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Science Information to Support Missouri River Piping Plover and Least Tern Effects Analysis

October 2015

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US Army Corps of Engineers

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Summary

This document contains the reviews of literature and models and assessment of available data conducted for the Missouri River Effects Analysis, a process which is being used to assess the effects of current and potential management actions on piping plovers and least terns on the Missouri River main stem. As part of ensuring the best available science is used during the effects analysis and management planning process, we reviewed and summarized the published literature, including research studies and models, about least terns and piping plovers. This document contains summaries of the review process and findings, as well as appendices with tables containing details of each study we reviewed. It also includes a description of the types of data that have been collected for further analysis and their intended use.

Acronyms and Abbreviations

AM	Adaptive Management
EA	effects analysis
ESH	Emergent Sandbar Habitat
ISAP	Independent Science Advisory Panel
MRRIC	Missouri River Recovery Implementation Committee
NPWRC	Northern Prairie Wildlife Research Center
PNNL	Pacific Northwest National Laboratory
PVA	population viability analysis
TPMP	Tern and Plover Monitoring Program
USACE	U.S. Army Corps of Engineers
USGS	U.S. Geological Survey

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1.0 Introduction

This document was prepared by Pacific Northwest National Laboratory (PNNL) as part of the Missouri River Effects Analysis (EA) for the Missouri River Recovery Program. The EA was initiated in October 2013 by the U.S. Army Corps of Engineers (USACE) and the U.S. Fish and Wildlife Service at the recommendation of the Missouri River Recovery Implementation Committee (MRRIC) and the MRRIC Independent Science Advisory Panel (ISAP). The purpose of the EA is to evaluate the effectiveness of past and current management actions implemented by the USACE to avoid jeopardy to piping plovers, least terns, and pallid sturgeon, which are listed as threatened or endangered under the U.S. Endangered Species Act.

The EA process, as informed by Murphy and Weiland (2011) and the ISAP, includes four steps: 1) develop comprehensive conceptual ecological models linking the Missouri River dynamics, management, and their drivers to the demographic processes and population dynamics of the three species; 2) compile existing scientific literature, data, and models and the assessment of their application and use to the EA; 3) develop predictive, quantitative models of the effects of management actions on the species; and 4) analyze and assess the effects of system operations and actions on species' habitat and populations. Central to the EA process is the need for identification and use of the best available science, including the identification of information and models that are "pertinent, reliable, and significantly robust" to populate the models used to evaluate management actions (Murphy and Weiland 2011).

The collection and analysis of information for the EA is intended to assess the uncertainty in the knowledge of the system that affects the ability to develop conceptual and numerical models, make predictions, and assess management hypotheses. Uncertainty in ecological systems such as the Missouri River can be categorized into four types:

1. **Environmental variability.** Future climate and weather cannot be known with certainty. The greatest source of environmental variability on the Missouri River is basin runoff and resulting system storage and flows, which in turn are a major driver of habitat availability. Local weather, including storms and heat waves, also affect reproduction and survival. This type of uncertainty is largely irreducible, though advances in modeling and data sets can lead to better estimates of the likely distribution of future conditions.
2. **Structural uncertainty.** Even when fundamental relationships between hydrology, habitat availability, and population dynamics are generally understood, uncertainty remains about the details and functional form of many relationships. Examples include: What aspects of habitat quality affect fledgling production? What factors best predict reproductive success on reservoirs? How do metapopulation dynamics affect the Missouri River population? Structural uncertainty can be reduced through research, monitoring, and improvements to models.
3. **Parametric uncertainty.** Once the structure of relationships is known, uncertainty remains about the strength of the relationships. For example, what is the survival rate for birds during their first winter? How strongly does population density affect chick survival? As with structural uncertainty, these uncertainties can be reduced through research and monitoring and incorporated into models. However, relationships and degrees of uncertainty may vary over time and location such that they are unlikely to ever be perfectly known.
4. **Observation uncertainty:** Population and productivity surveys are not 100% accurate. The degree of error and direction of bias can vary by habitat type and conditions and level of effort, and thus may differ across both space and time. The design and level of effort in a monitoring program can reduce observation error and, in some designs, estimate the error in the survey, which allows for more accurate use of the resulting information.

This document was developed as part of Step 2 of the EA process described above, concurrent with the development of the conceptual ecological models for piping plovers and least terns. It focuses primarily on data that address structural and parametric uncertainty. It consists of reviews and summaries, in table form, of the published literature about the two species, including research and modeling studies. It also includes a description of the data sets that are included for analysis and their intended use.

The materials included in this document were developed between October 2013 and September 2014. A draft report was shared with MRRIC and the ISAP for their review on March 19, 2014. Further information about how this material was used in the EA process was included in the draft integrated interim report, shared for review in October 2014.

2.0 Literature Review

The basis of our literature review was a collection of journal articles, reports, dissertations, and book sections that we have been compiling over the past 5 years. The collection is continually updated as new literature is published. Most of the journal articles were published between either 2000 and 2009 (n = 41 and 21 for plovers and terns, respectively) or between 2010 and 2014 (n = 38 and 14 for plovers and terns, respectively; Figure 1). The collection includes 107 articles about piping plovers and 63 about least terns. We continue to conduct additional searches for publications, follow references in reviewed papers, and compare our citations with other bibliographies to ensure that we have thorough coverage of the available literature.

All of the literature in our library addressing piping plovers and least terns has been summarized in table format followed by the literature cited for each table (Appendix A and B). Tables were created to cover major categories based on the topics of each study: reproductive success, nest-site selection, nest density, habitat quantity and quality, prey, predation, stage survival, site fidelity, immigration and emigration, overwintering habitat, population trends, genetics, and detection probabilities. A journal article may be cited in one or more tables depending on the results presented. Tables include a short citation including the year of publication, the study location (e.g., Missouri River, Great Lakes, Platte River), sample size of the study, years it was conducted, and study design. The plover literature includes 38 studies conducted on populations in the Missouri River and associated lakes (Figure 2) and 49 studies conducted on the east coast of the United States, Canada, and the Great Lakes. A large part of the tern literature consists of studies conducted on populations in the Mississippi River and on the California coast, with 12 publications including terns on the Missouri River. The tables also include results for each major topic presented in the article including tables and figures. For example, reproductive success was the most common topic for which results were presented for both species in our collection (Figure 3). For each table, the results are presented in columns associated with subtopics specific to that category (e.g., the reproductive success category has columns for information about nest initiation, nest success, nest fates, and the number of fledglings/nest).

The model parameter tables (Appendix C) were developed to allow for easy access to published estimates of piping plover and least tern performance (reproduction, survival, and movement over the landscape) in response to changes in habitat, supporting resources, and inter- and intra-species interactions. The tables were populated with the specific parameter estimates or calculated estimates based on the results presented in the tables or figures of reviewed literature. The current parameter set for both species includes annual survival by stage (juvenile and adult plovers, juvenile, young adult, and older adult terns); the linear slope and intercept of the number of fledglings/pair as a function of density; variability in the regression relationship of the number of fledglings/pair as a function of density; the probability of

detection of fledglings and adults; inter- and intra-reach dispersal by life stage; and nest fate (success, predation).

For each reviewed publication that presents parameter estimates or data that can be used to estimate parameters, a decision was made about whether or not the findings would be appropriate to use in the numerical population models. Both the short citation and the reason that the literature was not used are presented at the bottom of each parameter table (Appendix C). The reasons why journal articles that reported data or parameter estimates for survival or reproductive responses were not included were not related to data quality issues (Figure 5). Rather, studies were not used because the research was conducted more than 20 years ago or at a location far from the Missouri River. Research results may have been site-specific or they may have addressed specific effects such as cage or banding effects on mortality. We also did not include articles in the parameter tables that cite parameter values from other studies rather than present new data. Finally, studies that addressed a time-series response for which the data are only presented as a figure and are not readily extracted for parameter estimation were also not included.

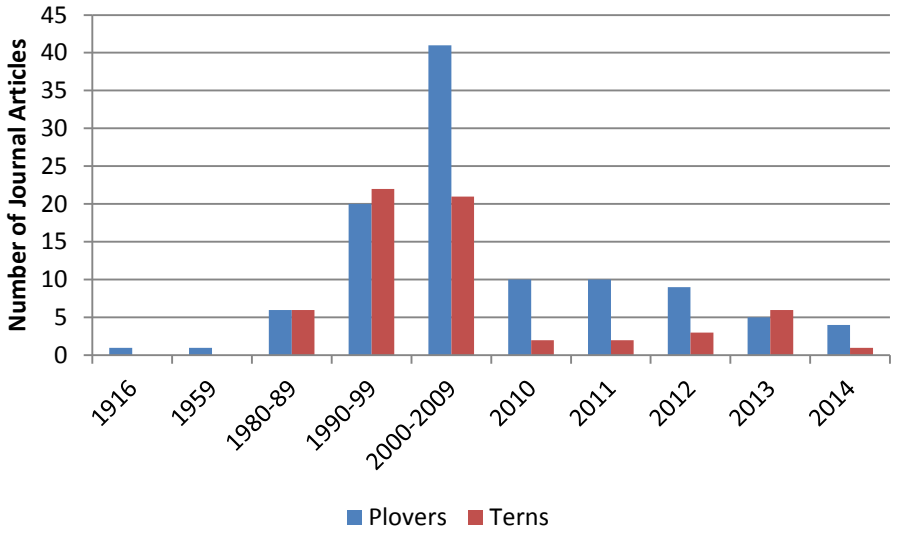

Figure 1. Distribution of Journal Article Years of Publication in the Collection

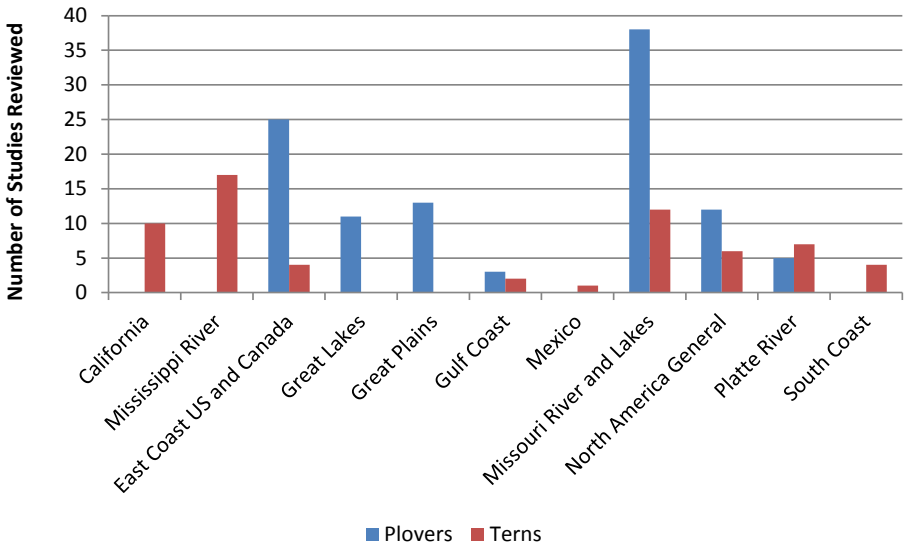

Figure 2. Distribution of Journal Article Study Locations in the Collection

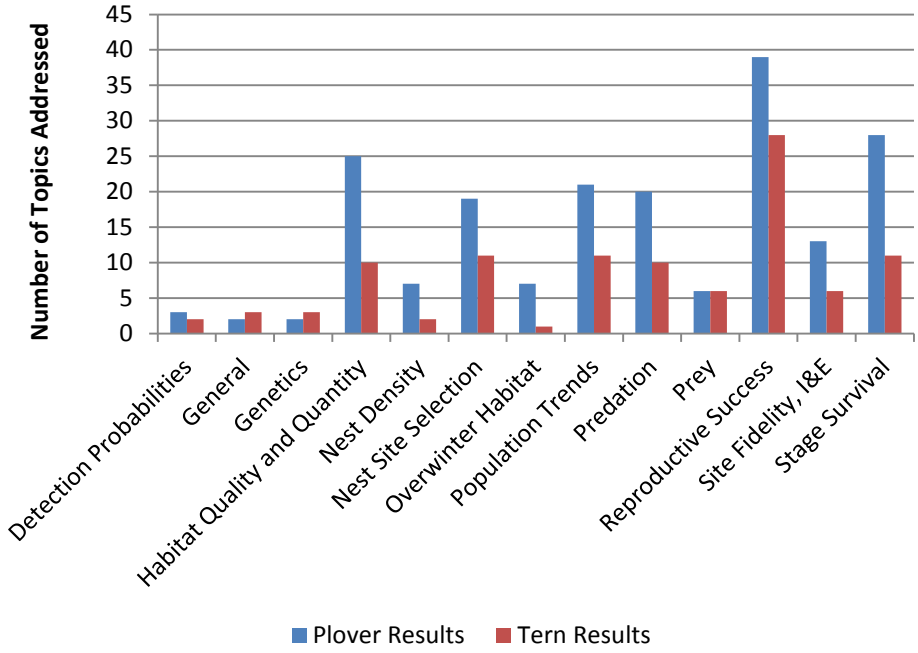

Figure 3. Distribution of Journal Article Study Topics in Collection

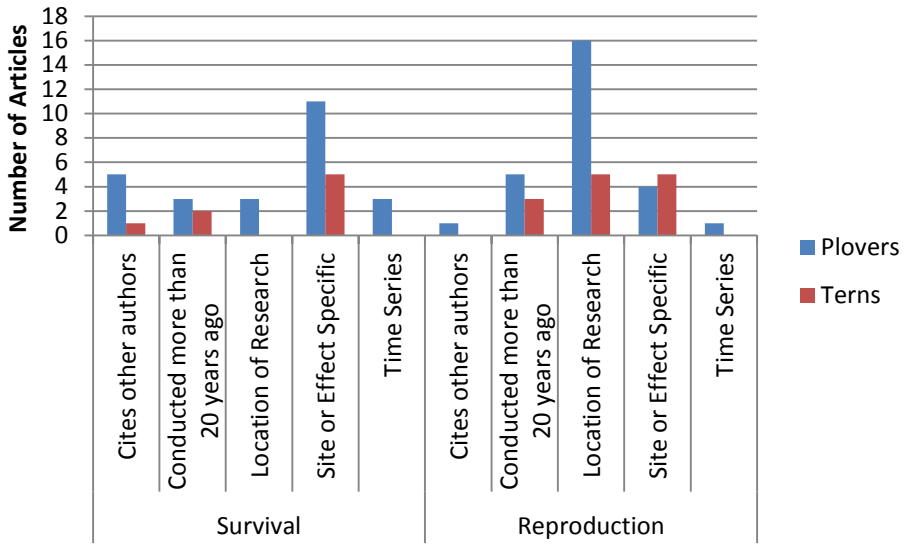

Figure 4. Number of Journal Articles Reviewed That Report Results Associated with Survival or Reproduction but Were Not Used for the Listed Reason in the Effects Model

3.0 Model Review

We collected and reviewed published models for piping plovers and least terns for three primary purposes: to compile information about approaches and techniques used for modeling these species, to compile data sources used in the models, and to identify additional evidence relevant to the species and management hypotheses developed in the EA. This review will inform numerical model development and help ensure the best available science is incorporated into the analysis and findings. While this does not serve as a comprehensive review of population modeling techniques, the reviewed models include a range of approaches to population viability analysis (PVA), which is recommended as a central component of an EA (Murphy and Weiland 2011).

We included in our review the journal article resulting from the modeling done for the Missouri River Recovery Program's Emergent Sandbar Habitat Adaptive Management (ESH AM) sub-program (Buenau et al. 2014). The model as published was a slightly simplified version of the full model used for ESH AM; the full technical description and application of this model can be found in the ESH Annual Adaptive Management Report (USACE 2013).

3.1 Plover Population Models

We reviewed 12 published model-based studies of piping plover population dynamics (Appendix D). All but two studies included piping plovers nesting on the Missouri River; the other two modeled populations on the Great Lakes (Wemmer et al. 2001) and in Eastern Canada (Calvert et al. 2006).

The majority of the reviewed modeling studies were based upon Ryan et al. (1993), a traditional PVA including fledgling production and juvenile and adult survival. Exceptions were the model for Great Lakes plovers, which included a more detailed set of mechanisms describing pairing and site selection (Wemmer et al. 2001); Calvert et al. (2006), which is structured as a standard matrix projection model with three life stages; and Buenau et al. (2014), which is also a PVA but designed differently than Ryan et

al.'s (1993). Plissner and Haig (2000) focused specifically on metapopulation dynamics including both Great Plains and Atlantic Coast plovers.

Some of the reviewed models implicitly included habitat quality at different locations by varying reproductive parameters by site (Wemmer et al. 2001; Calvert et al. 2006) or by habitat type, distinguishing between river and alkaline wetland habitats (Larson et al. 2002, 2003) and between Missouri River, other Great Plains rivers, and alkaline wetland habitats (McGowan and Ryan 2009; McGowan et al. 2011b; McGowan 2013; McGowan et al. 2014). Two models accounted for habitat area: Wemmer et al. (2001) included the number of nesting sites, allowing for acquisition of new habitat or exclusion of privately owned land; and Buenau et al. (2014) modeled the dynamics of sandbar habitat per river segment, with area affected reproductive rates through a density-dependence relationship. None of the models was spatially explicit, although Wemmer et al. (2001) included distance between sites as being relevant to dispersal and site selection.

Seven of the studies included some form of management action: protection, restoration and acquisition of shoreline habitat (Wemmer et al. 2001); predation management using nest cages and temporary or permanent fences (Larson et al. 2002, 2003); management of incidental take (McGowan and Ryan 2009; McGowan et al. 2011b; McGowan 2013); and sandbar habitat creation (Buenau et al. 2014). Results from these studies can be used as lines of evidence for evaluating the effects of management actions that are the same as or similar to the ones modeled.

Several models included features potentially useful for evaluating the effects of management actions and for AM. Models that break down the reproductive process into multiple steps (Wemmer et al. 2001; McGowan and Ryan 2009; McGowan 2013) facilitate the modeling of management actions affecting specific life stages (e.g., reductions in incidental take of eggs) if sufficient data are available to parameterize those functions. McGowan et al. (2011b) examined whether density dependence in juvenile overwinter survival could compensate for loss of eggs through incidental take and found low support for that hypothesis; they also suggested density dependence was more likely to occur during breeding than during overwinter survival. Buenau et al. (2014) explicitly included density dependence in fledgling production to determine the effects of river stage and habitat availability on population dynamics. Wemmer et al. (2001) included density dependence indirectly by including a finite number of nesting sites.

McGowan et al. (2011a) used a plover PVA to illustrate the importance of incorporating parametric uncertainty and a hierarchical process for doing so. Calvert et al. (2006) and Buenau et al. (2014) also included stochasticity in parameter estimates. Calvert et al. (2006) discussed the role of transient dynamics in modeled populations that have not yet reached a stable age structure. Those transient dynamics can affect population responses to disturbance or management and make interpretation of the responses more challenging.

Two papers used PVA models to demonstrate tools that can inform management decisions. One tool, linear programming modeling, optimizes management by determining which actions maximize population outcomes while also minimizing costs (Larson et al. 2003). McGowan (2013) used his previously published plover PVA to illustrate the PrOACT structured decision-making process applied to managing incidental take, first with a single objective of plover population growth, then with a competing objective of pallid sturgeon population responses. McGowan et al. (2014) used a metapopulation PVA to demonstrate a process for using population projection models to determine recovery targets.

3.2 Tern Population Models

There are considerably fewer published studies involving population models of least terns than of piping plovers; we identified four for review (Appendix E). Akcakaya et al. (2003) published a metapopulation study of California least terns using stochastic age-structured models for 17 populations. The model included local catastrophes (reproductive failure for a subpopulation in that year) due to predation. Predator control was implemented in the model by reducing the probability of local catastrophe to zero. Differences between the Interior and California least tern habitats and management scenarios limit the applicability of this model as a line of evidence for the EA.

Two studies (Lott et al. 2012a, 2012b; Kanapaux and Kiker 2013) developed agent-based, pattern-oriented models of least terns. These studies modeled site selection and nesting behavior and productivity for terns on an individual basis and linked their behavior to specific spatial locations. Kanapaux and Kiker (2013) used their model to examine the impact of human activity on terns nesting on a South Carolina beach and to test the effects of several human access restriction actions on tern productivity. Such a model is not a full population viability model because it only models reproductive success during a season, not changes in population size over a longer time frame. The TernCOLONY model uses a similar individual-based approach to examine the role of flooding of sandbar habitat in tern abundance and reproductive success, and also operates on the temporal scale of a breeding season with daily time steps. The TernCOLONY model, described by Lott et al. (2012b), was tested on the Arkansas River.

