cubic yards) of sand placed on nesting beaches; *STORMS* $\equiv$ number of storms that affected a beach;

However, more importantly, there appears to be not too much difference in the coefficients on $SCORE$ and $SCORE^2$ between the two sample of beaches. As a matter of fact, a z-test of the difference in coefficients across the two samples was 0.0721 and 0.0002 for $SCORE$ and $SCORE^2$, respectively. One can thus conclude that, despite the differences in some of the mean characteristics, the base impact of sea turtle friendly legislation on nesting activity is similar on Index and non-Index beaches, and we thus can confidently use our results from the last column in Table 6 to predict loggerhead population dynamics for the entirety of Florida.

5.3 Dynamic results

Starting from the calibrated initial population x_0, we provide projections of the population of female loggerheads in Florida by recursively applying (4). We focus on three scenarios, considering different $SCORE$ levels of the ordinances. The scenario represented by the L matrix directly computed from Table A.3 in Appendix B corresponds to the situation of "no ordinances" in place, while the other scenarios consist of introducing various levels of STFL regulations.

As noted before, we will assume that the ordinances modify the annual fecundity rate by (12) and set $NEST_{avg}$ at the average observed of our nesting data, namely 1,468 nests (see Table 1). We first consider the case corresponding to the current level of ordinances in the nesting beaches in Florida, called "ordinances" scenario. In our model this situation is characterized by the average score $S = 42$ of all Florida nesting beaches as is

Table 6: Regression by Beach Type

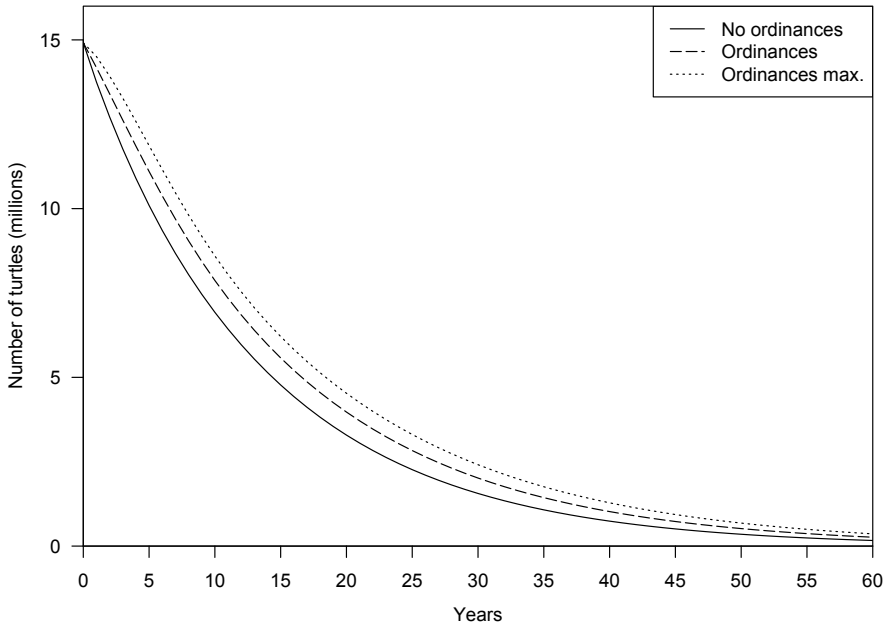
	Index Beaches	Non-Index Beaches
	(1)	(2)
$SCORE$	0.06008***	0.05273***
	(0.00524)	(0.00986)
$SCORE^2$	-0.00072***	-0.00072***
	(0.00027)	(0.00016)
Observations	164	796
Beaches	33	161
Log Likelihood	-686.5	-2413.7
χ^2-Test	252.3***	536.7***

Notes: (a) Robust standard errors in parentheses; (b) ***, **, and * indicate 1, 5, and 10 per cent significant levels; (c) Dependent variable is number of loggerhead nests; (d) All regressions include yearly indicator variables; (e) Column (1) is a standard negative binomial estimator, while columns (2) through (8) include time invariant beach specific effects.

evident from Table 1. This corresponds to a cumulative effect of $CSCORE_{42} = 502$ nests (see Figure 6) and implies an increase of the annual fecundity of about 15%. Finally, we study the prospect of setting the ordinances at the maximum effectiviness. Under this scenario ("ordinances max.") $S = 100$ across beaches and the corresponding cumulative effect is $CSCORE_{100} = 1,019$ nests, inducing the rate of annual fecundity to increase by 26%.

We depict the evolution of the Florida loggerhead population for our three different scenarios in Figure 7. Under the scenario of no ordinances $|\lambda_i| < 1$ for $i = 1, \ldots, 5$, and thus, as noted before, our model predicts that loggerheads will eventually become extinct. We find that this outcome is not reversed even when coastal areas adopt nesting friendly regulation at their maximum strength level (see rows 2 and 3 of Table A.2 in Appendix C). However, implementing the ordinances allows the population to increase along the transition due to the rise in fecundity.

Figure 7: Total Population



A direct implication of implementing STFL legislation is that loggerheads extinction will be delayed. We provide the exact years to extinction in Table 7. As can be seen, current level of ordinances increases years to extinction by 22 (10%), while implementing maximum effective legislation across Florida's nesting beaches will add a further 19 years.

Table 7: Time to Extinction

	No ordinances	Ordinances	Ordinances max.
Years to extinction	223	245	264
Relative change (%)	–	9.87	18.38

6 Monetary cost of light pollution

The model presented in the previous section has allowed us to study and quantify the dynamic effects of STFL ordinances on the different generations of sea turtles. In what follows, we will apply our results in order to assess the monetary cost of the loss of sea turtles due to not adopting STFL ordinances to varying degrees. This will be estimated by computing the replacement cost for sea turtles, defined as the cost of substituting turtles in the wild with individuals raised in captivity. Then, considering estimates for

the WTP (Willingness To Pay) for the protection of sea turtles in Florida and in the USA, we will evaluate to what extent actual public willingness to pay for the preservation of sea turtles can compensate for this cost.

6.1 Replacement cost

In the context of our paper, the replacement cost for sea turtles is used as a measure of the loss in ecosystem services due to the reduction in the population of turtles induced by artificial beach illumination, as in Brei *et al.* (2016). This cost should be understood, however, as a lower threshold of the true loss in ecosystem services since it ignores the potential survival differences between sea turtles raised in captivity and those raised in their natural environment (Freeman, 2003; and Troeng and Drews, 2004). Nevertheless, one should note that the notion of replacement cost in terms of the loss of population has already been recently applied in legal procedures against sea turtle eggs smugglers in the USA, where it was explicitly stated that "when an egg is destroyed, the defendant removes not only that specific potential animal from the population, but also all potential offspring that could have eventually been borne by that animal and its descendants" (Duffy, 2016b, page 7).

In order to determine the replacement cost for turtles we consider hypothetical fixed-term captivity-rearing programs of loggerheads in Florida. More specifically, by means of our calibrated population model we quantify the number of turtles that these programs would require to breed in order to achieve the same extinction reducing effect as STFL ordinances. Then, taking into account actual information about the rearing cost of loggerheads in conservation facilities, we compute the cost of such an alternative conservation management strategy.

An important element in our valuation exercise is the actual cost of raising sea turtles in captivity. As determined from a personal communication with Benjamin Higgins, NOAA Federal/National Marine Fisheries Service Galveston Laboratory, we note that turtles are typically raised in Florida/Gulf of Mexico until they are about 2 years old in order to benefit from the use of turtle excluder devices (TED), mandatory in shrimp trawls since 1987 (see Duffy, 2016b).[36] We thus focus on the rearing costs of 2 year-old loggerheads, and in line with information provided by Nicholas Blume, Florida Atlantic

[36]TEDs are scape hatches, which allows captured turtles to escape nets before drowning (, see, for instance, Crowder *et al.*, 1994; or Heppell *et al.*, 2003).

University (FAU), assume this to be US$836 per 2 year-old.[37] Alternatively, we use the figure of about US$2000 per loggerhead as suggested by Benjamin Higgins from the NOAA facility in Galveston Laboratory, and which was used in the aforementioned legal case (see Duffy, 2016ab).

With the rearing cost figures in hand we compute what the cost of a rearing program would be as an alternative to various STFL scenarios in terms of their effectiveness of reducing the time to extinction. As in Jin *et al.* (2010) and Brei *et al.* (2016), we concentrate on the cost of 5-year term programs, where we here inject two year old loggerheads raised in captivity in each of the five year intervals. To this end we insert into our population model with no ordinances the number of 2 year-old small juveniles loggerheads (stage 2) required to yield the same time to extinction (see Table 7) as under the scenario of "Ordinances" (245 years), and refer to this as the "Current Ordinances" program. Similarly, we can compute the cost of the required rearing program to having the equivalent effect of the maximum effective legislation, *i.e.*, achieving the scenario of "Ordinances max" (264 years to extinction), and call this the "Maximum Ordinances" program. Finally we consider the case of remaining at the current level of legal protection, but raising enough turtles to achieve the population dynamics if ordinances were set at their maximum level of effectiviness, termed the "Additional Ordinances" program.