The approach used by Buenau et al. (2014) is analogous to that described above for plovers: it is a stochastic, stage-based PVA that models population responses to changes in sandbar habitat availability on the Missouri River, including habitat created by construction or through high flows. The primary differences between the plover and tern model are the stage structure of the population and the very weak density dependence observed in terns at the sandbar scale compared to plovers.

A metapopulation model for Interior least terns is currently under development, but details of that model were not available for this review.

4.0 Data Review

We have assessed data acquisition and analysis needs for the EA by evaluating potential data sources from literature and scrutinizing existing data. Data mined from, and referenced in, literature have been used to diagnose observed bias and potential error associated with existing USACE Tern and Plover Monitoring Program (TPMP) data. The error structure in the data from the TPMP is unknown and varies with space and time, according to a U.S. Geological Survey (USGS) study and analysis of the program (Shaffer et al. 2013). Use of these data in our analysis warrants careful consideration of how to use them to limit error propagation and to understand the implications of variable observation error on results. The following sections describe our approach to retaining valuable information derived from the data that have been collected throughout the TPMP and integrating newly available data into our analysis.

4.1 USACE-Owned/Managed Data

Population data from the USACE's TPMP are available beginning in 1993. This data set includes nest numbers and fates; incidental take of nests, eggs, and chicks; fledgling numbers; and adult numbers, all linked to specific sandbars and summarized by reach.

The USACE has provided PNNL with ESH classifications and land-cover data, nest location data sets, and other ancillary physical habitat data for years 2006−2014. We are using the remotely sensed land-cover data, classified by the USGS-Northern Prairie Wildlife Research Center (NPWRC), together with population data to estimate the relationship between habitat area, population density of nesting birds, and fledgling production. We are also using spatial data to assess the quality of habitat with respect to the composition and connectivity of bars and islands, sandbar persistence, and distance to common predator habitat (large tree stands). Evaluating bars and islands in terms of quality will allow us to relate what is known about individual species' predation risks and needs (e.g., plover foraging habitat) to potential habitat area that is not solely confined to previously classified ESH. Because the USACE has also provided nest location data, we will be able to determine nest concentration and species selectivity as they relate to habitat quality. Measures of the variability of previous ESH classifications and other remotely sensed data are also being accounted for using data derived from areas where imagery collection overlaps.

Supplementary analysis of physical habitat variables to determine the characteristics and abundance of sandbar habitat is ongoing. Data used to determine channel width and braiding index were provided by the USACE and USGS and are being used in combination with publicly available National Hydrography Dataset data and other modern channel delineations. Pertinent physical variables can potentially become indicators of species-specific nest selectivity.

4.2 Data from USACE-Funded Research

Researchers from the NPWRC, Virginia Polytechnic Institute, and State University (Virginia Tech) have conducted a range of USACE-funded research on both plovers and terns on the Missouri River. We have requested data corresponding to years 2006−2009 that were collected by Virginia Tech and the NPWRC (Catlin and Fraser 2006, 2007; Catlin 2009; Felio et al. 2009, 2010). These data will aid in creating an estimated ranking of habitat type preferences for nest location and the associated nest fate and reproductive output. Other habitat classification and change data provided will potentially be used to calculate erosion and vegetation rates and evaluate post-2011 sandbar change.

We have also received updated demographic information, including information about survival, reproduction, and dispersal, from the researchers. We have also discussed with NPWRC the availability of additional geomorphic information including updated erosion and vegetation rates.

4.3 Other Data

Banding and encounter data for least tern and piping plover species were received on November 22, 2013, from Matthew Rogosky at the Patuxent Wildlife Research Center in Laurel, Maryland. Data were obtained for years ranging from 1923 to 2013 with some missing years prior to 1950. The banding and encounter data were acquired to potentially evaluate dispersal if more information is needed than can be found in published studies and estimates provided by researchers.

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Appendix A

Literature Review Table for Plovers

Appendix A

Literature Review Table for Plovers

Tabulated results from the literature review for plovers are presented by the following categories:

- Reproductive Success
- Invertebrate Prey
- Nest Site Selection
- Nest Density
- Habitat Quality and Quantity
- Predation
- Stage Survival
- Site Fidelity, Immigration, and Emigration
- Overwintering Habitat
- Population Trends through Time
- Genetics
- Detection Probabilities.

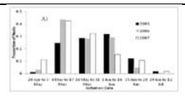
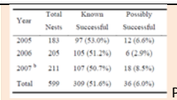
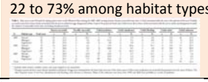
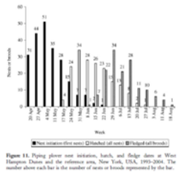
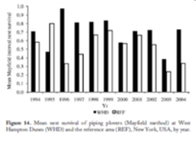
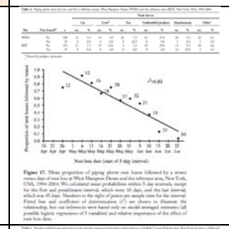
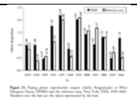
For convenience, the images inserted in the category tables have been placed in an accompanying appendix (Appendix F) so that they may be viewed in full size.

References are listed as “not used” or by the category in which they appeared in the table.

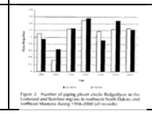
Tables

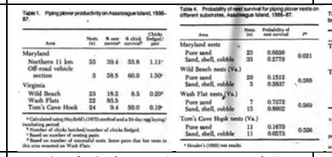
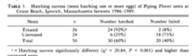
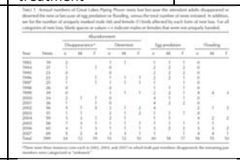
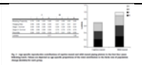
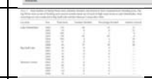
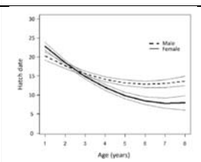
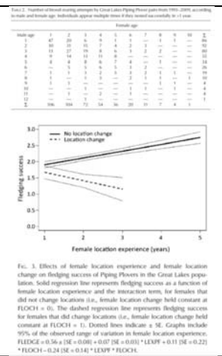
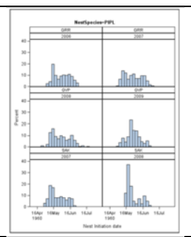
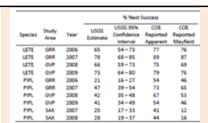
Information within the light orange cells was used in the parameter table.

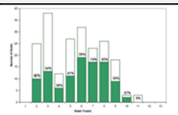
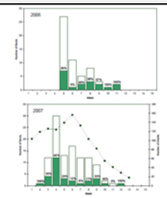
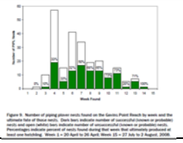
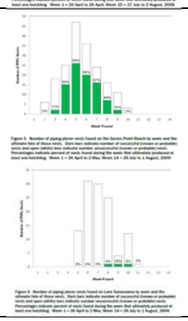
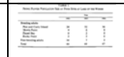
Reproductive Success

Reference	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
Anteau et al. 2012a	Lake Sakakawea	Late April to Late June	Apparent (±SE) was 24 (±6), 28 (±4), 24 (±5), and 3 (±2) % in 2006–2009	42–91 % of nest sites were inundated prior to hatch. Our data indicated that 54 nests failed with evidence of destroyed eggs in the bowl, 23 of these nests (42.6 %) had yolk coagulated in the bottom of the nest bowl.	0.46, 0.30, and 0.04 for 2007, 2008, and 2009	544 2-km segments 52,101,83,110 nests	2006-2009 Apr-Jul	Stratafied-Random Search every 3.4 days Status every 3 days	Low, Medium and High Nest Abundance
Anteau et al. 2014b	Lake Sakakawea		Hatchlings per segment averaged 6.7 (range 1 to 28; CV [SD/mean]=0.85). Nest survival did not vary by landform.	Predation rates of chicks may be higher on islands because gulls nest at and frequent islands at SAK.	Number of chicks fledged per segment averaged 2.1 (range 0 to 9; CV=1.14). We documented that fledging rate for plovers was lower on islands than on the mainland. Our results indicated that fledging rates were lower at areas that were more exposed to wind and waves than at areas that were more protected.	506 mainland segments (2 km length) and 38 island segments. The nest density classification resulted in 403 low-, 88 medium-, and 53 high-use segments.	We searched study segments for nests every 2 to 3 days throughout the nesting season in 2007 and 2008.	Stratafied-Random sample of mainland and island segments. Classified segments based on counts of plover nests: as low (<2 nests), medium (2–9 nests), or high (>9 nests). However, we restricted our analyses to those segments where at least one egg hatched (ntotal=26, n2007=14, n2008=12; nunique=15); these segments were split relatively evenly among island (42%) and mainland (58%) landforms.	
Aron 2005	Upper Missouri River		74.1%; 45.9%; 34.0%; and 55.3%	295 (10%) lost to Predation 66 (2%) abandoned 440 (15%) unknown	1.66; 0.84; 0.88; and 1.23	2914 nests; 10192 eggs	1986-2004	USCOE Monitoring Program	Lake Oahe; Fort Randall River; Lewis and Clark Res.; and Gavins Point River
Barber et al. 2010	Northeast Canada		102/301 = 33.9% nonexclosed 118/183 = 64.5% excosed	The mean number ± SE of fledglings from successful nests, where at least one egg hatched, was not significantly different between exclosed and nonexclosed nests (2.5 ± 0.14 vs 2.8 ± 0.13, respectively; Mann-Whitney, n1 = 114, n2 = 88, P = 0.10). However, when including failed nests, there were significantly more 20-day old chicks, i.e., fledglings, that hatched from exclosed than nonexclosed nests (2.0 ± 0.14 vs 1.4 ± 0.13, respectively; Mann-Whitney, n1 = 141, n2 = 178, P = 0.0009).		N = 484 nests (183 had exclosures)	Data were collected over 22 years (1984-2006 except for 1993; exclosures were used starting in 1988; from 11 sites within Prince Edward Island National Pak.		Enclosures and no enclosures
Catlin 2009	Upper Missouri River		 Page 38 of 106	30% lost to predation +unknown, see Table 2 page 38 of 106		599 nests	2005-2007	Sites were selected for specific characteristics	Natural, created, and managed sand bars
Catlin et al. 2011b	Upper Missouri River		Observed nest success rates ranged from 22 to 73% among habitat types and years. 			599 nests	2005-2007	Sites were selected for specific characteristics	Natural, created, and managed sand bars
Cohen et al. 2009	East coast US					Variable see tables	1993-2004	Sites were selected for specific characteristics	Reference and created
Doherty and Heath 2011	East coast US		Mean plover clutch size was 3.6 0.08 (n = 107) and hatching success was 0.60 0.02.		Overall productivity for the population was 0.99 chicks fledged per pair.	In 2006 and 2007, 81 plover pairs made 107 nesting attempts on SCP study sites. In total, plovers laid 387 eggs, hatched 234 eggs, and fledged 80 plover chicks.	April-July 2006-2007	Sites were selected for specific characteristics	Enclosures and no enclosures

Reference	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
Donlan et al. 2003	East coast US		Table 1. Charadrius melodus. Results of piping plover reproductive success at reference and impact area beaches. Source of data: McGourty (1996), Table 2 and Casey (1996, Appendix E) Year Sites Pairs Nests Eggs Hatched Hatched (%) Fledged (%) Fledged per pair Fledglings **Reference area beaches** 1996 6 20 22 87 75 86 42 56 2.1 1995 4 13 18 63 46 73 24 52 1.9 1994 4 9 10 39 31 80 22 71 2.4 1993 3 9 11 45 32 71 23 72 2.6 1992 2 5 6 23 15 65 12 80 2.4 **Impact area beach** 1996 1 9 16 53 22 42 9 41 1.0 1995 1 9 9 35 31 89 14 45 1.6 1994 1 8 9 33 17 52 8 47 1.0 1993 1 8 10 38 33 87 7 21 0.9 1992 1 6 6 19 7 37 4 57 0.7			1992-1996	Sites were selected for specific characteristics	Oil impact and reference beaches
Espie et al. 1996	Great plains Canada		Table 5. Sample means and standard deviations of habitat variables for successful, flooded, and deprodated nests. Variable: Successful nests; Flooded nests; Deprodated nests; F (ANOVA)[a] 1; 2 Near water: 68.23±40.10 (30); 43.92±26.45 (24); 53.00±20.53 (13); 6.06*; 0.53 Main water: 131.70±93.64 (30); 69.29±51.44 (24); 81.62±69.13 (13); 8.66**; 3.30 Vegetation: 120.83±142.12 (30); 92.87±82.32 (24); 23.83±15.28 (13); 0.06; 5.05* Pebbles: 1025.67±354.38 (21); 1051.92±390.11 (20); 946.89±21.74 (11); 2.53*; 0.25 Stones: 37.05±26.27 (21); 47.03±18.89 (20); 31.83±21.74 (11); 1.94; 0.32 Note: Numbers in parentheses are sample sizes. [a]1, successful vs. flooded nests; 2, successful vs. deprodated nests; *, $p < 0.05$; **, $p < 0.01$; others $p > 0.05$.			In 1992, we found 30 nests on three different beaches and in 1993 we located 39 nests on four beaches. In total, data from 67 nest sites and 116 random sites were collected; however, slide photographs were only available for 52 nest and 111 random sites.			
Espie et al. 1998	Great plains Canada		Table 1 Nest chronology, reproductive success, nest loss and nest elevations for piping plovers at Lake Diefenbaker, Saskatchewan, 1992–1993 1992: Mean; 1 SE; n. 1993: Mean; 1 SE; n; t; p Nest initiation site: 12 May; 2.14; 23; 9 May; 1.05; 9; 0.81; 0.42 Hatch date: 15 June; 2.14; 23; 10 June; 1.05; 9; 1.38; 0.18 Fledge date: 7 July; 1.29; 19; 3 July; 1.99; 2; 1.09; 0.29 Eggs laid/pair: 3.9; 0.05; 30; 4.0; 0.00; 39; −1.64; 0.10 Chicks hatched/pair: 2.8; 0.31; 30; 0.8; 0.26; 39; 4.82; <0.001 Fledglings/pair: 2.0; 0.27; 30; 0.1; 0.08; 39; 7.54; <0.001 % of nests lost to predators: 20.0; 30; 17.9; 39 % of nests lost to flooding: 3.3; 30; 59.0; 39 Nest elevation (m): 554.67; 0.21; 28; 555.10; 0.07; 39			May-July 1992-1993	Five study sites were chosen on beaches of the lake where concentrations of nesting plovers were found during the 1991 International Census and previous census work.	
Felio et al. 2010	Upper Missouri River		Of the 294 plover nests found in the area, 149 (49.7%) are known to have hatched at least one chick and 9 (3%) are likely to have hatched at least one chick, therefore 158 (52.6%) nests are considered successful in 2009.	*Table 3. Piping plover nest success for the 2008 breeding season.* Hatched: Known Hatch 149; Probable Hatch 9; Total 158 Failed: Abandonment 13; Known Predation 41; Flood 33; Weather (hail or rain) 12; Unknown 43; Total 142			2009	This is an annual report, results are preliminary. Sites were selected for specific characteristics	Natural, created, and managed sand bars
Gaines and Ryan 1988	North Dakota	Eggs were not laid until 19 May 1984 and 6 May 1985. Mean clutch size was 3.5 ± 0.1 (SE) eggs in 1984 and 3.7 ± 0.1 eggs in 1985.	We counted 140 piping plovers at Chain-of-Lakes in 1984 and documented breeding by-46 pairs. In 1985 we counted 200 adults and found nesting by 91 pairs. Table 2. Piping plover breeding population and reproductive success at Chain-of-Lakes, North Dakota, 1984–85. 1984; 1985 No. breeding pairs: 46; 91 No. nests: 55; 97 % egg success*: 54; 33 % nest success*: 41; 42 % chick success*: 77; 65 No. fledged: 68; 95 Fledged/breeding pair: 1.48; 1.04 *Calculated using the Mayfield method (Mayfield 1961, Johnson 1979).	Predation accounted for 296 of the 318 egg failures for both years. Twenty eggs were abandoned or infertile and 2 were lost as a result of human disturbance. In 1984, nest success on disturbed (32%) and undisturbed territories (40%) did not differ (P = 0. 7).	In 1984, chicks fledged in 21-28 days. In 1985, weather during the pre-fledging period was colder and may have limited chick growth rates. In 1985, few chicks fledged in <26 days. In both years, 0.48 chicks fledged/adult.	150 nests, 137 breeding pairs	1984-1985	Transects were established every 50 m along all available beaches.	None
Goldin and Regosin 1998	East coast US			The difference in fledging success by habitat can be attributed to differences in chick survivorship with 33 of 41 (80.5%) chicks with access to mudflat habitat surviving to fledging, compared to only 10 of 28 (35.7%) chicks limited to beach habitat	Mean fledging success was 3.0 (SD = 1.6, n =- 5) and 3.0 (SD = 1.3, n = 6) for broods with access to mudflats and 1.3 (SD = 1.0, n =- 4) and 1.7 (SD = 1.5, n = 3) for broods without access to mudflats	Eight and nine pairs of Piping Plovers nested in 1993 and 1994.	April-Aug 1993-1994	Sites were selected for specific characteristics	With and without Mudflat access
Greenwald 2009	East coast US		All first nest attempts hatched by June 9 (n = 30). The apparent hatching success for all nests combined was 69.1% (94 hatched/ 136 eggs laid).	The presence of predator exclosures at most sites likely contributed to this high hatching success rate; the relationship between exclosure presence and hatching success was positive but weak (r = 0.201; p = 0.287).	Of the eggs that hatched, 38.3% fledged (36/94), for a fledging success rate of 26.5% (36/136).	34 nests	April-June 2003	Sites were selected for specific characteristics	None
Ivan and Murphy 2005	North Dakota		Nest success was 0.32 (95% CL: 0.26, 0.39) for unprotected nests and 0.66 (95% CL: 0.52, 0.82) for nests protected by electric fence. Thus, nest success increased by 106% for nests where mammalian predators were excluded. When avian predators were excluded from plover eggs in addition to mammals, mean nest success (x-=0.73, 95% CL: 0.70, 0.77) was 128% higher than that of unprotected nests.	Nest abandonment by plovers on alkali lakes was infrequent (1%; Murphy et al. 2000). Roughly 83% (106/128) of nest predation at prairie alkali lakes in North Dakota and Montana can be attributed to mammals and the remainder (17%) to birds.	Fledging rate (number of 18-day-old chicks produced per breeding pair) and 95% CI.	310 nests unprotected, 65 nests protected by electric fence, and 913 nest with cage	May-June 1994-2002	Plover nests and chicks were unprotected during 1994–1997. Beginning in 1998, predator exclosure "cages" were applied to most plover nests to protect eggs from both mammalian and avian predators.	Enclosures and no enclosures
Johnson et al. 1997	Great Plains and Great Lakes	Egg laying begins the ·second or third week of May, although Schwalbach (1988) reported median initiation dates for South Dakota piping plover nests of	Nest success commonly ranges from 29-51 % (Gaines and Ryan 1988, Dirks 1990, Kruse 1993). Adults may renest up to four times if nests are destroyed (USFWS 1987) Typically, more than 50% of nest losses are attributable to predation (Dirks 1990, Kruse 1993) or flooding (Lingle 1993a, b)					Life history, citing others	