Table 8: Replacement Cost (US$ millions)

	Current Ordinances		Maximum Ordinances		Additional Ordinances	
	FAU	NOAA	FAU	NOAA	FAU	NOAA
Yearly cost	26,715.3	62,120	130,480.2	303,400	16,686.3	38,800
% of Florida GDP	2.9	6.7	14.1	32.7	1.8	4.2

Notes: (a) Florida 2016 GDP. All monetary values are 2016 $US prices; (b) Florida GDP data comes from the Bureau of Economic Analysis; (c) Current Ordinances: no ordinances & captivity-rearing *vs.* current ordinances; (d) Maximum Ordinances: no ordinances & captivity-rearing *vs.* ordinances max.; (e) Additional Ordinances: current ordinances & captivity-rearing *vs.* ordinances max.

The results for the rearing costs based on the estimates of both the FAU and NOAA labs are provided in Table 8. As can be seen, regardless of which costing figure we use, the annual cost of implementing a rearing program instead of using STFL ordinances to "buy" a postponement of extinction by 22 years ("Current Ordinances") would arguably be large, roughly between 26.7 and 62.1 $US billion, or about 2.9 and 6.7% of annual

[37] $US 418 per turtle and year according to the FAU Marine Laboratory in 2014.

GDP in Florida, respectively. Moreover, the replacement cost of not setting the legislation at the maximum effectiveness ("Maximum Ordinances") would be about 5 times larger. Figures are of considerable magnitude too when we we consider the cost of a rearing program equivalent to the current level of STFL ordinances ("Additional Ordinances"), implying a replacement cost of between 1.8 and 4.2% of annual GDP.

More generally, the rather large costs associated with the different programs can mainly be explained by the fact that captivity-rearing programs of sea turtles are known to be very costly (see, among others, Heppell *et al.*, 1995 or Bell *et al.*, 2005), although possible economies of scale from larger programs might reduce the cost somewhat. Nevertheless, due to their biological characteristics – in particular, sea turtles are slow-maturing species – these programs would still require the introduction of a great amount of juveniles to make any sort of impact.[38]

6.2 Willingness to pay

A number of investigations have been conducted in the US and in Florida in order to measure the public WTP for the protection of sea turtles. One group of studies focuses on WTP at the household level. For instance, considering North Carolina, Whitehead (1992) estimates an average WTP to prevent loggerheads from becoming extinct of about $US 54.72 per household and year.[39] Assuming a similar figure for Florida then would imply that Florida households as a whole would in total be willing to pay $US 399.48 million per year.[40] Allowing for the presence of an alternative preservation program for threatened or endangered especies in general (*i.e.*, not only sea turtles), Whitehead (1993) identified a lower WTP for the protection of loggerheads. Accordingly, each household is willing to pay $US 18.09 per year, implying a total WTP of $US 132 million per year in Florida. Wallmo and Lew (2012) examine the WTP of households for the entire US for a 10-year recovering program of loggerheads, and their results imply a total yearly WTP in Florida of $US 693.7 million. Finally, specifically for Florida, Hamed (2013) estimates the WTP for a 5 years protection program of sea turtles' nesting habitat as ranging from

[38]For instance, in our simulations, the scenarios "Current Ordinances" and "Maximum Ordinances" require to free around 31 and 152 million juveniles per year, respectively, and about 19 millions juveniles per year for the scenario of "Additional Ordinances".

[39]Consistently with our estimates of replacement cost, all figures of WTP are expressed in 2016 US prices.

[40]As stated by the US Department of Commence (https://www.census.gov/quickfacts/FL), the population of Florida in 2016 is of 20,612,439 residents, comprising 7,300,494 households (average of 2011-2015).

$US 22.48 to 29.80 per household per year, suggesting a total WTP of between $US 164 and 217.5 million per year.[41]

Oceana (2008) follows a different approach by focusing on scuba divers' WTP for the greater likelihood of seeing a sea turtle during a dive, and discovers this to be about $US 33.5. Since the average number of dives per year of a scuba diver is 5, the WTP per diver per year would be $US 167.42. In considering what these figures would mean for Florida, one should note that according to the Diving Equipment and Marketing Association (DEMA) there are between 2.7 and 3.5 million of "active" scuba divers in the US,[42] while other sources provide a more conservative estimate of about 1.2 millions (see, for instance, http://undercurrent.org). DEMA also provides data about the number of newly certificates each year per state, where Florida's share of these is 9.57%. Using these figures suggests that the number of active scuba divers in Florida is between 258,390 to 334,495, or, more conservatively, around 114,840. The WTP provided by Oceana (2008) then implies that yearly WTP in Florida would be between $US 43.28 and 56 million, with a conservative estimate of $US 19.21 million.

We summarize our WTP calculations in Table 9. Comparing these with the cost figures in Table 8 one can see that the actual public WTP for the protection of sea turtles in Florida would not nearly cover the cost of our hypothetical rearing programs. This is the case even if we consider the highest estimated WTP, where we assume that all divers are non-Florida residents, so that we can sum the WTP of households implied by the estimates of Wallmo and Lew (2012) and that of scuba divers as calculated from the results of Oceana (2008), leading to a total WTP per year of $US 749.7 million. Considering the least expensive scenario of replacement costs, where enough turtles are raised to achieve the equivalent of increasing current STFL ordinances to their maximum level of effectiveness ("Additional Ordinances" in Table 8), public WTP only provides about 4.5% of the yearly funds needed for a rearing program.

[41] Hamed (2013) mainly concentrates on sea level rise threat and on two different towns in Florida (coastal *vs.* inland locations): Cocoa Beach and Oviedo.

[42] DEMA is a major non-profit international organization (www.dema.org) to promote recreational scuba diving and snorkeling industry. It is the main available data source for the number of scuba divers certificates, collecting information provided by the three main certification agencies (PADI, SDI and SSI).

Table 9: Yearly WTP (Willingness To Pay) in Florida

	Per household (US$)	Total (US$millions)	# turtles bred (FAU)	# turtles bred (NOAA)
Whitehead (1992)	54.72	399.48	464,445	199,740
Whitehead (1993)	18.09	132	153,466	66,000
Wallmo and Lew (2012)	95.02	693.7	806,512	346,850
Hamed (2013)	22.48 - 29.80	164 - 217.5	190,670 - 252,871	82,000 - 108,750
Oceana (2008)	33.5	43.28 - 56	50,318 - 65,107	21,640 - 28,000
Oceana (2008)(*)	33.5	19.21	22,334	9,605

Notes: (a) all figures in terms of 2016 US prices; (b) We consider 5-year equivalent protection programs; (c) WTP per scuba diver in Oceana (2008); (d) (*) Considering the conservative estimate of the number of active scuba divers; (e) Turtles bred corresponds to 2 year-old loggerhead small juveniles, provided the rearing costs per turtle based on the estimates of FAU and NOAA labs.

We can also use our population model from Section 5 in order to measure how many years of reduction in extinction public WTP in Florida could buy. As before, we consider hypothetical 5-year captivity-rearing programs of loggerheads and use the total yearly WTP in Table 9 to finance the cost of breeding turtles (2 year-old small juveniles). The implied number of released turtles per year are reported in the last 2 columns. Table A.4 in Appendix D provides the corresponding number of extinction years avoided. According to our simulations, the total WTP in Florida delays loggerheads' extinction by a maximum of 2 years. More precisely, if one were to use such funds for captivity-rearing programs instead of STFL ordinances, at best (*i.e.*, for the highest estimated WTP) extinction would be delayed by 1 year. If we consider captivity-rearing programs financed by WTP as a supplement to the current STFL legislation, loggerheads would gain a maximum of 2 extra years. The simulations show a similar delay even if one sets the legislation at its maximum effectiveness level across beaches together with WTP-financed captive rearing programs.

7 Concluding remarks

In this paper we investigated the effectiveness of using legislation to protect endangered species by examining to what extent lighting ordinances have limited the negative impact of light pollution on sea turtles in Florida. To this end we constructed an index of ordinance effectiveness across counties and combined this with loggerhead nesting data

and a set of rich controls to create a panel data set covering 26 years. Our econometric findings showed that legislation can significantly increase nesting activity, where current legislation results in an additional 34 per cent increase in nests. Using our estimates within a calibrated population model we also demonstrated that the current legislation aids sea turtles by extending the number of years to extinction by 22. The findings here thus arguably suggest that carefully crafted species protection legislation can potentially be an effective means of wildlife management.

It is important to point out that, as it stands, the level of protection with regard to beach front lighting appears to be still far from what it could or should be, particularly since sea turtles are believed to be an important point of attraction for tourists in Florida.[43] More precisely, on average counties in Florida have implemented only enough legislation to be 42 per cent effective, according to our constructed index. While we could not determine what the cost of implementing further or refining current legislation would be due to a lack of data in this regard, we were able to calculate the cost of achieving a similar effect by instead raising sea turtles in captivity to be released in the wild. To this end our framework indicated that the costs of such an alternative strategy would range between $US 17 and 39 billion per year, *i.e.*, between 1.8 and 4.2 per cent of Florida's GDP. Taking a range of willingness to pay estimates derived from the existing literature we show that the Florida public is, however, likely to be ready to only finance at most 4.5 per cent of such a program.