Reference	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
		20 June in response to high Missouri River water levels.							
Knetter et al. 2002	North Dakota					922 nests	1994-2000	Subjective selection to represent extremes in success characteristics	Evaluated landscape and breeding lake scales on two regions
Kruse et al. 2001	Upper Missouri River		TABLE 1. Productivity summary for Piping Plovers and Least Terns nesting on the Gavins Point and Fort Randall river reaches of the Missouri River, South Dakota, 1991 and 1992. Piping Plover / Least Tern Total Nests: 238 / 354 Total Destroyed: 128 (53.8%) / 179 (50.6%) Predation: 61 (25.6%) / 80 (22.6%) Chicks Hatched: 368 / 330 Chicks Fledged: 57 (15.5%) / 81 (24.5%) Apparent nesting success of caged nests (n=86, 62%) was higher ($P<0.001$) than for control nests (n=122, 34%). Of 33 caged nests that were unsuccessful, 6 (18%) were inundated, 16 (48%) were lost to predation, 2 (6%) were abandoned for unknown reasons, and 9 (27%) were lost due to human disturbance, sandbar erosion, weather, or unknown causes. 18 plover nests with unknown fate were excluded from analysis. Apparent nest success was 46.2%, fledge rate was 0.33.			May-August 1991 and 1992	Nest were visited every 5-7 days	None
Larson et al. 2002	North Dakota		Table 1. Baseline input parameters for the stochastic demographic simulation model of the female segment of the Great Plains population of piping plovers. Parameter / Estimate / SD / CV / Source Beginning population size, N_0 / 1,500 / / / Censuses[a] Proportion on alkaline sites, P_a / 0.73 / / / Censuses[a] Alkaline fledging rate, F_a / 0.445[b] / 0.215 / 0.483 / Table 4 River fledging rate, F_r / 0.365[b] / 0.176 / 0.483 / Table 5 Annual adult survival, S_a / 0.737 / 0.205 / 0.279 / Larson et al. (2000) Annual immature survival, S_i / 0.48 / 0.13 / 0.279 / Melvin and Gibbs (1996) [a] Haig and Plissner (1993) and Plissner and Haig (2000a). [b] Female fledglings produced per breeding female per year (i.e., [fledglings per pair]/2). (Note, Plissner and Haig 2000a cited in this table is 2000b in this document.) Table 2. Fledging rates of piping plovers at alkaline wetlands and freshwater lakes in the Great Plains without predator-exclusion management. Location / λ[a] / SE[b] / Years / Source Williams Preserve[c], North Dakota / 0.86 / 0.12 / 13 / M. R. Ryan et al.[d] Mountrail County, North Dakota / 0.45 / 0.12 / 4 / Murphy et al. (2000)[e] North Dakota–Montana state line / 0.89[f] / 0.24 / 4 / Murphy et al. (2000)[e] Sheridan County, Montana / 1.04 / / 5 / Rabenberg et al. (1993)[g] Lake of the Woods, Minnesota / 1.30 / / 4 / Wilens (1986)[h], Haig and Oring (1987) Manitoba (large lakes) / 0.9 / 0.1 / 4 / Haig (1987) Big Gull Lake, Saskatoon / 0.76 / / 2 / Whyte (1985) Weighted mean/sum / 0.89[i] / 0.10 / 36 / [a] Fledglings produced per breeding pair per year. [b] Data were not available to calculate an SE for some means. [c] John E. Williams Memorial Nature Preserve, McLean County, North Dakota, USA. [d] Some data were published by Prindiville Gaines and Ryan (1988) and Mayer and Ryan (1991); the rest were unpublished data, collected in 1989–1990, 1992–1996, and 1999, from M. R. Ryan and The Nature Conservancy staff. [e] Mountrail and Slade Line data were kept separate for this analysis but were combined by Murphy et al. (2000). [f] Years do not overlap with data from Rabenberg et al. (1993). [g] Most of the data were presented earlier by Smith (1991). [h] Some of the data were presented earlier by Wilens and Cuthbert (1984). [i] Calculated using location means weighted by n years. Table 3. Fledging rates of piping plovers at river sites and reservoirs in the Great Plains without predator-exclusion management. Data were not available to calculate a standard error for any location mean. Location / λ[a] / Years / Source Lake Diefenbaker, Saskatoon / 0.49 / 10 / Espie et al. (1998) Gavins Point, South Dakota / 0.45 / 7 / Kruse (1993)[b] Missouri River, North Dakota / 0.86 / 4 / Mayer (1993) Northeastern Montana / 1.61 / 4 / Rabenberg et al. (1993) Platte River, Nebraska / 1.00 / 1 / J. Dinan, Nebraska Game and Parks Commission, personal communication Weighted mean/sum / 0.73[c] / 26 / [a] Fledglings produced per breeding pair per year. [b] Some of the data were presented earlier by Schwalbach (1988) and Dinkl (1990). [c] Calculated using location means weighted by n years; SE = 0.21. Table 4. Fledging rates of piping plovers protected with nest cages and temporary electric fencing on alkaline wetland beaches in 3 areas of northwestern North Dakota and northeastern Montana, USA. Location / λ[a] / SE / n / Years / Breeding pairs Nest cages Williams Preserve[b] / 1.25 / 0.11 / 5 / 1995–1999 / 55 Mountrail County / 1.24 / 0.15 / 6 / 1994–1999 / 114 North Dakota–Montana state line / 1.46 / 0.28 / 4 / 1996–1999 / 138 Weighted mean / 1.28[c] / 0.07 / Nest cages and temporary fencing Williams Preserve[b] / 1.39 / 0.52 / 2 / 1996–1997 / 27 Mountrail County / 1.59 / 0.22 / 6 / 1994–1999 / 105 North Dakota–Montana state line / 2.09 / 0.19 / 3 / 1996–1998 / 21 Weighted mean / 1.78[c] / 0.21 / [a] Fledglings produced per breeding pair per year. [b] John E. Williams Memorial Nature Preserve, McLean County, North Dakota, USA. [c] Calculated using location means weighted by SE^{-2}. Fledging rates without predator exclusion were 0.89 (SE = 0.10) fledglings/pair at alkaline wetlands and 0.73 (SE = 0.21) fledglings/pair at rivers and reservoirs. Predator exclusion increased local fledging rates to 1.15-2.25 fledglings/pair.			2 to 13 years spanning 1989 to 1999	Modeled population growth and fledgling production based on extensive sets of unpublished data for piping plovers breeding at alkaline wetlands in 3 areas near the center of their range in the Great Plains.	We calculated separate mean fledging rates for piping plovers under the following 5 levels of predator-exclusion management: no protection (hereafter, baseline), nest cages, permanent electric fencing, cages plus permanent fencing, and cages plus temporary electric fencing.
Maslo and Lockwood 2009	East coast US		Table 2 Effects of conservation treatments on hatching success (1998–2007). Table 3 Percentage of Abandoned Nests Under Varying Conservation Treatments.				We conducted a nest survival analysis for 10 years (1998–2007) of piping plover monitoring data to determine: 1) the effectiveness of predator exclosures and electrified predator exclosures, and 2) conditions associated with nest abandonments at electrified exclosures. We obtained piping plover nest monitoring data from the New Jersey Division of Fish and Wildlife Endangered and Nongame Species Program. For failed nests, ENSP staff listed a presumed cause of failure (flooded, predated, abandoned).		
Maxson and Haws 2000	Minnesota		Table 1. Fate of Piping Plover nests at Lake of the Woods, Minnesota, 1986-2000. Table 2. Piping Plover hatching and fledging success at Lake of the Woods, Minnesota, 1993-2000.			88 nests	1988-2000	All observed nests	None
McGowan et al. 2007	Missouri River		Table 1. Habitat characteristics within one square meter of four Piping Plover (*Charadrius melodus*) nests found on an island in the Missouri River between South Dakota and Nebraska, spring 2006. Nest / No. Cottonwood saplings / No. Willow saplings / Vegetation height (m) / % Vegetative ground cover / Distance to un-vegetated habitat (m) / Nest fate 1 / 5 / 1 / 0.3-0.9 / 1-5 / 32.6 / Failed Unknown 2 / 5 / 1 / 0.2-1.0 / 1-5 / 19.1 / Hatched 3 / 9 / 2 / 0.2-1.5 / 1-5 / 68.6 / Depredated Raccoon 4 / 1 / 3 / 0.3-1.8 / 1-5 / 13.3 / Abandoned				2006	All observed nests	None

Reference	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
Murphy et al. 2003a	North Dakota				In 20 replicated trials, fledgling production rates were: no protection, 0.72 chicks/pair (95% CI: 0.29- 1.15, N = 43 pairs); cage only, 1.73 (1.30-2.16, N = 46); fence plus cage, 2.06 (1.63-2.49, N = 50). A replicate consisted of three matched beaches that occurred within one km of each other (not necessarily on the same lake); we assumed all three were subject to similar predator and environmental influences. To maintain independence among treatments, matched beaches within a replicate were selected so that they were separated from each other by vegetation or open water.		1996-1997	Selected sites, treatment allocation of matched sites was determined randomly.	We compared success of nesting plover pairs provided with: (1) no protection, (2) cages that protected eggs in individual nests from both mammalian and avian predators, and (3) a combination of cages plus a temporary electric fence that excluded mammalian predators from the entire nesting beach where chicks were being reared.
Patterson et al. 1991	Atlantic coast					We found 125 nests during 1986 and 1987	1986-1987	Selected sites	Substrate and human access
Rimmer and Deblinger 1990	East coast US		92% (24/26) of treated nests successfully hatched one or more eggs. During the same period, only 25% (6/24) of untreated nests hatched. 	Most losses (94%, 17/18) were the result of predation.		26 nests with exclosures and 24 nest without exclosures, nonrandom application of treatment	1986 (no treatment), 1987 only second nest attempts were treated, and in 1988-1989 treated all known nests.	Selected site	With and without exclosures, note, years are confounded with treatment.
Roche et al. 2010a	Great Lakes			Approximately 16% of monitored clutches laid by Great Lakes Piping Plovers from 1993 to 2007 were lost before hatching. Our results reveal that most early-season nest abandonment in Piping Plovers was attributable to the death of attendant adults.				Banding data from 1993 to 2008 and 13 nesting sites in 2008	All nests are protected by predator exclosures
Roche et al. 2008	Great Lakes							Banding data from 1993 to 2008	Captive-reared and wild reared chicks
Roche et al. 2012	Great plains Canada						Banding data from 2002-2009		3 breeding areas, flood and non-flood years
Saunders et al. 2012	Great Lakes				The mean number of young fledged per brood was 2.0 ± 1.42 offspring (variance = 2.02). Female location familiarity was the most important predictor of fledging success, even though site fidelity in Piping Plovers is male-biased (Haig and Oring 1988b).	n = 320 unique pairs comprised of 181 females and 183 males	1993-2009	Known nest locations	100% of nests were exclosed from 2004– 2009
Shaffer et al. 2013	Upper Missouri River			Tables 9 – 21 (pages 130 -134)	Minimum	Gavins Point, Garrison Reach, and Lake Sakakawea	2006-2008	Stratified Random with Habitat searched every 2-3 days from mid-April- early Aug	USCOE % of population over time

Reference	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
Sherfy et al. 2008	Missouri River			Table 24. Numbers and fates of least tern and piping plover nests from intensive survey, mile river segments on the Garrison Reach during 2007. Nest Fate Nests / Successful / Probable successful / Probable Failure / Failure / Unknown Least tern 112 40 28 0 23 16 Piping plover 207 44 54 0 96 13 Total 319 89 82 0 109 29		Garrison Reach and Lake Sakakawea	2007	Stratified sample	High or low use based on USACE data
Sherfy et al. 2009a	Missouri River		Table 12. Numbers and fates of least tern and piping plover nests on created and natural sandbars on the Gavins Point Reach (GVP) and Lewis and Clark Lake (LCL) in 2008. Nest Fate Species / Study Area / Sandbar Type / Total Nests / Successful / Probable Successful / Probable Failure / Failure / Unknown Least Tern / GVP / Mechanically Created / 155 / 80 / 25 / 0 / 38 / 12 Natural / 41 / 15 / 5 / 0 / 19 / 2 LCL / Mechanically Created / 140 / 58 / 38 / 0 / 31 / 13 Piping Plover / GVP / Both / 237 / 94 / 13 / 0 / 108 / 22 Total / 537 / 247 / 81 / 0 / 196 / 49		Gavins Point and Lewis and Clark Lake	2008	Stratified sample	High or low use based on USACE data
Sherfy et al. 2009b	Missouri River		Table 7. Numbers and fates of least tern and piping plover nests on the Gavins Point Reach in 2009. Nest Fate Total Nests / Successful / Probable Successful / Probable Failure / Failure / Unknown Least Tern 136 91 6 1 33 5 Piping Plover 194 72 16 0 98 8 Total 330 163 22 1 131 13		Gavins Point and Lake Sakakawea	2009	Stratified sample	High or low use based on USACE data
Tremblay et al. 1994	Northeast Canada		Kouchibouguac National Park, New Brunswick. Year / Territorial pairs / Number of nests / Hatching rate[a] / Fledging rate[a] 1982 16 16 2.7[b] – 1983 21 21 2.8[c] 2.1[d] 1984 11 4 – – 1985 12 3 – – 1986 15 12 – – 1987 15 15 1.2 0.93 1988 13 9 2.6 2.2 1989 16 16 2.6 2.1 1990 15 15 3.4 1.9 1991 17 16 1.7 1.2 1992 15 13 3.3 2.3 1988–1992 15.2 13.8 2.5 1.94 Hatching rate for enclosed nests 3.1 Mean hatching rate for all nests (1988–1992) 2.5 [a] Hatching success and fledging rates have been estimated only since 1988, the start of the official monitoring program; [b] n = 11; [c] n = 20; [d] n = 12.			1982-1992	All known nest sites	Assess ness exclosures
Wemmer et al. 2001	Great Lakes		The model was more sensitive to changes in reproductive success than to dispersal. Population persistence was more strongly affected by variation in site specific reproduction probabilities than by changes in number of available breeding territories or sites. When site repro-				14 years (1984-1997) of monitoring data and 6 years (1993-1998) of banding data	Habitat based population model	None
Wiens and Cuthbert 1988	Minnesota		TABLE 3 RELATIONSHIP BETWEEN MATE RETENTION AND PREVIOUS REPRODUCTIVE SUCCESS IN LAKE OF THE WOODS PIPING PLOVERS Reproductive success in original season / Number of pairs / Fate of pairs in subsequent season: Reunite / Separate / Discontinue / Status unknown Succeeded 27 5 3 12 7 Failed 9 0 3 4 2 Reproductive success is defined here as producing at least one chick to at least 10 days old.			1982-1984	All known nest sites	Distance between successive year nest sites and mate changes
McGowan et al. 2014	Great Plains				Shaffer et al. (2013) reported that birds produced 0.32 (SE = 0.27) female fledglings per breeding female in the northern rivers region during a three year study.			Models the probability of extinction	
Dirks 1990	Missouri River	Nest initiation dates for piping plovers ranged from May 1 to July 12 with 90% of the nests initiated between May 8	Piping plover hatching success increased from 29% in 1988 to 37% in 1989.		Of 62 least terns and 66 piping plovers that fledged below Gavins Point Dam in 1988, over 20% did so the last week of August. In 1989 no major flooding events		Surveys were conducted in 1988 and 1989 to determine the distribution and reproductive success of least terns and piping plovers on the Missouri and Cheyenne rivers in South Dakota. All suitable habitat was surveyed, with research concentrated on two important areas: 1) the		

Reference	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
		and June 30 in 1988 and from May 3 to July 6 with 90% of the nests initiated between May 12 and June 26 in 1989. Piping plovers averaged 1.4 nest initiations per pair in 1988 and 1.7 initiations per pair in 1989.	Table 4. Piping plover nest fates along the Missouri River in South Dakota during 1988 and 1989. Gavins Pt. 1988: 139, 103, 42, 36, 97, 67, 25, 27, 4, 1, 4, 1, 5, 1, 4, 1, 55, 36, 19, 9, 158, 112 Gavins Pt. 1989: 7, 9, 1, 5, 6, 4, 0, 2, 3, 1, 0, 0, 0, 0, 0, 0, 3, 1, 3, 0, 10, 9 Ft. Randall 1988: 146 (100), 43 (26), 103 (61), 25 (24), 7 (7), 4 (4), 5 (5), 4 (4), 58 (36), 22 (13), 168 (100) Ft. Randall 1989: 112 (100), 41 (34), 71 (59), 29 (41), 2 (3), 1 (1), 1 (1), 1 (1), 37 (52), 9 (7), 121 (100) Nests with Known Fate; No. Hatched; No. Destroyed or Abandoned; Cause of Destruction/Abandonment: Predator; Flooding; Human Disturbance; Weather; Sandbar Erosion; Unknown Agent; Fate Unknown; Total No. Nests Found		occurred below Gavins Point Dam and 100% of 13 piping plover chicks that fledged, did so by August 15. Fewer piping plovers fledged in 1989 (18) than in 1988 (65) or 1987 {101) but more than in 1986 (5) {Schwalbach 1988). Although piping plover hatching success increased in 1989, the number of chicks fledged per nesting pair decreased from 0.55 in 1988 to 0.26 in 1989.		Missouri River from Fort Randall Dam to Springfield, SD, and 2) the Missouri River from Gavins Point Dam to Ponca State Park, NE. All colony sites were revisited every 7 to 10 days to determine nest fates and to search for new nests. Colony sites with successful nests were monitored to determine chick survival and fledging success.		

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Invertebrate Prey

Reference	Location	Findings	Sample Size	Years	Design
Anteau and Sherfy 2010	Upper Missouri River	Sticky traps deployed continuously for 12 h or more had lower catch rates than four consecutive-composited 3-hour deployments, suggesting that trap effectiveness declined for >3-hour deployments. Thus, if sticky traps are used to index plover forage abundance without controlling for time of day and wind speed, data may be highly variable or estimates could be biased. Findings:	2 replicates in Aug 2006 and 3 replicates in Aug 2008	Summers of 2006 and 2008	We conducted an experiment to examine diurnal variation in catch rates of sticky traps in relation to habitat type on a reservoir, Lake Sakakawea, and emergent sandbar habitat on the Garrison Reach of the Missouri River. We also examined if sticky traps have consistent catch rates among 3-, 12-, and 24-hour periods to determine if sticky traps could be set for an entire day to account for diurnal variation in catch rates. Within each habitat type [dry sand/substrate (<50% vegetation cover); wet sand/substrate (<50% vegetation cover); and vegetated dry sand/substrate (>50% vegetation cover; hereafter vegetated)] we randomly selected four locations (>25 m apart) for sampling.

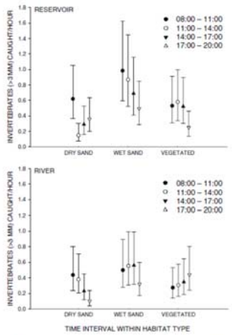


Fig. 1 Least-squares geometric mean (± 95% CL) counts per hour of large invertebrates (>3 mm) captured on sticky traps during four time intervals within three habitat types at reservoir (Lake Sakakawea) and riverine (Garrison Reach) study areas of the Missouri River in summers 2006 and 2008. Significant differences among estimates generally are indicated by a 95% CL not overlapping the point estimate of another level

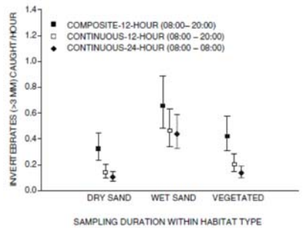
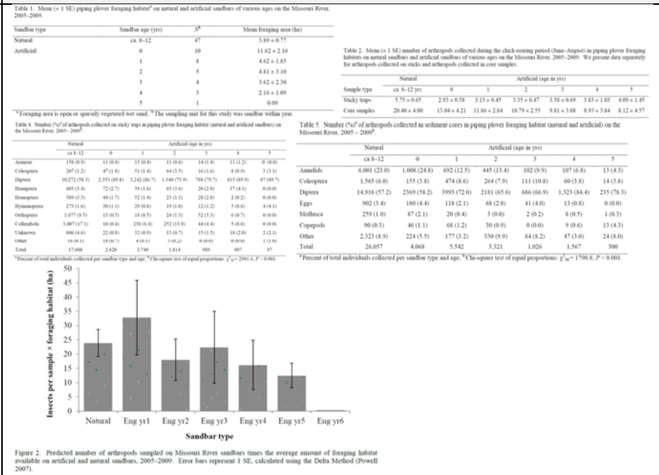
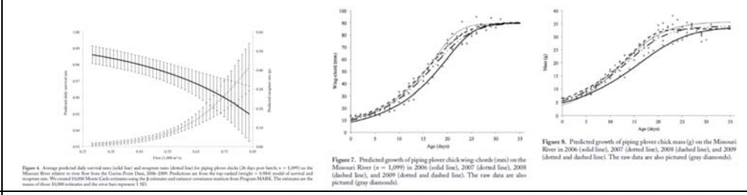
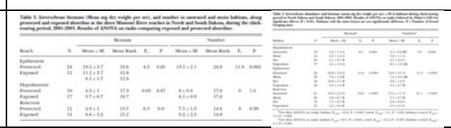
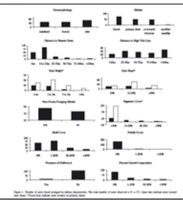


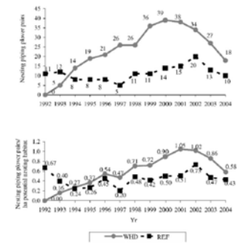
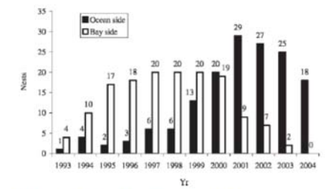
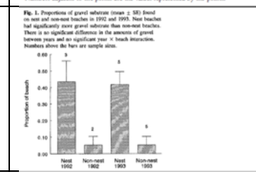
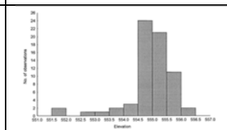
Fig. 2 Least-squares geometric mean (± 95% CL) counts per hour of large invertebrates (>3 mm) captured on sticky traps deployed for three durations by three habitat types for reservoir (Lake Sakakawea) and riverine (Garrison Reach) study areas of the Missouri River combined in summers 2006 and 2008. Significant differences among estimates generally are indicated by a 95% CL not overlapping the point estimate of another level

Reference	Location	Findings	Sample Size	Years	Design
Catlin et al. 2012	Upper Missouri River		Two parallel transects with 2 to 4 samples each using 30 minute time intervals.	2005-2009	We compared arthropod abundance indices from artificial and natural sandbars as part of an evaluation of foraging habitat. We sampled in two wet substrate cover types: damp sand and mud, and saturated sand and mud. We sampled every two weeks during the June–August chick-rearing period.
Catlin et al. 2013	Upper Missouri River	Predicted fledging (recapture probability < 0.05) occurred between 25 and 35 days from hatch, similar to the age of the oldest actual yearly captures within the study (26–33 days from hatch). Recapture rate varied by year and was positively correlated with flow (Fig. 4) Survival varied by year and was negatively correlated with flow. During the period in which most birds fledged (25–35 days), wing-chord lengths in 2006 and 2009 (high flow) were smaller than in 2008 and 2007 (low flow), but this difference was largely absent by 35 days of age (Fig. 7). Masses during this period were lowest for 2006 (high flow) and highest for 2007 (low flow); 2008 and 2009 were intermediate. 	We individually marked each chick (n = 1,192 from 1099 nests) with color bands and a color flag. We measured mass (±0.1 g) and wing-chord (±1 mm) every third day until fledged.	2006-2009 The study was characterized by 2 relatively high flow years (2006 and 2009) and 2 relatively low flow years (2007 and 2008).	We studied the relationship between river flow and the timing of fledging, growth, and survival of piping plover chicks. We developed several a priori structures for survival (ϕ) and recapture rate (*p*) meant to test hypotheses about fledging and the effects of flow on fledging. For an individual bird, we assumed that a recapture rate (*p*) < 0.05 would be equivalent to fledging. We analyzed the change in wing-chord length and mass as a function of age using nonlinear regression. We used a Cormack–Jolly–Seber (CJS) model in Program MARK package RMARK to calculate both survival (ϕ) and recapture rate (*p*). Assumed that increased flows equates to decreased prey availability.
Elias et al. 2000	East coast US	Arthropod abundance indices were correlated with foraging rates (Spearman's Rho = 0.81. P = 0.049, n = 9).	We randomly selected a sample of beach segments where broods foraged and within each segment we randomly selected 2 transects perpendicular to the long axis of the beach. We sampled once a week for 7 consecutive weeks each year starting in June. We sampled each segment for 3 hr on the same day to reduce temporal variability using sticky traps.	1992-1993	The goal of this study was to determine whether brood-rearing quality of beaches with ephemeral pools or bay tidal flats was superior to the quality of beaches without these habitats on the central New York barrier islands.
Le Fer et al. 2008b	Missouri River	Invertebrate numbers were higher in saturated and moist habitats than in vegetated and dry habitats (P < 0.05). At the epilimnetic (constant flow release) and hypolimnetic (pulsed flow release) reaches, foraging Piping Plover chicks used saturated and moist habitats more than vegetated and dry habitats, based on availability. On the Missouri River, protected shorelines were important foraging sites for Piping Plovers during the breeding season, and sandbars with low-lying moist habitat were important to foraging chicks.		2001-2003	We examined the relationship between site selection and invertebrate abundance indices within habitats.
Nordstrom and Ryan 1996	Great Lakes	The mean invertebrate biomass at two Great Lakes beaches (Platte Bay, MI = 0.16 g/m2/hr; Long Island, WI = 0.11 g/m2/hr) was similar (P > 0.05) to that at North Dakota alkali beaches (0.09 g/m2/lhr) occupied by breeding piping plovers. However, at four other Great Lakes beaches, we measured 3 to 6 times less (0.01--O.02 g/m2/hr; P < 0.01) invertebrate biomass than at the alkali wetland beaches.	We measured invertebrate biomass and abundance at six beaches at three National Lakeshores in the Great Lakes in 1988. Piping plovers did not occupy any of these sites in 1988. We sampled invertebrates at beach habitat adjacent to alkali wetlands at the John E. Williams Nature Preserve (JWMNP), McLean County, North Dakota. This site supported 90 breeding pairs of piping plovers in 1988 (X = 94 from 1984-1995).	1988	We assessed the potential for reestablishment of endangered piping plovers at selected Great Lakes beaches by comparing invertebrate biomass and abundance with that at alkali wetland beaches occupied by breeding plovers in the Great Plains. We measured invertebrate occurrence using sticky traps secured to beach foraging substrates.

Nest Site Selection

Reference	Location	Elevation	Vegetation Cover	Slope	Finding
Anteau et al. 2012b	Lake Sakakawea				SAK and other reservoirs were used heavily by plovers including during drought years (e.g., 2006, 2007), years when their natural wetland habitats should be readily available in adjacent areas.
Anteau et al. 2014b	Lake Sakakawea				We also examined how landform, wind fetch, and chick density influenced chick growth rates at our 2-km segment scale. We found 103 and 83 nests and banded 94 and 81 chicks, representing 30 and 36 unique broods, during 2007 and 2008, respectively. Amount of habitat at surveyed segments with chicks that hatched averaged 5.7 ha and ranged from 0.2 to 19.17 ha (CV=0.87). Piping Plovers selected islands for nesting over the mainland.
Burger 1994	East coast US				I studied habitat use (using transects) and foraging behavior (using focal animals) at three habitats on each of three nesting beaches over a 2-yr period (1988-1989) in New Jersey, USA.

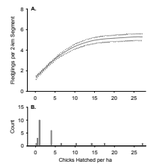
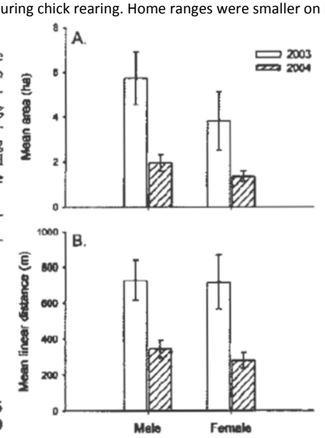
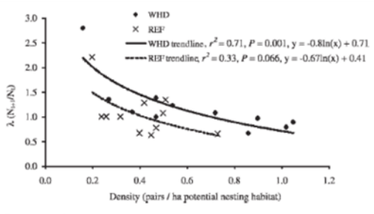
Reference	Location	Elevation	Vegetation Cover	Slope	Finding
					TABLE 2. Comparison of foraging piping plovers on the three study sites (1988 and 1989) in New Jersey. Each sample was 2-min observation of a Foraging Plover. Given are x ± SD. Beginning: Number of samples 801; Seconds spent: Feeding 73.2 ± 28.3; Alert 24.4 ± 20.6; Conspecific aggression 0.2 ± 2.2; People within 100 m 1.9 ± 2.4; Number of people: Fishing 0.05 ± 0.6; Sunbathing 0.26 ± 0.9; Walking 2.12 ± 1.1; Number of feeding birds 2.3 ± 2.8 Holgate: 821; 106.1 ± 21.7; 7.3 ± 40.7; 0.4 ± 2.4; 0.1 ± 0.5; 0; 0.33 ± 2.1; 0; 0.9 ± 2.5 Corson's Inlet: 579; 85.0 ± 37.2; 13.4 ± 18.2; 1.3 ± 6.9; 2.8 ± 2.1; 0.04 ± 0.3; 2.26 ± 2.8; 3.4 ± 8.5 Wilcoxon χ^2 (p): 168.9 (0.0001); 253.8 (0.0001); 38.0 (0.0001); 520.4 (0.0001); 151.5 (0.0001); 151.3 (0.0001); 49.8 (0.0001); 589.2 (0.0001)
Cohen et al. 2008	East coast US		We studied Piping Plover nest site selection on renourished beaches in New York in 2004 at sites where future renourishment was planned. Our objective was to learn if Piping Plover nest sites differed from nearby random sites in vegetative cover, coarse materials in the substrate, and number of large objects such as stones and woody debris.		TABLE 1. Characteristics at Piping Plover nests and random sites, Westhampton Island, New York, 2004 (n = 32, 1-m^2 nest and paired random plots within 50 m). Vegetation Cover (%): Nest $\bar{x}$ 7.6, SE 1.7, Max 45.7, Plots in[a] >5% cover 0.41; Random $\bar{x}$ 4.4, SE 1.9, Max 48.6, Plots in[a] >5% cover 0.19 Coarse grains Cover (%): Nest $\bar{x}$ 9.1, SE 2.6, Max 55.6, Plots in[a] >5% cover 0.41, Plots near[b] ≥ 1 large object 0.28; Random $\bar{x}$ 5.5, SE 2.2, Max 47.2, Plots in[a] >5% cover 0.22, Plots near[b] ≥ 1 large object 0.16 [a] Mean of the index used in our nest site selection model (0 = < 5% cover, 1 = 5–~100% cover). [b] Mean of the large objects (rock, shell, wood >64 mm in at least one dimension) index used in our nest site selection model (0 = no large objects, 1 = ≥ 1 large object). Where large objects were present, counts ranged from 1 to 5.
Flemming et al. 1992	Northeast Canada				Table 3. Variety of nest types used by piping plovers in New Brunswick and Nova Scotia, 1983, 1987, 1988, 1989. Pebble: Eastern New Brunswick 12, Northern Nova Scotia 9, Southern Nova Scotia 14 Grass: 4, 1, 34 Mixed: 3, 1, 16 Sand: 0, 0, 9 Although nest site characteristics differed among study areas, individual plovers used a variety of nest types in each area.
Marcus et al. 2007	Platte River				Table 3. Substrate composition and vegetation cover at Least Tern (LT) or Piping Plover (PP) nests (representing what the birds selected) and at random points within used plots (representing what was available to them in the area around their nest) at gravel mines along the Platte and Elkhorn rivers, Nebraska, 2000 and 2001. A used plot is defined as a plot with at least one nest initiated in it. Values with the same letter within a row are not significantly different (P > 0.05). Fine Sand LT: Attractant plots Nests 16.8 A, Random 13.4 AB; Control plots Nests 16.1 A, Random 8.5 B Fine Sand PP: 18.7 A, 17.2 A; 19.8 A, 12.5 B Coarse Sand LT: 56.5 A, 53.5 A; 64.9 A, 85.4 B Coarse Sand PP: 43.2 A, 54.7 A; 43.3 A, 79.2 B Small Gravel LT: 18.9 A, 23.3 A; 9.0 AB, 3.0 B Small Gravel PP: 28.4 A, 20.4 A; 29.6 A, 17.6 A Large Gravel LT: 7.9 A, 10.0 A; 9.5 A, 3.0 B Large Gravel PP: 10.0 A, 6.2 A; 11.6 A, 1.6 A Vegetation LT: 2.2 A, 3.1 A; 1.5 A, 2.0 A Vegetation PP: 4.6 A, 1.7 A; 3.2 A, 3.0 A The possibility of shifting nesting from active to inactive mining areas by using a deterrent (mylar flagging), an attractant (gravel and driftwood spread on bare sand), and a control (untreated sand) was evaluated. Experimental plots (mean 0.36 ha) were established at 18 different gravel mines, twelve in 2000 and seven (one repeat) in 2001 along the Platte and Elkhorn rivers prior to nesting season. Of 117 tern nests, 73% were in attractant, 2% in deterrent, and 26% in control plots. Of 23 plover nests, 61% were in attractant, 9% in deterrent, and 30% in control.
Maslo et al. 2011	East coast US		We translate these results into the following pragmatic target design parameters: (1) vegetative cover: less than 10% (backshore), 13% (primary dune); (2) shell/pebble cover: 17–18%; (3) dune height: ≤1.1 m; and (4) dune slope: ≤13%.		We collected data on piping plover nests at 19 breeding beaches in New Jersey, United States, from 2006 to 2008. Sites consisted of three main geomorphic types—mainland, barrier, and inlet beaches. 
Sidle and Kirsch 1993	Platte River				Table 2. Numbers of adult Least Terns (LT) and Piping Plovers (PP), percentages of all individual birds using sand pits, and the number of sandbar and sand pit nesting sites on the central and lower reaches of the Platte River, 1988-1991. Data were derived from Lingle (1993a), and Nebraska Game and Parks Commission (1988, 1989, and unpublished data), and Sidle *et al.* 1991. Central Platte, Year 1988: Number of birds LT 124, PP 96; % birds on pits LT 61, PP 62; No. nesting sites Bar 6, Pit 16 1989: 194, 94; 94, 64; 6, 16 1990: 176, 74; 91, 78; 3, 17 1991: 158, 58; 87, 76; 3, 14 Mean: 163.0, 80.5; 83.2, 70.0; 4, 15.8 SD: 29.9, 18.0; 15.1, 8.2; 1.7, 1.3 Lower Platte, Year 1988: Number of birds LT 497, PP 161; % birds on pits LT 26, PP 19; No. nesting sites Bar 24, Pit 11 1989: 409, 188; 14, 13; 38, 19 1990: 361, 149; 40, 35; 23, 13 1991: 295, 73; 30, 37; 35, 11 Mean: 390.5, 142.8; 27.5, 26.0; 30.0, 11.2 SD: 85.0, 42.3; 10.8, 11.8; 7.6, 1.3 We found 225 sand pits of which 78 were suitable and 187 were unsuitable for nesting. Along the central Platte, where sandbar habitat is severely degraded, birds nested at 81% of the suitable sand pits (N = 32) at least once during 1988-1991, and most birds (61-94%) nested on sand pits. Along the lower Platte, where both sandbar and sand pit habitat are plentiful, birds nested at 60% of the suitable sand pits (N = 35) at least once during 1988-1991, and most birds (60-86%) nested on sandbars.

Reference	Location	Elevation	Vegetation Cover	Slope	Finding
Catlin 2009	Upper Missouri River		Juveniles from densely-populated, engineered sandbars were more likely to breed on natural habitat the following year than those on less densely-populated sandbars, and, in general, movements of both juveniles and adults suggested a preference for engineered habitat by adults.		Table 1: Use vs. availability of sandbar habitat for piping plover nesting on the Missouri River during the 2005 – 2007 field seasons with respect to habitat modifications. Use was defined as the number of nests that were initiated on each type of habitat within each year. Subscripts on the proportion of available habitat represent selection (+ selection for, - selection against, and = no selection). Natural Sandbars: Year[a] n^b Pu_i (95% CI)[c] Pa_i[d]; Modified Natural Sandbars: n Pu_i (95% CI) Pa_i; Engineered Sandbars: n Pu_i (95% CI) Pa_i 2005: 87, 0.431 (0.348, 0.514), 0.510 =; 42, 0.208 (0.140, 0.276), 0.193 =; 73, 0.361 (0.280, 0.442), 0.297 = 2006*: 48, 0.232 (0.162, 0.302), 0.500 -; 31, 0.150 (0.091, 0.209), 0.275 -; 128, 0.618 (0.537, 0.699), 0.224 + 2007*: 54, 0.252 (0.181, 0.323), 0.384 -; 18, 0.084 (0.039, 0.129), 0.161 -; 142, 0.664 (0.587, 0.741), 0.455 + Page 37/106
Catlin et al. 2011b	Upper Missouri River		As with the engineered habitat, vegetation was probably one of the driving factors behind our observed selection.		Piping plovers used all habitat types in proportion to their availability in 2005 but used engineered sandbars more and managed sandbars less than expected in 2006 and 2007. We found no evidence that managed sandbars affected nest survival, but selection against them brings their usefulness into question.
Cohen et al. 2009	East coast US				 Figure 9. Piping plover nesting-pair numbers and density at West Hampton Dunes (WHD) and the reference area (REF), New York, USA, 1992–2004. Numbers adjacent to the points are the values represented by the points.  Figure 12. Piping plover first nest attempts at West Hampton Dunes, New York, USA, 1993–2004. Numbers over the bars are the number of nests represented by the bar.
Espie et al. 1996	Great plains Canada				 Fig. 1. Proportions of gravel substrate (mean ± SE) found on nest and non-nest beaches in 1992 and 1993. Nest beaches had significantly more gravel substrate than non-nest beaches. There is no significant difference in the amounts of gravel between years and no significant year × beach interaction. Numbers above the bars are sample sizes.
Espie et al. 1998	Great plains Canada	 Fig. 1. Frequency histogram of piping plover nest elevations (m above sea level) at Lake Diefenbaker for 1992 and 1993 combined (n = 67).			
Felio et al. 2010	Upper Missouri River				Table 2. Nest distribution across sandbar types 2005 2006 2007 2008 2009 Engineered (Year 3): n 87 na 16 64 39 Engineered (Year 4): n 16 119 na 43 76 Engineered (Year 5): n na 9 115 na 37 Engineered (Year 6): n na na 11 95 na Engineered (Year 7): n na na na 10 32 Engineered (Year 8): n na na na na 4 Engineered Total: n 73 128 142 111 148; % .38% 62% 66.3% 72% 79.3% Natural: n 87 48 64 42 41; % 43.5% 23% 23% 22% 20.5% Natural/Modified: n 42 31 18 20 1; % 20.5% 15% 8.3% 7% 0.5% Total: 202 207 214 204 188 This is an annual report, results are preliminary.
Gaines and Ryan 1988	North Dakota				Table 1. Mean beach width and substrate characteristics of piping plover nesting territories and unoccupied areas at Chase, Lake, North Dakota, 1984–85. Habitat variables: Year; Territories: $\bar{x}$, SE, n; Unoccupied area: $\bar{x}$, SE, n; P Beach width (m): 1984 20.9 1.1 218 11.6 1.1 91 <0.0001; 1985 30.1 1.1 300 15.6 1.3 206 <0.0001 Vegetation: % cover 1984 4.0 0.2 218 4.2 0.4 90 0.5; 1985 3.1 0.2 300 3.3 0.5 106 0.2 Spatial heterogeneity* 1984 120.5 5.2 180 93.7 6.8 41 0.007; 1985 150.5 2.7 282 116.4 6.7 93 0.004 Gravel: Cover (no./0.1 m^2) 1984 48.1 2.0 220 30.1 0.2 92 <0.0001; 1985 73.4 2.4 300 54.4 0.8 101 <0.0001 Spatial heterogeneity 1984 75.4 5.8 204 34.5 2.5 37 0.004; 1985 70.9 2.2 287 86.4 6.5 90 0.007 *Dis. Intercepts *CV% of distance (Ruth 1976)
Greenwald 2009	East coast US				Nests located on more heterogeneous substrate and on coarse-grained sand contained more shell fragments.
Johnson et al. 1997	Great Plains and Great Lakes	Dirks (1990) found that 25% of Missouri River nests were initiated at elevations of 6 in or less; 55% at 9 in or less; and 75% at 11 in or less above water.	Dirks (1990) found that 25% of Missouri River nests were initiated at elevations of 6 in or less; 55% at 9 in or less; and 75% at 11 in or less above water.		Haig and Plissner (1993) repotted that 60% of northern Great Plains piping plovers used shorelines around small, alkaline lakes, 18% used large reservoir beaches, 20% used river islands and sandbars, and 2% used beaches on large lakes.
Le Fer et al. 2008b	Upper Missouri River	If allowing natural processes to create sandbars is not feasible, artificial sandbars should incorporate inlets and pools as suitable habitat and should slope gently into the water. Water levels should be managed to maintain availability of these moist and saturated shoreline habitats throughout the Piping Plover breeding season.			Sandbars on the epilimnetic (constant flow) and hypolimnetic (pulsed flow) river reaches and the shoreline of the reservoir were surveyed during the pre-chick period (plovers selecting territories, laying, and incubating eggs, late April to late May) and the chick-rearing period (early July to late July). Four habitat types (saturated, moist, dry, vegetation) were compared. There was no difference in the ratio of adult plover use of protected shoreline to availability for reach ($F_{1,21}$ = 0.9, P = 0.36), year ($F_{2,21}$ = 0.9, P = 0.42) or breeding period ($F_{1,21}$ = 1.3, P = 0.27). Most foraging adult plovers (92% ± 3%) were observed on protected shoreline but such shoreline accounted for only 58% ± 5% of the available shoreline (paired t-test: t_{21} = -5.4, P < 0.001).
McGowan et al. 2007	Upper Missouri River				We report on the atypical habitat characteristics of four Piping Plover nests that we found amongst young cottonwood saplings on a sandbar island in the Missouri River along the South Dakota-Nebraska border. Atypical habitat use could have implications for estimating demographic parameters

Reference	Location	Elevation	Vegetation Cover	Slope	Finding
					and management of this protected species. It is important to explore the extent to which Piping Plovers use atypical breeding habitat throughout their range and the quality of this potentially poor habitat.