Finally, while our results suggest that at face value legislation is probably the lesser costly policy option than a head starting program, there are number of other cost and benefit factors that ideally would have been incorporated in our analysis. For one, since sea turtle lights generally project less luminosity than regular lighting, the loss in tourism due to the potential drop in beach safety at night during nesting season, or at least the perception thereof, should ideally be quantified (Witherington and Martin, 1996). There is also the cost of substituting regular beach front lighting with more sea-turtle friendly lighting that we were unable to consider due to a lack of data (see Ernest, 2002). In this regard it should, however, be noted that while sea turtle friendly lighting itself may not be inexpensive, it is more efficient and thus more energy-saving than regular illumination

[43]Stokes and Lowe (2013) state that, in 2011, about 11.5 million tourists visited Gulf states for wildlife viewing, spending 6.5 US$billion. According to Carr *et al.* (2016) over 500,000 tourists visit annually the coastal communities in the Southeast of the US to watch sea turtles. Moreover, there are 23 centers in Florida organizing sea turtles walks, which attract about 10,000 visitors per year.

and could, as noted by Witherington and Martin (2000) and Barshel *et al.* (2014), result in potentially significant reductions in the electricity bill of housing facilities and hotels.[44] Again, only further data could resolve this issue.

References

Allison P. and R. Waterman (2002), "Fixed effects negative binomial regression models", in *Sociological Methodology*, Ross M. Stolzenberg (ed.), Oxford: Basil Blackwell.

Anderson S., S. Nuernberge, K. Yamamoto and P. Sutton (2013), "Evaluating the compliance of sea turtle light ordinances in Florida using remote sensing", *Geography Compass*, 12, 867-878.

Barshel N., R. Bruce, C. Grimm, D. Haggitt, B. Lichter and J. McCray (2014), *Sea turtle friendly lighting. A model ordinance for local governments*, Levin College of Law, Conservation Clinic, University of Florida.

Bell C., J. Parsons, T. Austin, A. Broderick, G. Ebanks-Petrie and B. Godley (2005), "Some of them came home: the Cayman Turtle Farm headstarting project for the green turtle Chelonia mydas", *Oryx*, 19(2), 137-148.

Bjorndal K. and A. Meylan (1983), "Sea Turtles Nesting at Melbourne Beach, Florida, I. Size, Growth and Reproductive Biology ", *Biological Conservation*, 26, 65-77.

Boersma P., P. Kareiva, W. Fagan, J. Clark and J. Hoekstra (2001), "How Good are Endangered Species Recovery Plans", *Bioscience*, 51, 643-649.

Borkovic B. and L. Nostbakken (2017), "The cost of endangered species protection: Evidence from auctions for natural resources", *Journal of Environmental Economics and Management*, 81, 174-192.

Boyd J. (2014), "Best Available Science and Imperiled Species Conservation: Challenges, Opportunities, and Partnerships", *Resources for the Future Discussion Paper*, 14-44.

Brei M., A. Pérez-Barahona and E. Strobl (2016), "Environmental Pollution and Biodiversity: Light Pollution and Sea Turtles in the Caribbean", *Journal of Environmental Economics and Management*, 77, 96-116.

Broderick A., B. Godley and G. Hays (2001), "Trophic status drives interannual variability in nesting numbers of marine turtles", *Proceedings of the Royal Society*, 268, 1481-1487.

[44]Ernest (2002) reports that the Florida Power and Light Company estimated for 2001 that, per light, replacing non-cutoff with cutoff lighting cost US$ 674, repositioning head to redirect light US$ 300, reducing the wattage US$ 675, lowering the mounting height 300 US$, installing amber filter lens US$ 280, and turning the lights off for nesting season US$ 0. However, Barshel *et al.* (2014), observe that the general principles "Keep it Low, Keep it Long, and Keep it Shielded" imply significant savings on outdoor electricity bills. They point out, as an example, that La Playa Condominiums in Satellite Beach (Florida) realized a nearly 70% cost savings after installing STFL.

Brost B., B. Witherington, A. Meylan, E. Leone, L. Ehrhart and D. Bagley (2015), "Sea turtle hatchling production from Florida (USA) beaches, 2002-2012, with recommendations for analyzing hatching success", *Endangered Species Research*, 27, 53-68.

Bustard H. (1967), "Mechanism of nocturnal emergence from the nest in green turtle hatchlings", *Nature*, 244-317.

Cameron A. and P. Trivedi (2013), *Regression Analysis of Count Data*, Cambridge University Press, New York.

Carr B., S. Emmert, P. Mustain and L. Snyder (2016), "TEDs for All Trawls: A Net Positive for Fishermen and Sea Turtles", *Oceana Reports*, May 2016.

Crouse D., L. Crowder and H. Caswell (1987), "A Stage-Based Population Model for Loggerhead Sea Turtles and Implications for Conservation", *Ecology*, 68(5), 1412-1423.

Crowder L., D. Crouse, S. Heppell and T. Martin (1994), "Predicting the Impact of Turtle Excluder Devices on Loggerhead Sea Turtle Population", *Ecological Application*, 3(4), 437-445.

Davenport J. J. and Davenport (2006), "The impact of tourism and personal leisure transport on coastal environments: a review", *Esturine, Coastal and Shelf Science*, 67, 280-292.

Driscoll J. and A. Kraay (1998), "Consistent Covariance Matrix Estimation with Spatially Dependent Panel Data", *Review of Economics and Statitistcs*, 80, 549-560.

Duffy L. (2016a), "United States' Sentencing Memorandum, case 15-cr-2867-JLS", *Government's Sentencing Memorandum*, 15-cr-2867-JLS, 1-15.

Duffy L. (2016b), "United States' Sentencing Memorandum, case 15-cr-2867-JLS, supplemental sentencing", *Government's Sentencing Memorandum*, 15-cr-2867-JLS, supplemental sentencing, 1-28.

Ernest R. (2002), *Coastal roadway lighting manual*, Ecological Associates, Inc., Jensen Beach, Florida.

EF (2016), "Do Habitat Conservation Plans Deserve Wider Implementation", *The Environmental Form*, May/June 2016 Issue, 48-57.

Ferraro P., C. McIntosch 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.

Frazer. N. (1983), "Demography and life history evolution of the Atlantic loggerhead sea turtle, *Caretta caretta*", Dissertation. University of Georgia, Athens, Georgia. USA.

Freeman A. (2003), *The Measurement of Environmental and Resource Values*, Resources for the Future, Washington D.C.

Gibbs K. and D. Currie (2012), "Protecting Endangered Species: Do the Main Legislative Tools Work?", *PLOS ONE*, May 2012.

Mazaris A., G. Matsinos and J. Pantis (2009), "Evaluating the impacts of coastal squeeze on sea turtle nesting", *Ocean and Coastal Management*, 52, 139-145.

Hamed A. (2013), "Economic Valuation Of Florida Sea Turtles In Face Of Sea Level Rise", *Electronic Thesis and Dissertations, University of Central Florida*, Paper 2750.

Heppell S., L. Crowder and J. Priddy (1995), *Evaluation of a fisheries model for the harvest of Hawksbill Sea Turtles, Eretmochelys imbricata, in Cuba*, NOAA Technical Memorandum NMFS-OPR-5, U.S. Department of Commerce, Washington, DC.

Heppell S., M. Snover and L. Crowder (2003), "Sea Turtle Population Ecology", in Lutz P., J. Musick, and J. Wyneken, Eds, *The biology of sea turtles*, CRC marine science series, Boca Raton, Florida.

Houtan K. and O. Bass (2007), "Stormy oceans are associated with declines in sea turtle hatching", *Current Biology*, 17, 590-591.

Jin J., A. Indabb, O. Nabangchangc, T. Thuyd, D. Harderb and R. Subadee (2010), "Valuing marine turtle conservation: A cross-country study in Asian cities", *Ecological Economics*, 69(10), 2020-2026.

Johnson S., K. Bjorndal and A. Bolten (1996), "Effects of organized turtle watches on loggerhead (Caretta caretta) nesting behaviour and hatchling production in Florida", *Conservation Biology*, 10, 570-577.

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.

Krippendorff K. (2013), *Contenct Analysis, An Introduction to Its Methodology*, 3th edition, SAGE Publications, Inc, Thousand Oaks, CA, USA

Langpap C. and J. Kerkvliet (2012), "Endangered species conservation on private land: Assessing the effectiveness of habitat conservation plans", *Journal of Environmental Economics and Management*, 64, 1-15.

Langpap C., Kerkvliet J., and Shogren, J. (2018), "The Economics of the US Endangered Species Act: A Review of Recent Developments", *Review of Environmental Economics and Policy*, 12, 69-91.

Lorne K. and K. Salmon (2007), "Effects of exposure to artificial lighting on orientation of hatchling sea turtles on the beach and in the ocean", *Endangered Species Research*, 3, 23-30.

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

Miller J. (1997), "Reproduction in sea turtles", in P. Lutz and J. Musick, Eds, *The biology of sea turtles, vol.1*, CRC Press, Boca, Fl, 51-81.

Murphy T. and S. Hopkins-Murphy (1990), "Homing of translocated gravid loggerhead turtles", in T. Richardson, J. Richardson, and M. Donnelly, Eds, *Proceedings 10th annual symposium on sea turtle biology and conservation*, NOAA Technical Memorandum NMFS-SEFSC-278, Miami, Fl, 123-124.

Phillips K., K. Mansfield, D. Die and D. Addison (2014), "Survival and remigration probabilities for loggerhead turtles (*Caretta caretta*) nesting in the eastern Gulf of Mexico", *Marine Biology*, 161, 863-870.

Oceana (2008), "Sea the Value: Quantifying the Value of Marine Life to Divers", *Oceana Reports*, August 2008, http://oceana.org/reports/sea-value-quantifying-value-marine-life-divers

Raymond, P. (1984), *Sea Turtle Hatchling Disorientation and Artificial Beachfront Lighting*, Center for Environmental Education, Washington, DC, USA.