Dirks, B.1. 1990. Distribution and productivity of least terns and piping plovers along the Missouri and Cheyenne rivers in South Dakota. M.S. Thesis, South Dakota State University, Brookings, SO.

Nest Density

Reference	Location	Findings
Anteau et al. 2014b	Lake Sakakawea	 The majority (81%) of our segments had hatched chick densities less than 5 hatchlings per ha. Density of chicks hatched per segment averaged 3.8 chicks/ha and ranged from 0.2 to 27.1 (CV=1.70). Fledging-rate estimates from the density-dependent model increased with density of hatchlings, but the slope stabilized between approximately 5-15 hatchlings per ha (Fig. 2). When we reran our analyses on segments with <5 hatchlings per ha (n=21), there was considerable model uncertainty between the full density dependent and the density independent model. We found little support for density dependent processes influencing fledging rates of chicks when densities were <5 hatchlings per ha, which comprised >80% of areas we studied. However, there was support for density dependence influencing fledging rates of chicks at densities >5 hatchlings per ha. Our data suggest that fledgling rates no longer increase with increasing hatchling density greater than about 10 to 15 hatchlings per ha. There was no support for the notion that chick density influenced chick growth rates, suggesting that food resources may not have been limiting even in dense chick rearing areas.
Haffner et al. 2009	Great Lakes	The mean size of home ranges of Piping Plovers that fledged at least one chick was 2.9 ± 0.5 (SE) ha (range = 0.4-11.2 ha), and the mean linear beach distance traversed was 475 ±53 m (range= 130-1435 m). Individuals used 3 times more beach area and 1.5 times more shoreline distance in 2003 than in 2004. Females used smaller areas than males overall and during chick rearing. Home ranges were smaller on beaches with low public use, suggesting that human disturbance may cause greater movement by individual plovers and that large protected areas may be warranted on beaches frequented by the public. Table 1. Space use by breeding male and female Great Lakes Piping Plovers expressed as minimum convex polygon (95% MCP) and linear beach distance (95% linear) for the 2003–2004 breeding seasons. Data are shown for the entire breeding season and separately for incubation and chick-rearing periods, and are summarized separately for males and females when analysis suggested differences in range use between the sexes (see Table 2). *N* Mean SE Range *95% MCP (ha)* Season 35 2.9 0.5 0.4–11.2 Male 17 3.5 0.7 0.6–10.9 Female 18 2.3 0.6 0.4–11.2 Incubation 35 1.8 0.4 0.1–10.7 Chick rearing 35 2.0 0.3 0.1–6.0 Male 17 2.2 0.4 0.2–5.5 Female 18 1.4 0.3 0.1–4.6 *95% Linear (m)* Season 35 475 53 130–1435 Incubation 35 340 40 50–1210 Chick rearing 35 370 40 100–980 Male 17 431 66 130–980 Female 18 317 53 100–930  Fig. 2. Mean (± SE) home range sizes (A) and mean linear distances (B) for male and female Great Lakes Piping Plovers in 2003 and 2004.
Hunt 2013	Upper Missouri River	We studied the nest success and survival of Snowy Plovers nesting on the Missouri River between Nebraska and South Dakota from 2008-2011 by monitoring nests every 2-3 days and banding adults and chicks. In our study, Snowy Plovers nested among Piping Plovers and Least Terns. Currently in our study area, the limited numbers of Snowy Plover pairs do not seem to be having an effect on Piping Plovers sharing the same nesting habitat. However, if numbers increased it is conceivable that Snowy Plovers could compete with Piping Plovers for territories and resources. Nest and adult survival of Piping Plovers at the same study area were higher (DSR: 0.99, adult survival: 0.82; Catlin 2009; Catlin et al. 2011a) than Snowy Plovers in our study, but Snowy Plover chick survival was comparable or better than that of Piping Plovers, depending on the year (0.30-0.67; Catlin et al. 2011b). Because most Piping Plover and Least Tern habitat was inundated in 2011 which had low nest success and high predation, the study sandbars had very high nesting densities, which may have attracted predators (Catlin 2009). American mink that may have been moved out of their normal marsh habitat due to inundation were responsible for the majority of documented nest and chick predation.
Maxson 2000	Minnesota	My objective was to quantify interspecific interactions of Piping Plovers during the incubation and brood periods and to determine whether these interactions were likely to be detrimental to Piping Plover reproductive success. Management activities at this site include controlling mammalian predators, placing wire mesh predator exclosures (Melvin et al. 1992) around plover nests, and using elevated-string gull deterrents (Maxson et al. 1996) near plover nests. Protected nests have high hatching success, yet typically fewer than half the chicks fledge. Samples (30 min observations) of n = 111 were obtained during 8June-21July 1995 and between 0600-2010 h (CST). I defined an interaction opportunity as occurring any time a plover was within five m of another species. During the brood period, I noted when Piping Plover chicks were within two m of another species and whether.any interactions occurred. These distances were arbitrary and conservative as some interactions were initiated at greater distances. The interaction index for all species peaked during the brood period reflecting the increased aggressiveness of Piping Plovers after chicks hatched. No measure of nesting density.
Catlin 2009	Upper Missouri River	From our nest monitoring data, we calculated the maximum number of nests active at one time on each sandbar and summed the habitat availability and number of active nests to estimate the average nesting density on natural and engineered sandbars.
Cohen et al. 2009	East coast US	 Figure 10. Correlation between piping plover population growth rate (λ) and nesting-pair density at West Hampton Dunes (WHD) and the reference area (REF), New York, USA, 1993–2004, n = 11 yr.
Wilcox 1959	East coast US	Seldom will one pair nest nearer than 100 feet from the nest of another pair. Nests found were usually spaced 200 feet or more apart. If a bird is on the nest, it usually will chase away adult plovers other than its mate when they approach within about 100 feet. A Least Tern nest was only four feet from a plover's nest and the tern was most aggressive. The author does not say whether or not the plover nest was abandoned.

Habitat Quality and Quantity

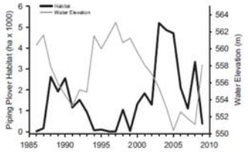
<table>
<tr><th>Reference</th><th>Location</th><th>Findings</th></tr>
<tr><td>Anteau et al. 2014a</td><td>Lake Sakakawea</td><td>Using atmospheric corrected Landsat 5 Thematic Mapper Normalized Difference Vegetation Index data, we developed procedures to measure breeding habitat abundance and we developed a model to predict habitat abundance based on past and projected water levels, vegetation colonization rates, and topography. Previous studies define plover habitat as flat areas (<10%slope) with ≤30% obstruction of bare substrate. Compared to ground-based data, remotely-sensed habitat classifications (≤30/>30% bare-substrate obstruction) were 76% correct and omission and commission errors were equal. Due to water level fluctuations, habitat abundance varied markedly among years (1986–2009) ranging from 9 to 5195 ha. We calculated slope for SAK on a digital elevation model(DEM; 5 m pixels) with vertical accuracy <1 m (Intermap Technologies, Inc., Englewood, CO). Specifically, slope was the maximum elevation change of each pixel to the 8 neighboring pixels; this calculation represented the maximum slope for a 5-m distance. We extracted slope values for 388 plover nests at SAK; 90% of plover nests had slopes <10% (defined as the threshold for suitable-flat habitat). We calculated the total hectares of suitable-flat habitat area at SAK within each 1-m-elevation con-tour (hereafter contour) for each segment. We evaluated our slope threshold with a separate sample of 1320 nests found on SAK during 1998–2005 by a monitoring crew (U.S. Army Corps of Engineers data); 93% of plover nests were situated in areas with less than 10% slopes. We used approximately half of our field observations to develop models to predict habitat using satellite imagery (n = 2355), and the other half to evaluate the accuracy of those models (n = 2116). We classified each pixel as >30% or ≤30% bare-substrate obstruction. Indeed, 81% of nest locations were situated in pixels with ≤30% bare-substrate obstruction.
 

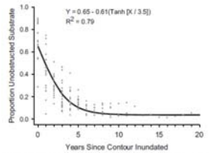
Fig. 8. Annual summaries of piping plover habitat (ha × 1000) for Lake Sakakawea 1986–2009, including water-surface elevations (m MSL). Adapted from U.S. Army Corps of Engineers (2010).
 

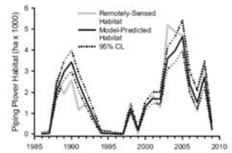
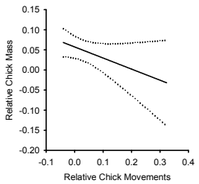
Fig. 9. Relationship of the years since a 1-m-elevation contour was last inundated and the proportion of unobstructed substrate (≤30% cover of vegetation or debris) for Lake Sakakawea.
 

Fig. 11. Remotely-sensed and model-predicted (±95% CL) habitat for piping plovers at Lake Sakakawea, 1986–2009.</td></tr>
<tr><td>Anteau et al. 2014b</td><td>Lake Sakakawea</td><td> 

We also examined how landform, wind fetch, and chick density influenced chick growth rates at our 2-km segment scale. Mean fetch for segments averaged 1.5 km and ranged from 0 to 10.4 km (CV=1.49). While holding wind fetch and density at median values, fledging rate was 4.3 times greater on the mainland (x̄=3.0; 95% CL: 2.9, 3.2) than it was on islands (x̄=0.7; 95% CL: 0.5, 0.9). Chick fledging rate declined by -0.7 (SE=0.04) fledglings per 2-km segment for every km increase in wind fetch. Based on our second-ranked model, relative chick growth was negatively correlated with relative movement (Fig. 4). Regardless, our findings suggest that quantity or quality of habitat at SAK did not appreciably limit fledging rate of plovers on most areas during our study years. Further, if our study years are representative of long-term habitat quality, then 5-15 hatchlings per ha could be used to help define a brood-rearing habitat goal for plovers at SAK. Growth rates we documented were greater than those reported for plovers on sandbars of the Missouri River near Bismarck, ND and Yankton, SD during 2001–2003 (Le Fer et al. 2008a). These comparisons suggest that forage conditions on SAK were similar to other areas that have been studied and perhaps are of no greater management concern than at other plover breeding sites.</td></tr>
<tr><td>Goodale et al. 2007</td><td>Northeast Canada</td><td>Used Lidar data over a barrier beach and estuary on the south shore of Nova Scotia. Logical filters improved classification of habitat compared to standard unsupervised and supervised classification.
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Table 1. Logical and hierarchical statements used in the logical filter for the intermediate and final classifications of Johnston's Pond.
<table><tr><th>Classification</th><th>Logical statement</th></tr><tr><td>Beach substrate</td><td>If mixed substrate overlaps sand and cobble, then classify as mixed
If sand overlaps cobble, then classify as sand</td></tr><tr><td>Intertidal flats</td><td>If beach substrate overlaps mudflat, vegetated mudflat, or sandflat, then keep beach substrate classification
If mudflat overlaps vegetated mudflat and sandflat, then classify as mudflat
If vegetated mudflat overlaps sandflat, then classify as mudflat
Merge mudflat and sandflat into one class (mudflat–sandflat)</td></tr><tr><td>Thirty metre (30 m) coastline buffer</td><td>If vegetated mudflat (from intertidal flats output) is within 30 m of the coastline, then classify as mudflat–sandflat</td></tr><tr><td>Intertidal–beach classification</td><td>Merge 30 m coastline buffer output with intertidal flats output</td></tr><tr><td>Patchy vegetation</td><td>If thick vegetation overlaps sand, mixed, or cobble, then classify as patchy vegetation</td></tr><tr><td>Final classification</td><td>Merge trees–shrubs over patchy vegetation over thick vegetation over water over intertidal–beach classification</td></tr></table>Table 2. Logical statements used to classify critical piping plover nesting and feeding habitats.
<table><tr><th>Piping plover habitat classification</th><th>Logical statement</th></tr><tr><td>Critical feeding</td><td>If slope is less than or equal to 3.0° and beach habitat is classified as mudflat–sandflat or sand, then classify as feeding habitat</td></tr><tr><td>Critical nesting</td><td>If slope is less than or equal to 3.0° and beach habitat is classified as sand, mixed, or patchy vegetation, then classify as nesting habitat</td></tr></table></td></tr>
<tr><td>Haig et al. 2011</td><td>General</td><td>TABLE 1. Issues addressed by avian conservation genetics.
<table><tr><th>Taxonomy</th><th>(Meta) populations</th><th>Landscapes</th></tr><tr><td>• What are the evolutionary and geographic boundaries of species, subspecies, and management units?
• What is the extent of introgression or hybridization?
• Which species are represented in illegal wildlife trade?
• What cryptic species have been misclassified?
• How is biodiversity changing in response to climate change?
• What is the identity of a hybrid, cryptic specimen, or the remains of an ancient or recent specimen?
• What is the best avian tree of life for prioritizing biodiversity conservation by phylogenetic diversity?</td><td>• What are levels of genetic diversity, population structure, effective population size, or gene flow and how have they changed over time?
• What is the best strategy for managing small populations to maximize conservation of genetic diversity (pedigree analyses)?
• What is the extent of population connectivity or isolation?
• Which populations would be the best source for a translocation or reintroduction?
• How do individuals and populations move throughout the annual cycle?
• How are disease transmission pathways linked to bird movements?</td><td>• What are the effects of landscape features and landscape heterogeneity on genetic diversity and population structure?
• How are phylogeographic patterns of birds changing in response to climate change?
• Which populations or life stages are most affected by contaminant exposure; are certain genotypes more vulnerable?
• In mixed-stock populations, what proportion of each stock is being harvested or affected by environmental perturbations?</td></tr></table></td></tr>
</table>

Reference	Location	Findings
Le Fer et al. 2008a	Missouri River	We studied plovers on 3 stretches of the Missouri River (2 riverine and one reservoir) and in the alkali wetlands of North Dakota from 2001 to 2003. We tested the hypothesis that piping plover habitat quality and chick survival on the Missouri River were lower on a cold-water reservoir and downstream from a hypolimnetic (cold-water) release dam with diel water fluctuations (Garrison Dam) than downstream from an epilimnetic dam (Gavins Point Dam). Chicks gained weight more rapidly in the alkali wetlands than on epilimnetic and hypolimnetic river reaches. Invertebrate numbers and biomass were higher in the wetlands and epilimnetic reach, but chick survival was lower on the epilimnetic reach. Thus, piping plovers adapted to a variety of prey densities, and other factors, likely predation, reduced survival rates in the epilimnetic reach.

Table 1. Mean invertebrate biomass (mg dry wt), invertebrate numbers, Coleoptera biomass (mg), Coleoptera numbers ± standard error per set of sticky traps, and individual invertebrate biomass (mg) in saturated and moist habitats during the chick-rearing period. We collected invertebrates in North and South Dakota, USA, 2001–2003, on paint stirrers covered with Tanglefoot insect coating and placed for 30 minutes. If global model was significant, we show main effects.

Site	2001 n	2001 x̄	2001 SE	2002 n	2002 x̄	2002 SE	2003 n	2003 x̄	2003 SE	2001–2003 n	2001–2003 x̄	2001–2003 SE
Invertebrate biomass[a]												
Alkali	7	35.9 A[b]	17.4	8	40 A	12.9	8	9.5 A	2.4	23	22.2 A	6.1
Hypolimnetic	6	5.3 B	1.4	10	4.5 B	1.4	8	2.9 B	0.3	24	4.1 C	0.7
Epilimnetic	8	24.9 Aa[c]	4.9	11	17.2 Aa	3.7	9	6 Ab	1.2	28	15.4 A	2.3
Reservoir	7	15.7 Ba	5.4	10	4.8 Bb	1.3	4	4 ABb	1.3	21	8.3 B	2.2
Invertebrate no.[d]												
Alkali	7	29.3 Aab	7.3	8	49.8 Aa	15.5	8	12.9 b	3.7	23	30.7 A	6.6
Hypolimnetic	6	4.5 C	0.8	10	5.9 C	1	8	7	1.2	24	5.9 C	0.6
Epilimnetic	8	11.9 ABa	1.4	11	13.8 Ba	3.6	9	5.1 b	0.7	28	10.5 B	1.6
Reservoir	7	10.7 Ba	2.7	10	8.2 BCb	1.4	4	7.6 b	3	21	8.9 BC	1.2
Coleoptera biomass[e]												
Alkali	7	0.4 B	0.2	8	0.1 B	0	8	0	0	23	0.2 B	0.1
Hypolimnetic	6	0.7 B	0.6	10	1.2 B	1.1	8	0.2	0.1	24	0.8 B	0.5
Epilimnetic	8	12.8 Aa	4.4	11	10.5 Aa	3.5	9	1.5 b	0.5	28	8.3 A	2.1
Reservoir	7	2 B	1.6	10	0.5 B	0.5	4	0.1	0	21	0.9 B	0.6
Coleoptera no.[f]												
Alkali	7	0.2 BC	0.1	8	0.1 B	0.1	8	0.1	0.1	23	0.2 B	0
Hypolimnetic	6	0.1 C	0	10	0.1 B	0.1	8	0.1	0	24	0.1 B	0
Epilimnetic	8	2.3 Aa	0.6	11	1.1 Aab	0.3	9	0.7 b	0.3	28	1.3 A	0.3
Reservoir	7	1.6 B	1.1	10	0.2 B	0.1	4	0.3	0.2	21	0.7 B	0.4
Individual invertebrate biomass[g]												
Alkali	7	1.1 a	0.3	8	0.5 Bb	0.1	8	1 ABab	0.3	23	0.9 B	0.2
Hypolimnetic	6	1.3	0.3	10	1 B	0.4	8	0.5 C	0.1	24	0.9 B	0.2
Epilimnetic	8	2 a	0.2	11	1.6 Aab	0.4	9	1.2 Ab	0.2	28	1.6 A	0.2
Reservoir	7	1.5 a	0.2	10	0.6 Bb	0.1	4	0.7 BCb	0.3	21	0.9 B	0.1

[a] Invertebrate biomass = sum of individual invertebrate biomass trapped/30 min. Analysis of variance (ANOVA) on ranks: site: F = 19.9, $P < 0.001$; yr: F = 15.2, $P < 0.001$; site × yr: F = 1.4, $P = 0.21$.
[b] Site means with the same letter within columns are not significantly different ($\alpha = 0.05$).
[c] Site means with the same letter within rows are not significantly different ($\alpha = 0.05$).
[d] ANOVA on ranks: site: F = 13.1, $P < 0.001$; yr: F = 3.9, $P = 0.06$; site × yr: F = 2.7, $P = 0.02$.
[e] ANOVA on ranks: site: F = 17.5, $P < 0.001$; yr: F = 3.3, $P = 0.04$; site × yr: F = 0.66, $P = 0.7$.
[f] ANOVA on ranks: site: F = 20.3, $P < 0.001$; yr: F = 2.1, $P = 1.3$; site × yr: F = 1.1, $P = 0.36$.
[g] Individual invertebrate biomass = sum of individual invertebrate biomass/no. individuals. ANOVA on ranks: site: F = 13.9, $P < 0.001$; yr: F = 14.5, $P < 0.001$; site × yr: F = 1.6, $P = 0.15$.

Table 2. Mean foraging rates (pecks/min), proportion of time foraging, and estimated biomass consumed (mg/min) ± standard error for piping plover chicks aged day 3–10 and day 11 to fledging, in North and South Dakota, USA, 2001–2003. Results of analysis of variance (ANOVA) on ranks comparing sites.

Site	Day 3–10 n	Day 3–10 $\bar{x}$	Day 3–10 SE	Day 11–fledge n	Day 11–fledge $\bar{x}$	Day 11–fledge SE
Foraging rate[a]						
Alkali	21	5.8 B[b]	0.9	15	6.5 B	1.5
Hypolimnetic	16	10.8 A	1.9	11	14.2 A	1.3
Epilimnetic	21	6.7 B	0.7	14	9.2 B	0.7
Reservoir	13	7.2 B	1.3	4	8.4 B	3.4
Proportion of time foraging[c]						
Alkali	21	0.43 B	0.06	19	0.51 B	0.09
Hypolimnetic	16	0.69 A	0.07	12	0.82 A	0.05
Epilimnetic	21	0.78 A	0.06	14	0.83 A	0.04
Reservoir	13	0.64 A	0.08	5	0.53 B	0.18
Biomass consumption index[d]						
Alkali	21	5 C	1	15	5.6 B	1.2
Hypolimnetic	16	9.7 AB	2.1	11	12.7 A	2.6
Epilimnetic	21	10.6 A	1.2	14	15.1 A	1.7
Reservoir	13	6.2 BC	0.9	4	9.6 B	1.8

[a] ANOVA on ranks: day 3–10: $F_{3,70} = 3.3$, $P = 0.02$; day 11 to fledge: $F_{3,43} = 5.1$, $P = 0.004$.
[b] Site means with the same letter within columns are not significantly different ($\alpha = 0.05$).
[c] ANOVA on ranks: day 3–10: $F_{3,70} = 5.4$, $P = 0.002$; day 11 to fledge: $F_{3,44} = 4.7$, $P = 0.007$.
[d] Biomass consumption index = average invertebrate individual biomass × foraging rate; ANOVA on ranks: day 3–10: $F_{3,70} = 3.9$, $P = 0.001$; day 11 to fledge: $F_{3,43} = 5.9$, $P = 0.001$.

Table 3. Slopes of regressions of natural log of mass of piping plover chicks on age in days at 4 sites in North and South Dakota, USA, in 2001–2003. Mass was the mean mass of surviving chicks in each brood for each chick age.

Site	2001–2003[a] Slope	2001–2003[a] SE	2001[b] Slope	2001[b] SE	2002[c] Slope	2002[c] SE	2003[d] Slope	2003[d] SE
Alkali	0.081 A[e]	0.002	0.086 AB	0.004	0.079 AB	0.002	0.083 A	0.002
Hypolimnetic	0.076 B	0.001	0.078 B	0.003	0.076 B	0.002	0.077 B	0.002
Epilimnetic	0.077 B	0.001	0.092 A	0.003	0.081 A	0.002	0.067 C	0.002
Reservoir	0.078 AB	0.002	0.08 B	0.005	0.078 AB	0.002	0.075 B	0.003

[a] $F_{3,581} = 1641$, $P < 0.001$.
[b] $F_{3,148} = 481$, $P < 0.001$.
[c] $F_{3,237} = 738$, $P < 0.001$.
[d] $F_{3,196} = 747$, $P < 0.001$.
[e] Sites with the same letter within yr (columns) are not significantly different ($\alpha = 0.05$).

Maslo et al. 2012	East coast US	We evaluated the success of a restored piping plover breeding habitat in New Jersey. We identified the major factors influencing foraging rates, compared foraging activity budgets over 3 yr at restored and natural habitats, and explored the potential of artificial tidal ponds as a viable restoration alternative. Adult foraging rates were higher in artificial pond and ephemeral pool habitats, during low tide, and after breeding activity ended. Adult foraging rates were impeded by the presence of people and vehicles within 50 m. Chick foraging rates were highest at artificial ponds and bay shores and lowest in dunes and on sand flats. Chick foraging rates were strongly hindered by the presence of corvids and the number of people within 50 m. In addition, at artificial tidal ponds, piping plovers spent more time foraging and less time engaged in defensive behaviors (vigilance, crouching, and fleeing) compared to other potential habitats. Our findings support the hypothesis that artificial tidal ponds are a valuable, perhaps superior, foraging habitat.

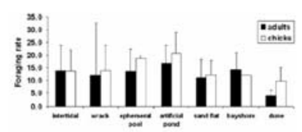
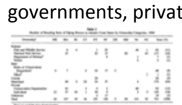
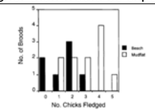
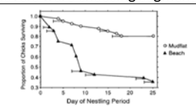
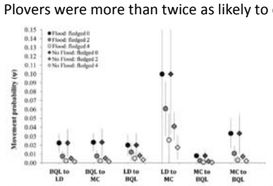


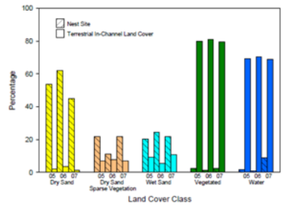
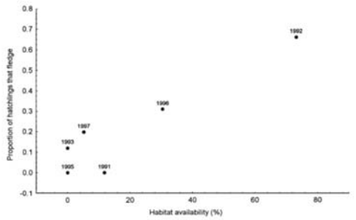
Figure 2. Mean foraging rates (pecks/min) and standard deviations for piping plover adults and chicks in each foraging habitat within New Jersey during the 2007–2009 study. Absence of error bars indicates that only 1 individual was observed in that habitat.

Melvin et al. 1991	Atlantic coast	Piping plovers that nest and winter along the U.S. Atlantic coast are threatened by cumulative impacts of habitat loss and alteration, human disturbance, and predation. Coastal stabilization activities may degrade plover habitat by altering natural processes of dune and beach erosion and accretion. Dredging and beach nourishment projects may be beneficial or detrimental to plovers, depending on timing and location. Management strategies to reduce disturbance and mortality caused by beachgoers and off-road vehicles include fencing of nesting and brood-rearing areas, partial or complete beach closures to pedestrians or vehicles during the breeding season, restrictions on pets, and public education. Beaches where piping plovers nest are owned and managed by a variety of federal and state agencies, county and municipal governments, private conservation organizations, and individuals.



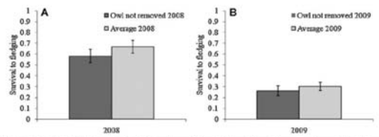
Root and Ryan 2004	North Dakota	We chose two alkaline wetland complexes in North Dakota to investigate long-term vegetation change (1938 to 1997) We obtained black-and-white aerial photographs (1: 7,920 scale) from the U.S. Department of Agriculture Aerial Photography Field Office and U.S. National Archives to measure long-term changes in piping plover breeding habitats. Photographic series for all study wetlands began in 1938 and were available every 3– 15 years through 1997 (n 5 8–9 photographs per wetland). Among years, the seasonal timing of the photographs varied from April to September.. We interpreted habitats at potential piping plover nesting beaches into five classes: (1) surface water; (2) basin substrates (unvegetated mudflats); (3) unvegetated gravel and sand beaches; (4) sparsely vegetated beaches, where substrates were visible through the vegetative cover; and (5) densely vegetated beaches, where no substrate could be observed through the vegetation. We correlated the five-year precipitation index with areal extent of surface water at each study site to test our a priori assumption that basin water levels were associated with precipitation. We also correlated the quantities of available, unvegetated, and sparsely vegetated upper-beach habitats with the five-year precipitation index. During years of reduced surface water, habitat availability of lower beach increased and potentially provided substantial amounts of suitable nesting substrates.
Schapaugh and Tyre 2012	Platte River	The fundamental goal of conservation planning is biodiversity persistence, yet most reserve selection methods prioritize sites using occurrence data. We describe a method that integrates correlates of persistence for multiple species into a single currency – site quality. Site quality is, in turn, an explicit measure of performance used in optimization. We develop a Bayesian network to assess site quality, which assigns an expected value to a property based on criteria arrayed into a causal diagram. We then use stochastic dynamic programming to determine whether an organization should acquire or reject a site placed on the public market. Our framework for assessing sites and making land acquisition decisions represents a compromise between the use of generic spatial design criteria and more intensive computational tools, like spatially-explicit population models. There is certainly a loss of precision by using site quality as a surrogate for more direct measures of persistence. However, we believe this simplification is defensible when sufficient data, expertise, or other resources are lacking.
Seavey et al. 2011	East coast US	Our study area encompassed the barrier island system of Suffolk County, which spans 93 km of barrier island and peninsula shoreline along the southern coast of Long Island, New York. We modeled two possible responses of plover habitat to sea level rise (SLR): static and dynamic. We determined the extent of habitat change over the next 100 years under several SLR estimates, as well as the interactive effects of coastal development and storm surge. We found that if plover habitat cannot migrate, SLR is likely to reduce breeding areas. However, if habitat is able to migrate upslope and inland, breeding areas could actually increase with SLR. Unfortunately, this potential habitat gain is stymied by human development, which we found to reduce migrating habitat by 5–12%, depending on SLR estimates. We also found that the spatial configuration of developed areas mattered more than intensity of development in blocking the migration of potential habitat area. Our results raise concern over the likelihood of increased conflict between plover habitat protection and human recreation as habitat is likely to become a larger proportion of the barrier islands in the future. Finally, our results highlight risk from the synergism between SLR and coastal storms, as we estimate that a large hurricane could flood up to 95% of plover habitat. To assure the future of plover habitat on these barrier islands, management needs to promote natural overwash and habitat migration, while minimizing development adjacent to future breeding habitat.
Sidle et al. 1992	Platte River	We observed extensive mortality (eggs and chicks) of the endangered interior population of the Least Tern and threatened Piping Plover caused by natural flooding during the 1990 breeding season along the Platte River, Nebraska. Aerial videography of the Platte River before and after the flood revealed a 78% reduction of perennial vegetation on sandbars. The flood scoured vegetation from sandbars and greatly increased the amount of barren sandbar habitat that nesting Least Terns and Piping Plovers use.

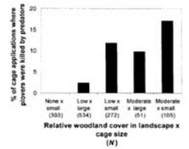
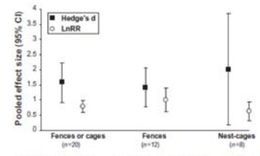
Reference	Location	Findings
		Table 1. The number of monitored Least Tern (LT) and Piping Plover (PP) nests inundated along the Platte River between the Loup River confluence and the Missouri River, 1986-90. Yr; No. Nests Before First Flood; No. Flooded; % Flooded; No. Renests[1]; No. Renests Flooded; Total % Flooded; Maximum Daily Flow (cms) NB/LV[2] LT/PP; LT/PP; LT/PP; LT/PP; LT/PP; LT/PP 1986[3]; 136/28; 18/6; 13/21; 0/0; 0/0; 13/21; 421.7/795.0 1987[3]; 69/17; 5/0; 7/0; 0/0; 0/0; 7/0; 302.8/484.5 1988[4]; 77/32; 3/1; 4/3; 0/0; 0/0; 4/3; 145.2/162.6 1989[5]; 61/13; 21/7; 34/54; 24/5; 0/0; 28/39; 331.1/320.0 1990[6]; 94/28; 94/28; 100/100; 93/13; 4/6; 52/85; 911.3/1713.3 [1]Least terns and piping plovers renested after losing eggs or chicks from other causes in 1986 and 1988 but initiated no new nests on the river after flooding. [2]Flows measured at U.S. Geological Survey gauging stations at North Bend and Louisville, Nebraska. [3]Flooding occurred in late June after most nests had hatched. [4]Flooding occurred in late May-early June before birds had initiated many nests. [5]The first flood inundated sandbars along the entire river and occurred during peak nesting. The second flood inundated sandbars below Salt Creek (U.S. Army Corps of Engineers river mile 26) and occurred in late July after most tern but no plover renests had hatched. The second flood killed all chicks on sandbars below river mile 26.
Sidle et al. 1991	General	We summarize several biological opinions issued by the USFWS to protect the threatened piping plover. The extent of take that is not likely to jeopardize the species is that which will not cause the fledge ratio to drop below 1.44 during a given nesting season. Practically speaking, it remains to be seen if the biological opinions will have an effect on the protection and recovery of the piping plover. It took over four years to produce the Missouri River biological opinion and during that time untimely discharges from dams frequently inundated nests in direct violation of Section 9 of the Endangered Species Act.
Burger 1994	East coast US	**TABLE 3. Habitat comparisons (ocean, dunes, bay) of feeding piping plover and presence of people at three study sites in New Jersey.** Brigantine; Holgate; Corson's Inlet Seconds (±SD) devoted to feeding Ocean: 76 ± 28; 106 ± 22; 87 ± 36 Dune: 50 ± 20; 107 ± 22; 87 ± 41 Bay: 70 ± 24; 109 ± 16; 75 ± 40 Wilcoxon χ^2 (p): 48.3 (0.0001); NS; 6.23 (0.01) People within 100 m (mean ± SD) Ocean: 2.03 ± 0.3; 0.09 ± 0.6; 2.7 ± 2.1 Dunes: 3.3 ± 3.3; 0.02 ± 0.3; 0.08 ± 0.4 Bay: 1.2 ± 1.2; 0.10 ± 0.5; 3.03 ± 2.0 Wilcoxon χ^2 (p): 19.9 (0.0001); 30.5 (0.0001); 11.8 (0.0006)
Cohen et al. 2008	East coast US	All Piping Plover nests in our study were in <50% cover with most on bare ground, but they occurred in vegetation more often than expected based on its availability. Apparent selection of vegetated nest sites may result if plovers nest near vegetated upland zones away from the water line to avoid flooding (Burger
Elias et al. 2000	East coast US	No broods that hatched in one type of beach segment (1 km lengths) switched to another type. Eighty-two percent of broods (n = 80) foraged in 1 segment 100% of the time. Ten percent (n = 10) foraged in 1 segment ≥75% to <100% of the time, and 8% (n = 8) foraged in 1 segment ≥56% to <75% of the time. All beach segments with ephemeral pools or bay tidal flats were used by nesting plovers, whereas fewer. than half of segments without these habitats were used by birds. Proportions of pairs using ephemeral pool or bay tidal flat segments were higher than would be expected if adult plovers were choosing segments at random.
Goldin and Regosin 1998	East coast US	 FIGURE 1. Fledging success for Piping Plover broods with and without access to mudflat habitat at Goosewing Beach, Rhode Island. Brood of five chicks resulted from an inter-pair adoption (see Methods).  FIGURE 2. Survivorship curves of prefledging Piping Plover chicks with access to mudflat and ocean beach habitat at Goosewing Beach, Rhode Island. Error bars (±1) are shown only for mortality events that could not be dated to within two consecutive calendar days.
Knetter et al. 2002	North Dakota	**Table 2. Mean beach characteristics of piping plover nesting territories for the Lostwood and Stateline regions in northwest North Dakota and northeast Montana during 1999.** Lostwood (n^a = 45); Stateline (n^a = 38) Habitat variables; Mean; SE; Mean; SE; t; P % cover vegetation; 26.6; 2.3; 19.5; 2.0; −2.22; 0.03 Spatial heterogeneity vegetation[b]; 108.6; 11.0; 121.4; 14.4; 0.72; 0.47 Gravel cover (no./0.1 m2); 13.8; 1.1; 8.6; 0.8; −3.51; 0.001 Spatial heterogeneity gravel[b]; 99.7; 7.0; 78.8; 6.5; −2.15; 0.03 Spatial heterogeneity cobble[b]; 97.7; 6.7; 59.0; 8.6; −3.57; <0.001 [a] No. nests. [b] CV of distance (Roth 1976). **Table 4. Mean beach characteristics of successful and unsuccessful piping plover pairs for both regions in northwest North Dakota and northeast Montana during 1999.** Successful (n^a=46); Unsuccessful (n^a= 37) Habitat variables; Mean; SE; Mean; SE; t; P % cover vegetation; 23.98; 2.33; 22.67; 2.30; −0.40; 0.69 Spatial heterogeneity vegetation[b]; 116.49; 11.61; 111.62; 13.72; −0.27; 0.79 Gravel cover (no./0.1 m2); 10.65; 0.94; 12.55; 1.34; 1.20; 0.23 Spatial heterogeneity gravel[b]; 89.22; 6.14; 91.38; 8.17; 0.22; 0.83 Spatial heterogeneity cobble; 84.14; 7.66; 76.28; 8.85; −0.67; 0.50 [a] No. nests. [b] CV of distance (Roth 1976). Our results suggested that the landscape pattern variables we analyzed were not associated with piping plover reproductive success. Beach habitat characteristics did not explain variation in piping plover reproductive success. The cause of year-to-year variation in piping plover reproductive success is unknown. We suggest that managers continue using predator exclosures and urge them to protect nests as early in incubation as possible to provide maximum effectiveness. However, predator exclosures should not be used in situations where there is a history of adult plover depredation. In these situations, temporary electric fences may be a better alternative (R. K. Murphy, personal communication). In addition, we suggest managing beach habitat to create extensive habitat that could support high densities of breeding piping plovers wherever possible.
Roche et al. 2012	Great plains Canada	Plovers were ~3× more likely to emigrate from a breeding population following a year of poor fledging success (fledging no chicks), compared with a year of moderate fledging success (fledging two chicks); flooding helped explain Piping Plover movement. We found that female Piping Plovers were more than twice as likely to exhibit breeding dispersal than males following both flood and non-flood years.  FIG. 4. Probability of interannual breeding-season movement (ψ) following flood and nonflood years among three Piping Plover breeding areas in Saskatchewan (BQL = Big Quill Lake, LD = Lake Diefenbaker, and MC = Missouri Coteau) (2002–2009). See Table 1 for movement-specific sample sizes. Diamonds represent movement following nonflood years; circles represent movement following flood years. Vertical lines represent 85% confidence intervals. Estimates were generated from the top-supported model ranked at the annual mean (recaptur value for "fledge," using a sample of 782 marked after hatch-year Piping Plovers (logit(ψ)=μ+sex;μ = sex + flood + fledge).
Sherfy et al. 2008	Missouri River	A focus of the Missouri River Research Program is the need to quantify acreage of emergent sandbar habitat as identified in the Biological Opinion. Evaluate emergent sandbar and river channel habitat variables for predicting nest success, fledgling success, and nest density for least terns and piping plovers on 4 reaches of the upper Missouri River. We are gathering data needed for a model to predict the amount of habitat available to piping plovers at various lake elevations. This model will be more thoroughly developed in future years when more data are available.