Richards P., S. Epperly, S. Heppell, R. King, C. Sasso, F. Moncada, G. Nodarse, D. Shaver, Y. Medina and J. Zurita (2011), "Sea turtle population estimates incorporating uncertainty: a new approach applied to western North Atlantic loggerheads *Caretta caretta*", *Endangered Species Research*, 15, 151-158.

Roman, J. (2011), *Listed: Dispatches from America's Endangered Species Act*, Harvard University Press, Boston.

Rumbold D., P. Davis and C. Perretta (2001), "Estimating the Effect of Beach Nourishment on *Caretta caretta* (Loggerhead Sea Turtle) Nesting ", *Restoration Ecology*, 9, 304-310.

Sedjo R. (2008), *Perspectives on Sustainable Resources in America*, Resources for the Future, Routledge, Washington DC.

Stokes S. and M. Lowe (2013), *Wildlife Tourism and the Gulf Coast Economy*, Datu Research, LLC, Durham, NC.

Troeng S. and C. Drews (2004), *Money Talks: Economic Aspects of Marine Turtle Use and Conservation*, WWF-International, Gland, Switzerland, www.panda.org

Tuxbury S. and M. Salmon (2005), "Competitive interactions between artificial lighting and natural cues during sea finding by hatchling marine turtles", *Biological Conservation*, 121, 311-316.

USFWS (2015), *Federal and State Endangered and Threatened Species Expenditures: Fiscal Year 2015*, U.S. Fish and Wildlife Service, Washington DC.

Verheijen F. (1985), "Photopollution-artificial light optical spatial control systems fail to cope with incidents, causations, remedies", *Experimental Biology*, 44(1), 1-18.

Wallmo K. and D. Lew (2012), "Public Willingness to Pay for Recovering and Downlisting Threatened and Endangered Marine Species", *Conservation Biology*, 26(5), 830-839.

Whitehead J. (1992), "Ex Ante Willingness to Pay with Supply and Demand Uncertainty: Implications for Valuing a Sea Turtle Protection Program", *Applied Economics*, 24, 981-988.

Whitehead J. (1993), "Total Economic Values for Coastal and Marine Wildlife: Specification, Validity, and Valuation Issues", *Marine Resource Economics*, 8, 119-132.

Witherington B. (1992), "Behavioral responses of nesting sea turtles to artificial lighting", *Herpetologica*, 48, 31-39.

Witherington B. and N. Frazer (2003), "Social and Economic Aspects of Sea Turtle Conservation", in P. Lutz, J. Musick, and J. Wyneken, Eds, *The Biology of Sea Turtles*, CRC marine science series, Boca Raton, Florida.

Witherington B. and R. Martin (1996), "Understanding, assessing, and resolving light-pollution problems on sea turtle nesting beaches", *Florida Marine Research Institute Technical Report*, TR-2.

Witherington B. and R. Martin (2000), "Understanding, assessing, and resolving light-pollution problems on sea turtle nesting beaches, second edition". *Florida Marine Research Institute Technical Report*, TR-2, second edition, revised 2000.

Witherington B., P. Kubilis, B. Brost and A. Meylan (2009), "Decreasing annual nest counts in a globally important loggerhead sea turtle population", *Ecological Applications*, 19(1), 30-54.

Woolridge J. (2015), "Control Function Methods in Applied Econometrics", *Journal of Human Resources*, 50, 420-445.

Worth D. and J. Smith (1976), "Marine turtle nesting on Hutchinson Island, Florida, in 1973", *Florida Marine Research Publication*, 18, 1-17.

Zhang D. (2004), "Endangered Species and Timber Harvesting: The Case of Red-Cockaded Woodpeckers", *Economic Enquiry*, 42, 150-165.

Appendices

A Legislation statements

Barshel *et al.* (2014) identify 17 statements, called "Sea Turtle Friendly Lighting Principles Component", that define favourable conditions for sea turtle nesting. The set "Implementation Component" considers 9 statements in order to measures to what extent the ordinance ensures such nesting conditions. We provide below a detailed list of these statements.

Sea Turtle Friendly Lighting Principles Component (17 items):

1.- Exterior artificial light for existing development must be low lumens.

2.- Exterior artificial light for existing development must be full cut off (*ex.*, no light emitted above 90 degree angle).

3.- Exterior artificial light for existing development must be downward directed.

4.- Exterior artificial light for existing development must not be visible from the beach.