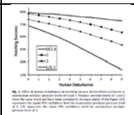
Reference	Location	Findings
Sherfy et al. 2009a	Missouri River	 Figure 2. Distribution of least tern and piping plover nests among land cover classes and the percentage of the land cover class for the Gavins Point Reach in 2005, 2006 and 2007.
Sherfy et al. 2009b	Missouri River	Our work on Lake Sakakawea for Task 5 is an example of both predictive and descriptive modeling for piping plover habitat. We are exploring ways to develop descriptive models for abundance and distribution of breeding habitat on the lake shoreline using a time series of satellite imagery. We are also evaluating approaches for predicting habitat changes and responses by plovers to future changes in lake elevation.
Wemmer et al. 2001	Great Lakes	As expected, the tendency of pairs to search for breeding territories when they encountered a full breeding site had a positive effect on population growth rates, but assumptions about pairing and movements may have the opposite effect on population trend when population size is very small. For example, in the model, Our model suggests the Great Lakes population has the greatest chance of long-term persistence if, in addition to improving productivity, piping plovers can nest at much higher densities and successfully colonize currently unoccupied habitat. Piping plovers once bred commonly
Espie et al. 1998	Great plains Canada	 Fig. 2. Scatter diagram of proportion of piping plover hatchlings that fledged and availability of chick-rearing habitat at Lake Diefenbaker on the mean fledge date (7 July). $\tau = 0.643$, $p = 0.07$, $n = 6$. Observed and predicted fledging success at Lake Diefenbaker from 1988–1997 Year / Observed reproduction / Model I predicted / Model II predicted / Model III predicted 1988[a] 0.7 1.7 1.4 1.7 1989[b] 0.6 1.7 0 0.4 1990[c] 0.0 0.2 0 0.1 1991[d] 0.0 1.6 0.2 0.4 1992 2.0 1.7 1.3 1.7 1993 0.1 0.6 0 0.1 1994[e] 0.7 1.5 0.5 0.8 1995[e] 0.0 1.3 0 0.3 1996[e] 0.6 1.1 0.3 0.6 1997[f] 0.3 0.9 0.1 0.2 Mean 0.49 1.23 0.38 0.63
Johnson et al. 1997	Great Plains and Great Lakes	Loss of sandy beaches and other littoral habitats due to recreational/commercial developments and dune stabilization on the Great Lakes and Gulf of Mexico are partially responsible for the decline of northern Great Plains and Great Lakes populations (Bent 1929, Cairns 1977, Flemming et al. 1988, Haig and Oring 1985, USFWS 1985, and others). Reservoirs, channelization of rivers, and modification of river flows have eliminated sandbar nesting habitat along hundreds of kilometers of the Missouri and its tributaries in the Great Plains (Sidle et al. 1992).
Faanes 1983	Platte River	During 1979, sandbars within about 84 km of river channels were surveyed for tern and plover nests. Nest searches were initiated on 19 May and continued through 26 June. All nests, both hatched and active, were submerged by rising water on 21 June. Normal vegetational succession created some problems interpreting the impact of vegetative growth on nest site selection. When early arrivals began selecting nest sites the sandbars were virtually bare. By mid-June, nesting cover had changed from the date of nest initiation. Thus, later nesting birds were required to choose and evaluate habitat that was much different from that found earlier. The greater mean depth to moisture also suggested that terns preferred higher and drier habitat.
Anteau et al. 2014c	Lake Sakakawea	We examined how remotely-sensed landscape features influenced the distribution of breeding plover pairs among 2-km shoreline segments during 2006–2009 at Lake Sakakawea in North Dakota, USA. Knowledge of landscape features, such as bluffs, that exclude use by birds refines habitat suitability and facilitates more accurate estimates of habitat and population abundance, by decreasing the size of the sampling universe.

Bent, A.C. 1929. Life histories of North American shorebirds. U.S. Natl. Bull. 146:236-246.
Cairns, W.E. 1977. Breeding biology and behavior of the piping plover (*Charadrius melodus*) in southern Nova Scotia. MS Thesis, Dalhousie University.
Flemming, S.P., R.D. Chiasson, P.C. Smith, PJ. Austin-Smith, and R.P. Bancroft. 1988. Piping plover status in Nova Scotia related to its reproductive and behavioral responses to human disturbance. Journal of Field Ornithology 59:321-330.
U.S. Fish and Wildlife Service (USFWS). 1985. Detennination of endangered and threatened status for the piping plover. Federal Register 50(238):50720-34.

Predation

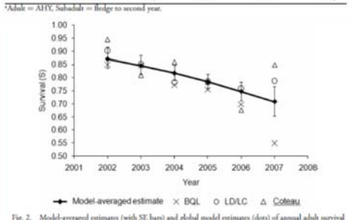
Reference	Location	Findings
Catlin et al. 2011a	Upper Missouri River	We studied the effect of great-horned owl removal on piping plover hatchling survival on Missouri River sandbars (2008–2009). Owl removal increased daily survival of piping plover chicks in 2008 (b = 2.03, 95% CI: 0.04–4.02), but this effect decreased with increasing age of the chick (b = 0.42, 95% CI: 0.81 to 0.03). Results for 2009 were similar in direction but not significant. Survival was higher in 2008 than in 2009, regardless of owl capture, indicating that even if owl capture consistently were effective at increasing survival, overall survival resulting from trapping may vary annually. The minimum distance between the nearest sandbar with an owl trap was 3 km in 2008 and 1 km in 2009.  Figure 2. Probability of survival to fledging (approx. 25 days) of piping plover chicks on the Missouri River with average owl removal and without owl removal in (A) 2008 and (B) 2009. We calculated these values for the mean hatch date for a given year. Error bars represent 1 standard error.
Deblinger et al. 1992	Atlantic coast	Information was obtained on 211 exclosures in 8 states and 3 Canadian provinces. Nests were discovered between 23 April and 12 July (median = 27 May). The exclosures were erected soon after nest discovery (23 Apr-28 Jul; median = 30 May). Nests hatched between 21 May and 3 August (median = 6 Jun). Nests contained 1-4 eggs when they were first located (x = 2.74, SE = 0.079). Exclosures were usually erected around nests with full clutches of 4 eggs (67%); some incomplete clutches also were exclosed (1 egg [2%], 2 eggs [8%], 3 eggs [23%]). Of the 211 nests sampled, 64% were first nesting attempts when exclosed, whereas 25% were renests and 12% were unknown. Predation occurred at only 10% of the nests protected by exclosures. We suggest that exclosures should be made of metal mesh fence (5 x 5 or 5 x 10 cm) and supported by at least 4 sturdy metal or wooden fence posts. Fencing should be at least 90 cm above the sand and 20 cm below the sand.

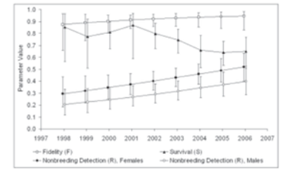
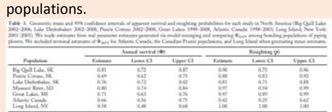
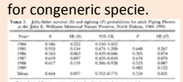
Reference	Location	Findings
Larson et al. 2003	North Dakota	Table 1. Financial costs (US$) and reproductive benefits of applying predator-exclusion management on piping plover nesting beaches in the northern Great Plains of North America. Initial cost to protect 4 breeding pairs[a] Management technique: Materials / Transportation / Labor[c] / Sum; AAEV per pair[d]; Fledglings per pair[b]: Mean / SE No protection: 0; 0.00; 0.89; 0.10 Nest cages: 80; 133; 540; 753; 180.05; 1.28; 0.07 Temporary fencing: 885; 106; 592; 1,583; 224.13; 1.02[e] Permanent fencing: 1,060; 106; 1,000[f]; 2,170; 221.43; 1.15; 0.32 Nest cages and temporary fencing: 965; 106; 608; 1,679; 230.94; 1.78; 0.21 Nest cages and permanent fencing: 1,140; 106; 1,020[f]; 2,266; 228.24; 2.25; 0.37 [a] The average fence protected approximately 4 breeding pairs. Financial cost data were collected in North Dakota and Montana during the 1996–1998 breeding seasons. [b] Fledging rates were published previously by Larson et al. (2002) and represent data from throughout the northern Great Plains during various intervals from 1986 to 1999. [c] Included 13.1 hr/pair for transportation and monitoring. We assumed that removal of cages and temporary fencing required half as long as deployment. [d] AAEV = average annual equivalent value (Farmer et al. 1988:15) using a 5% annual discount rate. It included the costs of application, maintenance, and replacement of materials over 50 years. [e] This fledging rate does not represent field data; it was calculated using equation 1. [f] Time required to erect a permanent fence was estimated by Mayer and Ryan (1991).
Murphy et al. 2003b	Great Plains and Great Lakes	To boost productivity in the Piping Plovers breeding in the northern Great Plains, predator exclosure "cages" constructed of wire mesh fence were placed over 1,355 plover nests on alkali lake beaches in Alberta, Saskatchewan, North Dakota, and Montana during 1993-2002. Nesting plovers were killed, apparently by raptors, near cages at 68 (5%) of the nests. In contrast, no losses of adult plovers were detected at 420 nests that were not covered by cages. The predation was greatest (up to 48% of applications) when small (1-1.7 m) diameter cages with wire mesh tops were used at sites with low (mean, 4%) or moderate (15%) tree cover within two km. In areas with low tree cover, predation decreased to 0.7% of applications/year when large (3-4 m) diameter cages with soft netting tops replaced other designs. No predation was recorded in 393 applications of small cages at plover nests along the relatively treeless North Dakota-Montana border Table 2. Number of applications of predator exclosure cages at which adult Piping Plovers were killed (N_d) and total number of applications (N_{tot}) for two sizes of cages used to protect plover nests at alkali lakes in the northern Great Plains during 1993-2002. Year: Small cages[a] N_d / N_{tot}; Large cages[b,c] N_d / N_{tot} 1993: 0; 8 1994: 0; 9 1995: 0; 41; 1; 20 1996: 10; 89 1997: 4; 111 1998: 9; 122; 0; 110 1999: 27; 198; 11; 121 2000: 0; 44; 4; 116 2001: 0; 75; 1; 122 2002: 0; 73; 1; 96 Total: 50; 770; 18; 585 % 6.5; 3.1 [a] 1-1.7m diameter. [b] 3-4m diameter. [c] Changed from wire mesh top to netting top in 2000.  Figure 2. Frequency of predation on adult Piping Plovers at predator exclosure cages as a function of surrounding woodland cover (none: <1% of landscape wooded within 2 km; low: 4%; moderate: 15%) and cage size (small: 1- to 1.7-m diameter; large: 3- to 4-m diameter), at five alkali lake areas in the northern Great Plains during 1993-2002. Numbers of applications are indicated parenthetically.
Smith et al. 2011	General	Nest predator exclusion is one of the most widely used methods for reducing predation, particularly for ground-nesting species, but studies testing its effectiveness have shown mixed results. We used explicit systematic review methodology to determine the impact of nest predator exclusion on hatching success. We used meta-analysis to summarise results from 16 predator exclusion studies. We also investigated whether factors relating to characteristics of the prey, predator species, location and study methodology explained heterogeneity in effect sizes. Predator exclusion using either exclusion fences or nest-cages resulted in a significant increase in hatching success. This was the case for declining as well as increasing bird populations indicating that nest predator exclusion is an effective method of increasing hatching success of vulnerable species. Nest-cages had a larger effect on hatching success than exclusion fences, although this difference was not significant and additional nest-cage studies are required as sample sizes were small. Heterogeneity in effect sizes was not explained by any of the covariates investigated. We derived effect sizes using Hedges' standardized mean difference, calculated within each study as the treatment effect size relative to the variability observed for each study. We also calculated response ratios for each study, which are defined as the ratio of the means measured in the experimental compared to control areas.  Fig. 1. Pooled effect sizes and 95% confidence intervals for studies of the effectiveness of predator exclosures (fences or nest-cages) for increasing hatching success, where Hedge's d and response ratios (RR) were used as the effect size estimators. Sample sizes (n) are also shown. If nest-cages increase incubating adult mortality rates and juvenile mortality is high in a particular species, even if hatching success is increased by nest-cages, they may have serious detrimental effects on population growth.
Vaske et al. 1994	General	This paper examines the impact of different types of predator exclosures on Piping Plover nest abandonment. The data were obtained from state or provincial agencies. Of the 211 exclosed nests sampled, only 22 (10%) were abandoned. The exclosure construction process was not related to nest abandonment. Exclosure size, shape, mesh size and fence height were also not significant. The odds of nest abandonment, however, were significantly higher for covered exclosures and those lacking fence post.
White et al. 2010	Great plains Canada	We describe 3 cases of predation on Piping Plover nests in Saskatchewan using 24-h video monitoring. However, caution should be taken when using video data to identify predators because a bias in visitation by certain species might exist.
Barber et al. 2010	Northeast Canada	For exclosed nests, 11/16 cases of adult mortality were confirmed to be due to predation, whereas only one of the adults on nonexclosed nests was confirmed as being depredated. However, 7/183 exclosed nests and 46/301 nonexclosed nests failed because of unknown causes.
Doherty and Heath 2011	East coast US	Table 2. Fates of exclosed and unexclosed piping plover nests on Suffolk County Park beaches, New York, in 2006 and 2007. Predator exclosure use increased piping plover hatching success. Fate category Management: Hatching success; Successful n / %; Mixed-fate n / %; Abandoned n / %; Depredated n / %; Flooded n / % Exclosed: 0.68; 20; 49; 11; 27; 7; 17; 1; 2; 2; 5 Not exclosed: 0.56; 26; 39; 10; 15; 10; 15; 14; 21; 3; 5 Total: 0.60; 46; 46; 21; 17; 17; 16; 15; 14; 5; 5 Managers could implement dummy exclosures (exclosures set at random points) to deceive predators by reducing the opportunity for associating exclosures with bird activity (object permanence; Triana and Pasnak 1981). Alternatively, exclosure use could be limited; however, egg loss to predators can become substantial at sites where exclosure use is discontinued. In our study, overall hatching success was higher for nests that had predator exclosures because they effectively deterred egg depredation. However, the ability of exclosures to mitigate predator impacts may be limited in some cases without the additional management strategy of direct predator removal, as both exclosed and unexclosed nests were abandoned in areas with high predator PTI scores.
Ivan and Murphy 2005	North Dakota	The percentage of chicks lost between hatching and fledging stages when no exclosures were available to protect chicks (about 50%) was no different than the percentage lost when chicks were protected from mammals. We thus attributed nearly all predation on chicks to avian sources.
Kruse et al. 2001	Upper Missouri River	Predation accounted for nearly half of all nest loss, the remaining was due to inundation, weather, abandonment, human, and unknown.

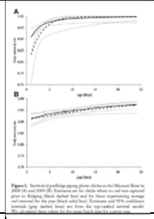
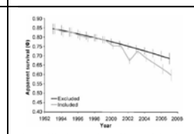
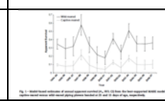
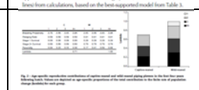
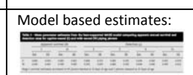
Reference	Location	Findings
Larson et al. 2002	North Dakota	Table 5. Current and proposed numbers of piping plovers in the Great Plains population receiving predator-exclusion management on alkaline wetland sites and their effect on the mean reproductive success rate for all plovers at alkaline wetland sites. Model results: [illegible]
Maslo and Lockwood 2009	East coast US	 The influence of human disturbance is at least as high as that of mammalian predators in our analysis. Sandy Hook, where approximately 55% of the electrified nests in our dataset were abandoned, accommodates over two million (human) visitors per year, and the beaches and access paths are densely populated by recreation seekers throughout the breeding season.
Maxson and Haws 2000	Minnesota	Some management activities at PCI (e.g., predator trapping, predator exclosures) have been effective in increasing hatching success of Piping Plover nests. Use of gull deterrents has caused nesting Ring-billed Gulls to abandon their colony sites making those areas available to Piping Plovers. Nevertheless, low fledging rates continue to be a serious problem for Piping Plovers at LOTW.
Murphy et al. 2003a	North Dakota	Lest results from the use of cages in this study inspire unwarranted optimism, we caution that the cages can expose nesting adult plovers to predation. Since 1996, adult Piping Plovers have been killed by predators at cages at several areas in the northern Great Plains (up to 48% of 34-110 applications/ year in some areas; Murphy et al. 2003b). Besides the loss of at least one adult, these instances almost always resulted in complete loss of eggs or neonatal chicks.
Patterson et al. 1991	Atlantic coast	Table 2. Probable causes of piping plover nest loss, Assateague Island, 1986–87 [illegible]
Rimmer and Deblinger 1990	East coast US	We observed during this study that Piping Plover chicks hatching earlier in the breeding season had, in general, a higher survival rate. Our data show that chicks hatching before 1 July (the traditional beginning of summer in Massachusetts) had a 79% (78/99) survival rate compared to a 26% (4/15) survival rate for chicks hatching after 1 July. Nest loss from predation often stimulates renesting, resulting in young, vulnerable chicks on crowded, disturbed beaches in midsummer. Although our data set is small, it suggests that protection of early nests can reduce renesting and enhance chick survival.
Espie et al. 1996	Great plains Canada	In 1992, we found 30 nests on three different beaches and in 1993 we located 39 nests on four beaches. In total, data from 67 nest sites and 116 random sites were collected; however, slide photographs were only available for 52 nest and 111 random sites. Depredated nests were closer to the vegetation line than successful nests. Table 1. Description of habitat variables used in analysis of Piping Plover nest-site selection. [illegible] Table 5. Sample means and standard deviations of habitat variables for successful, flooded, and depredated nests. [illegible]
Catlin 2009	Upper Missouri River	Adults and juveniles emigrated from the study area at a higher rate after the 2006 breeding season, a year when water discharge was higher, nesting densities were higher, and reproductive success was lower (as a result of predation) than in the other years. Our results suggest that habitat creation could be a viable short-term solution to population declines in shorebird populations limited by habitat loss, but high densities and increased predation associated with habitat creation indicate that other, long-term solutions may be required. Exclosures may not be a long-term solution to predation problems in some cases because of increased adult mortality, predator adaptation, or cost. The negative effects of predation on populations through reduced productivity often are a symptom of other problems such as habitat loss or degradation, increased human population, or introduced species. When evaluating the effect of habitat alterations or other management interventions on a population, managers should evaluate the overall fitness consequences of these actions. For example, high use of an engineered habitat may be counterproductive if the habitat is an ecological trap in which colonizing animals suffer reduced fitness due to low reproductive output or survival. For managed or created habitat to prove an effective conservation tool, it must both be selected for by the species of interest and provide at least comparable fitness relative to the habitat that it is replacing or augmenting. We studied the effects of nest exclosures, habitat modification, and habitat creation by observing plover nest site selection and by comparing the success of plovers nesting on 16 sandbars below Gavins Point Dam with various combinations of these treatments. Our study did however suffer in that the placement of exclosures was not random and there were not equal sample sizes of exclosed and unexclosed nests through time or through space. The USACE monitoring crew was responsible for the exclosing of nests and their monitoring season began after the initiation of the breeding season, nests were not exclosed until clutches were completed, and areas of higher nesting density tended to receive greater management attention (D. Catlin, pers. obs.). Table 2: Raw nest success (% total) for piping plover nests on the Missouri River during the 2005 – 2007 nesting seasons. Known successful nests had one or more chicks associated with the nest or the parents of the nest. Possibly successful nests did not have chicks associated with the nest or adults but eggs disappeared within 2 days of the projected hatch date. Failed nests did not have chicks associated with the nest or adults and disappeared outside the window of reasonable hatch dates (excluding abandoned nests). [illegible]
Johnson et al. 1997	Great Plains and Great Lakes	Success of up to >90% when has been common when success for unprotected nests has been <25% (Rimmer and Deblinger 1990, Deblinger et al. 1992, Melvin et al. 1992).

Melvin, S.M., L.H. MacIvor and C.R. Griffin. 1992. Predator exclosures: a technique to reduce predation at piping plover nests. Wildlife Society Bulletin 20: 143-148.
Triana, E., and R. Pasnak. 1981. Object permanence in cats and dogs. Animal Learning and Behavior 9:135–139.