5.- Exterior artificial light for existing development must be long wavelength (*i.e.*, 580 nm or greater).
6.- Exterior artificial light for existing development must be shielded.
7.- Exterior artificial light for new development must be low lumens.
8.- Exterior artificial light for new development must be full cut off (*ex.*, no light emitted above 90 degree angle).
9.- Exterior artificial light for new development must be downward directed.
10.- Exterior artificial light for new development must not be visible from the beach.
11.- Exterior artificial light for existing development must be long wavelength (*i.e.*, 580 nm or greater).
12.- Exterior artificial light for new development must be shielded.
13.- Artificial light shall not be visible (*ex.*, directly/indirectly/cumulatively) from the beach.
14.- Areas seaward of the frontal dune are not to be directly illuminated.
15.- Areas seaward of the frontal dune are not to be indirectly illuminated.
16.- Areas seaward of the frontal dune are not to be cumulatively illuminated.
17.- The building of campfires or bonfires shall be prohibited during the nesting season.

Implementation Component (9 items):

1.- Is a provision made for a compliance inspection during the nesting season?
2.- Does the ordinance provide for a pre-enforcement warning?
3.- Does the ordinance provide for a notice of violation?
4.- The ordinance establishes civil penalties for non-compliance.
5.- The ordinance establishes criminal penalties for non-compliance.
6.- Are the enforcement penalties incorporated into the ordinance by reference?
7.- Shall each day of any such violation constitute a separate and distinct offense?
8.- Does the ordinance provide for the education of the general public?
9.- Does the ordinance provide for the education of the affected public (*ex.*, those submitting an application for construction)?

B Stage-based life table

Crowder *et al.* (1994) set five-stage life history parameters for loggerheads. This is based on the stage-based life table of Crouse *et al.* (1987), which is built on data from Frazer *et al.* (1983):

Table A.1: Loggerhead Sea Turtles

Stage	Description	Stage duration in years (d_i)	Annual survival rate (σ_i)	Annual fecundity (F_i)
1	Eggs/Hatchlings	1	0.6747	0
2	Small juveniles	7	0.75	0
3	Large juveniles	8	0.6758	0
4	Subadults	6	0.7425	0
5	Adults	>32	0.8091	76.5

C Matrix L properties

Taking the information in Table A.1, we can compute the eigenvalues of L for the loggerheads in Florida. We also provide these values for the modified L when we consider the effect of coastal ordinances on the annual fecundity:

Table A.2: Eigenvalues

	λ_1	λ_2	λ_3	λ_4	λ_5
No ordinances	0.9281	0.7318+0.2037i	0.7318-0.2037i	0.4744	0.0059
Ordinances	0.9344	0.7328+0.2110i	0.7328-0.2110i	0.4651	0.0069
Ordinances max.	0.9388	0.7336+0.2161i	0.7336-0.2161i	0.4585	0.0076

Since λ_1 is the dominant eigenvalue, we calculate the stable stage distribution (11) by considering the corresponding eigenvector v_{λ_1}:

Table A.3: Long-run Stage Distribution, No Ordinances

v_{λ_1} ($v_{\lambda_1 k}$)	Stage	Description	Distribution (%)	x_0 (x_{k0})
0.3028	1	Eggs/Hatchlings	21.72	3159771
0.9432	2	Small juveniles	67.65	9842492
0.1360	3	Large juveniles	9.76	1419475
0.0084	4	Subadults	0.61	88162
0.0037	5	Adults	0.26	38334

D WTP: extinction years

Table A.4: Number of Extinction Years Avoided

	Substitute		Complement		Complement max.	
	FAU	NOAA	FAU	NOAA	FAU	NOAA
Whitehead (1992)	< 1	< 1	1	1	1	1
Whitehead (1993)	< 1	< 1	1	1	< 1	< 1
Wallmo and Lew (2012)	1	< 1	2	1	2	1
Hamed $(2013)_l$	< 1	< 1	1	1	1	< 1
Hamed $(2013)_u$	< 1	< 1	1	1	1	< 1
Oceana $(2008)_l$	< 1	< 1	< 1	< 1	< 1	< 1
Oceana $(2008)_u$	< 1	< 1	1	< 1	< 1	< 1
Oceana (2008)(*)	< 1	< 1	< 1	< 1	< 1	< 1
Whitehead (1992)(**)	< 1	< 1	1	1	1	1
Whitehead (1993)(**)	< 1	< 1	1	1	1	< 1
Wallmo and Lew (2012)(**)	1	< 1	2	1	2	1
Hamed $(2013)_l$(**)	< 1	< 1	1	1	1	< 1
Hamed $(2013)_u$(**)	< 1	< 1	1	1	1	< 1

Notes: (a) Substitute: captivity-rearing as substitute for current STFL ordinances; (b) Complement: captivity-rearing as complement to current STFL ordinances; (c) Complement max: captivity-rearing as complement to max. STFL ordinances; (d) subindex l for WTP lower bound; (e) subindex u for WTP upper bound; (f) (*) Considering the conservative estimate of the number of active scuba divers; (g) (**) WTP of households + WTP of scuba divers in Oceana (2008).