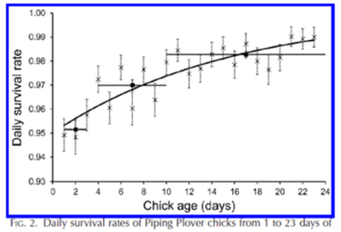
Stage Survival

Reference	Location	Nest Survival	1st Winter Survival	Adult Survival	Study Design	Sample Size
Cohen et al. 2006	East coast US		We calculated juvenile return rate each year as the proportion of marked fledglings that returned to breed. The rates at which juveniles returned to breed were 0.11 ± 0.06 (SE; N = 28) and 0.13 ± 0.09 (SE; N =15) in 2003 and 2004, respectively.	We captured and marked 66 nesting adult Piping Plovers from 2001 to 2005. In addition, 43 banded chicks fledged and 5 of those entered our sample of breeding adults in a subsequent year. Assuming that the probability of observing an animal before it died (R′) of 0.025, apparent survival rate (S), site fidelity (F), and resighting rates of adult Piping Plovers at West Hampton Dunes and Westhampton Beach for 2002–2004 were (mean ±SE) 0.70 ± 0.03, 0.83 ± 0.07, and 0.97 ± 0.01, respectively. Table 1. Number of adult Piping Plovers captured and number of resightings at West Hampton Dunes and Westhampton Beach, Long Island, NY, from 2001 to 2005. Year / New captures[a] / New recruits[b] / Breeders resighted[c] / Nonbreeders resighted[d] / Nonbreeders missed[e] / Total population[f] 2001: 36, 0, 0, 0, 0, 36 2002: 14, 0, 26, 1, 0, 41 2003: 16, 3, 21, 10, 0, 50 2004: 0, 1, 24, 6, 2, 33 2005: 0, 1, 6, 6, —, 13 [a]Number of birds trapped on the nest and marked in year *i*. [b]Number of birds marked as chicks prior to year *i* that nested at our site for the first time in year *i*. [c]Number of marked birds breeding at our site in year *i* that were marked prior to year *i* and nested at our site in year *i* − 1. [d]Number of marked birds resighted in year *i* that did not breed at our site in year *i*, were marked prior to year *i*, and bred at the site in at least one year prior to year *i*. [e]Number of marked birds not resighted in year *i* but resighted in at least 1 year after year *i* that were marked prior to year *i*, and bred at the site in at least 1 year prior to year *i*. [f]Total number of marked birds known to be in the population in year *i*. Table 3. Model-averaged estimates and unconditional confidence intervals on apparent survival rate (*S*), resighting rate (*R*), and apparent site fidelity (*F*) of adult Piping Plovers, West Hampton Dunes and Westhampton Beach, Long Island, NY 2001–2005 (*N* = 71 adults). R′[a] / Year[b] / Survival: S, SE, 95% CI / Resighting rate: R, SE, 95% CI / Site fidelity: F, SE, 95% CI 0.025[c] / 2002 / 0.714, 0.091, 0.551–0.856 / 0.984, 0.022, 0.807–0.999 / 0.945, 0.082, 0.443–0.997 2003 / 0.751, 0.095, 0.527–0.891 / 0.984, 0.022, 0.807–0.999 / 0.707, 0.117, 0.443–0.880 2004 / 0.643, 0.087, 0.462–0.791 / 0.948, 0.051, 0.660–0.994 / 0.830, 0.084, 0.601–0.940 Mean / 0.703, 0.032, 0.637–0.762 / 0.972, 0.012, 0.936–0.988 / 0.827, 0.069, 0.650–0.925 0.000[d] / 2002 / 0.731, 0.092, 0.521–0.871 / 0.984, 0.022, 0.797–0.999 / 0.904, 0.094, 0.532–0.987 2003 / 0.753, 0.091, 0.539–0.888 / 0.984, 0.022, 0.797–0.999 / 0.702, 0.116, 0.443–0.874 2004 / 0.651, 0.087, 0.468–0.798 / 0.946, 0.060, 0.635–0.994 / 0.812, 0.084, 0.595–0.927 Mean / 0.712, 0.031, 0.648–0.769 / 0.971, 0.013, 0.931–0.988 / 0.806, 0.058, 0.668–0.896 0.046[e] / 2002 / 0.721, 0.119, 0.447–0.892 / 0.984, 0.022, 0.813–0.999 / 0.947, 0.078, 0.464–0.997 2003 / 0.738, 0.098, 0.512–0.884 / 0.984, 0.021, 0.813–0.999 / 0.721, 0.119, 0.447–0.892 2004 / 0.638, 0.087, 0.457–0.787 / 0.950, 0.053, 0.677–0.994 / 0.844, 0.084, 0.607–0.950 Mean / 0.699, 0.031, 0.635–0.756 / 0.973, 0.011, 0.941–0.988 / 0.837, 0.065, 0.669–0.929 [a]*R′* = The probability that a bird sighted between 15 March and 15 April died later in that interval without being reported dead. Because we fixed *r* (the probability of a bird being found dead and reported) at 0.0, *R′* equals the mortality rate of resighted birds in the territory establishment period (15 March to 15 April). [b]Year of resighting. [c]Assumes an annual survival rate of 0.74 (USFWS 1996) and constant monthly survival rates. [d]Assumes no mortality of resighted birds in the territory establishment period. [e]Assumes an annual survival rate of 0.74 and that mortality during the territory establishment period is twice as high as in other months. Our mean annual adult survival estimate (0.70) was similar to that for Piping Plovers in Maryland in the late 1980s (0.71; Loegering 1992), Massachusetts in the late 1980s (0.74; USFWS 1996), and the Great Plains during a 10-yr study in the 1980s and 1990s (0.74; Larson et al. 2000).		
Cohen and Gratto-Trevor 2011	Great plains Canada		Average adult and subadult survival rates during our study were 0.80 and 0.57, respectively. Adult survival declined over time, possibly due in part to the loss of one breeding site to flooding. Male and female Piping Plovers had similar survival rates. Based on our survival estimates, productivity needed for a stationary population was 0.75, a benchmark used for plover management on the Atlantic Coast, but not previously estimated for Prairie Canada. In stochastic simulations incorporating literature-based variation in survival rates, productivity needed for a stationary population increased to 0.86, still lower than that previously estimated for western populations. Mean productivity for our study sites ranged from 0.87 to 0.96 fledged young per pair. Our results suggest that fledging rates of Piping Plovers in Saskatchewan were sufficient to ensure a stationary or increasing population during our study period. Table 2. Mean Saskatchewan Piping Plover true survival (*S*), breeding/natal site fidelity (*F*), breeding site recapture probability (*p*), wintering grounds resighting probability (*R* = birds that survived to breed, *R′* = birds that did not), and re-immigration rate of emigrants (*F′*), 2002–2009. Parameter / Age[a] / Estimate / SE / 95% LCL / 95% UCL *S* / Adult / 0.80 / 0.03 / 0.74 / 0.84 Subadult / 0.57 / 0.05 / 0.47 / 0.67 *F* / Adult / 0.86 / 0.04 / 0.76 / 0.92 Subadult / 0.46 / 0.11 / 0.27 / 0.66 *p* / Adult / 0.90 / 0.03 / 0.83 / 0.94 Subadult / 0.84 / 0.16 / 0.34 / 0.98 *R* / Adult / 0.24 / 0.03 / 0.18 / 0.30 Subadult / 0.20 / 0.04 / 0.13 / 0.30 *R′* / Adult / 0.09 / 0.01 / 0.07 / 0.11 Subadult / 0.07 / 0.03 / 0.02 / 0.17 *F′* / Adult / 0.15 / 0.02 / 0.12 / 0.20 Subadult / 0.19 / 0.10 / 0.07 / 0.45 [a]Adult = AHY, Subadult = fledge to second year.  Fig. 2. Model-averaged estimates (with SE bars) and global model estimates (dots) of annual adult survival of Piping Plovers in Saskatchewan, 2002–2007. The global model, *S*(*g*×*t*), was not well supported based on information-theoretic criteria, probably because sample sizes were not adequate to estimate survival for each site x year combination precisely. However, we provide the estimates for reference. LD/LC = Lake Diefenbaker/Lake Chaplin and BQL = Big Quill Lake.	We estimated true survival and site fidelity of adult and subadult (from fledging to second year) Piping Plovers breeding in Saskatchewan using mark-resight data from 2002 to 2009. By estimating true survival rather than apparent survival (which is confounded with permanent emigration), we were able to provide more accurate projections of population trends.	Table 1. Number of Piping Plovers captured in Saskatchewan and number resighted in Saskatchewan and on the wintering grounds, 2002–2009. Age / Year / New captures / Study area resightings / Winter resightings Adult / 2002 / 221 / 0 / 31 2003 / 172 / 140 / 65 2004 / 145 / 245 / 74 2005 / 122 / 297 / 117 2006 / 122 / 246 / 140 2007 / 0 / 161 / 76 2008 / 0 / 114 / 104 2009 / 0 / 66 / 27 Subadult / 2002 / 21 / 0 / 2 2003 / 45 / 3 / 14 2004 / 35 / 18 / 9 2005 / 53 / 14 / 23 2006 / 28 / 22 / 23 2007 / 0 / 12 / 15 2008 / 0 / 8 / 17 2009 / 0 / 6 / 9
Drake et al. 2001	Gulf coast			High rates of survival and strong site fidelity throughout the nonbreeding period suggest that this period of the annual cycle may not contribute to the declining population size for Piping Plovers wintering in this region. We documented no mortality of marked plovers during fall (n = 705 transmitter days), winter (n = 791 transmitter days), or spring (n = 1208 transmitter days).	We studied movements, habitat use, and survival rates of 49 radio-marked Piping Plovers overwintering along the southern Laguna Madre of Texas during 1997-1998.	We radio-marked and monitored 49 plovers between 10 August 1997 and 25 April 1998. We obtained 1371 radio locations averaging 29.6 :± 1.3 locations/ individual. Mean number of days between successive locations was 1.97 :±: 0.04.
Larson et al. 2000	North Dakota		Fledging success (survival of chicks to ≥16 d of age, Prindiville Gaines and Ryan 1988) of banded juveniles was 96% (n = 55) in 1987. Immature annual survival was 0.318 (SE = 0.075), but true immature	The discrepancy between the previous adult survival estimate for the Great Plains Piping Plover population (0.664, SE 5 0.057) and estimates from other regions and closely related species prompted us	We used published data plus three additional years of band resightings, data from banded juveniles, and a new modeling approach to estimate local annual survival rates of adults and	Adult and juvenile Piping Plovers were captured and color-banded at JWP nesting sites in 1984–1987 and 1985–1987. We resighted 97 adults 189 times and 26 juveniles 40 times. Observed band loss was similar between the two resighting periods (1984–1990: 12%, n = 221 resightings [includes resightings of birds missing $1 band, which do not appear in Table 1]; 1992–1994: 8%, n = 37; x2 = 0.43, df = 1, P = 0.5).

Reference	Location	Nest Survival	1st Winter Survival	Adult Survival	Study Design	Sample Size
			survival is probably higher, mostly due to unknown but likely high dispersal rates. When plovers banded as juveniles were >1 yr old their local survival was 0.766 (SE = 0.055).	to re-examine banding data for Great Plains plovers. Mean adult survival was 0.737 (SE = 0.092), and the temporal variance was 0.040–0.045.	immatures for a breeding site in central North Dakota in 1984–1994. We separated individual encounter histories into three data sets: birds originally banded as adults, birds originally banded as juveniles, and both adults and juveniles combined.	TABLE 1. Number of Piping Plovers captured and number of band resightings at the Williams Preserve, North Dakota, 1984–1994. Adults: New captures / Resightings; Juveniles: New captures / Resightings 1984: 8 1985: 54 / 1 / 44 1986: 111 / 20 / 34 / 4 1987: 31 / 64 / 65 / 7 1988: 43 / 11 1989: 22 / 6 1990: 14 / 3 1991: 0a / 0a 1992: 15 / 3 1993: 6 / 3 1994: 4 / 3 Total: 204 / 189 / 143 / 40 a No attempt was made to observe color-banded plovers in 1991.
Ledee et al. 2010	Great Lakes			Survival varied substantially by year, with strong evidence from both trend and fully temporal models, declining from 1998 to 2006 (Fig. 1). In any given year, however, the 95% CI around the survival estimate was large and overlapped broadly with that in adjacent years (Fig. 1). From 1998 to 2006 survival averaged 0.755 (σ = 0.051); we excluded the terminal survival rate for 2007 from this overall average because it was poorly estimated (S2007 = 0.515, SE = 0.196).	Using a Barker model and 11 years of mark–resighting data of breeding and nonbreeding Piping Plovers (1998–2008), we estimated true adult survival (S), probability of detection of breeding birds (p), probability of detection nonbreeding birds (R), and site fidelity (F).	 FIGURE 1. Model-averaged parameter estimates and 95% confidence intervals from Barker model for fidelity (F), survival (S), and nonbreeding detection probability (R) of Piping Plovers (n = 150) at the Great Lakes from 1998 to 2006.
Plissner and Haig 2000a	General		Table 1 Baseline metapopulation structure of Atlantic Coast piping plovers[a] Atlantic metapopulation: 1. Atlantic Canada (Newfoundland, Nova Scotia, Prince Edward Island, New Brunswick, Quebec, St. Pierre and Miquelon (France)) K 4000, Initial population size 2340, 1st year/adult mortality 51.64/26.13, Mean no. fledged per pair 1.37 2. New England (Maine, Massachusetts, Rhode Island, Connecticut) K 800, Initial population size 384, 1st year/adult mortality 51.64/26.13, Mean no. fledged per pair 1.28 3. Mid-Atlantic (New York, New Jersey) K 1250, Initial population size 1118, 1st year/adult mortality 51.64/26.13, Mean no. fledged per pair 1.75 4. Southern Region (Delaware, Maryland, Virginia, North Carolina, South Carolina) K 1150, Initial population size 718, 1st year/adult mortality 51.64/26.13, Mean no. fledged per pair 1.07 K 800, Initial population size 320, 1st year/adult mortality 51.64/26.13, Mean no. fledged per pair 0.98 [a] Source: personal communications/unpublished annual reports of authors listed in Acknowledgements. Table 2 Baseline metapopulation structure of Great Lakes/Northern Great Plains piping plovers[a] Metapopulation: 1. Great Lakes (L. Superior, L. Michigan, L. Erie) K 5891, Initial population size 3332, 1st year/adult mortality 56.8/34.0, Mean no. fledged per pair 1.25 L. Superior, L. Michigan, L. Erie K 300, Initial population size 48, 1st year/adult mortality 56.8/34.0, Mean no. fledged per pair 1.46 2. Manitoba and Lake of the Woods (Manitoba and Lake of the Woods) K 250, Initial population size 73, 1st year/adult mortality 56.8/34.0, Mean no. fledged per pair 0.53 3. Northern Prairie (Alberta, Saskatchewan Montana, and Dakotas, N and S Saskatchewan Rivers, include L. Diefenbaker) K 4700, Initial population size 2786, 1st year/adult mortality 56.8/34.0, Mean no. fledged per pair 1.28 4. Nebraska Rivers (Platte, Loup, Elkhorn, Niobrara, and Missouri below Fort Randall Dam) K 500, Initial population size 412, 1st year/adult mortality 56.8/34.0, Mean no. fledged per pair 1.11 5. Colorado (Reservoirs in SE quadrant of state) K 40, Initial population size 13, 1st year/adult mortality 56.8/34.0, Mean no. fledged per pair 0.73 [a] Source: personal communications/unpublished annual reports of authors listed in Acknowledgements.			The metapopulation viability analysis package, vortex, was used to examine viability and recovery objectives for piping plovers.	
Roche et al. 2010b Not available in endnote library.	Great Lakes	Leg bands are commonly used to mark shorebird chicks as young as 1-d old, but little is known about the possible impacts of bands on survival of prefledging shorebirds. We used a mark-recapture framework to assess the impact of bands and banding-related disturbance on prefledging survival in a federally endangered population of Piping Plovers breeding in the Great Lakes region from 2000 to 2008. We banded approximately 96% of all surviving chicks hatched prior to fledging, typically between 5 and 15 d of age. We used a multistate approach in program MARK whereby individuals contributed data as unbanded chicks before capture (N = 1073) and as banded chicks afterward (N = 780). The cumulative probability of surviving through 24 d of age was 0.63 and did not differ between banded and unbanded chicks. In addition, we found a positive effect of banding-related disturbance on survival up to 3 d following banding (beta = 0.60 CI: 0.17-1.02), possibly due to increased postbanding vigilance on the part of chicks and adults. Our results indicate that banding has no detrimental effect on survival of Piping Plover chicks prior to fledging and that current capture and banding methods are appropriate for this endangered species.					
Roche et al. 2010c	General			Average survival estimates were higher for Great Plains populations (range = 0.69–0.81) than for Great Lakes and Atlantic Coast populations (range = 0.56–0.71). Linear trend models indicated that apparent survival declined in 4 out of 7 populations, was unchanged in 3, and was generally highest among Great Plains populations. Based on a post hoc analysis, we found evidence of correlated year-to-year fluctuations in annual survival among populations wintering primarily along the southeastern United States Atlantic Coast and Gulf Coast. Our results indicate shared overwintering or stopover sites may influence annual variation in survival among geographically disparate breeding populations. 	We used long-term (1998–2008) mark–recapture data on piping plovers collected from 7 separate studies located throughout North America to conduct a range-wide analysis of after hatch year apparent survival (ϕ_{AHY}). Our objectives were to compare concurrent survival estimates from disparate breeding sites and determine whether estimates followed similar trends or were correlated among breeding populations with shared wintering grounds. Our analysis included resighting data from 2,040 AHY (after hatch year) piping plovers, including 451 from the Missouri River, 249 from Big Quill Lake, 216 from the Prairie Coteau, 333 from Lake Diefenbaker, 286 from the Great Lakes, 435 from Atlantic Canada, and 70 from New York.	
Root et al. 1992	North Dakota			Mean annual survival of adult Piping Plovers in the Great Plains was 0.664 (SE = 0.057), slightly lower than that reported for congeneric specie. 	Survival was estimated for the Great Plains population of the threatened/endangered Piping Plover from resightings of 352 (214 adult, 138 juvenile) uniquely color-banded individuals in 1984-1990. One hundred one (47.2%) adults (for a total of 166 resightings, ≥ 1 location/bird/year) and 19 (13.8%) juveniles returned to the North Dakota study site in years after banding. Low return rates of juveniles precluded estimation of survival for that age class. TABLE 1. Program JOLLY capture-recapture input data (Leslie Method B Table format, Brownie et al. 1986) for 214 adult Piping Plovers at the John E. Williams Memorial Nature Preserve, North Dakota, 1984–1990. Entries show the number of plovers resighted in year j that were last resighted in year i. Year last resighted (i) / Year resighted (j): 1984: 0, 3, 1, 0, 0, 0, 0 1985: 0, 16, 12, 6, 0, 0 1986: 0, 51, 4, 3, 1 1987: 0, 30, 8, 3 1988: 0, 15, 5 1989: 0, 8 Resighted: 0, 3, 17, 63, 40, 26, 17 Captured: 8, 55, 111, 29, 10, 1, 0

Reference	Location	Nest Survival	1st Winter Survival	Adult Survival	Study Design	Sample Size
Ryan et al. 1993	North Dakota		**Table 1. Means of Great Plains Piping Plover reproductive rate, adult survival rate, and four levels of immature survival rate used in population simulations, with projected extirpation times.** *Reproductive rate ($\bar{X} \pm SE$)[a]* — *Adult survival ($\bar{X} \pm SE$)[b]* — *Immature survival ($\bar{X} \pm SE$)[c]* — *Years to extirpation (min–max)[d]* 0.86 ± 0.09 — 0.66 ± 0.06 — 0.46 ± 0.06 — 44 (25–54) 0.53 ± 0.06 — 56 (28–77) 0.60 ± 0.06 — 81 (29–105) 0.66 ± 0.06 — 120 (36–148) [a] *n = 32; sources: Haig, personal communication; L. Weber, personal communication; Wiens & Cuthbert 1984; Whyte 1985; Wiens 1986; Haig 1987; Haig & Oring 1987; Prindiville-Gaines & Ryan 1988; Schwalbach 1988; Mayer 1990, 1991.* [b] *Source Root et al. 1992.* [c] *Mean values = 70%, 80%, 90%, and 100% of adult survival; see text for additional explanation.* [d] *Value is the first year the mean population size = 0, based on simulations with 500 replications/year; minimum is first year one of the 500 replications = 0; maximum is first year all 500 replications = 0.*	**Table 2. Reproductive and survival rates necessary for stabilization and growth of the Great Plains Piping Plover population, with reproductive rate or survival rates alternately held constant (dashes within columns).** *Population status* — *Chicks fledged per pair (% increase)* — *Adult survival (% increase)* — *Immature survival (% increase)* Recent[a] — 0.86 — 0.66 — 0.60 Stable — 1.13 (31.4) — — — — Stable — — — 0.72 (8.7) — 0.65 (8.7) 1% Growth — 1.16 (34.8) — — — — 1% Growth — — — 0.73 (9.6) — 0.66 (9.7) 2% Growth — 1.19 (38.4) — — — — 2% Growth — — — 0.74 (10.7) — 0.67 (10.9) [a] *Recent population decline = 7.6%; see Table 1 for sources.*		
Aron 2005	Upper Missouri River		31.8% (Larson et al. 2000) 50.7-63.4% (Gaines and Ryan 1988)	63.4 to 73.7% (Larson et al. 2000, Root et al. 1992, Gaines and Ryan 1988)	Citing other studies	
Catlin et al. 2011a	Upper Missouri River	 Overall, our survival estimates were comparable to those from other piping plover studies throughout the species' range (fledging probability = 0.02–0.78).			In 2008, 5 adult owls were captured on 5 of 6 sandbars, and in 2009 6 adult owls were captured on 4 of 5 sandbars (3 owls on one sandbar and 1 owl on each of 3 other sandbars. No statement on number of owls living on each sandbar or the number of chicks taken by owls.	We used only those chicks that were from nests with a known hatch date so that we could accurately determine age (n = 431 chicks). 2008 - N = 222 piping plover chicks and 5 great-horned owls captured. 2009 - N = 209 piping plover chicks and 6 great-horned owls captured.
Le Fer et al. 2008a	Missouri River		There was no relationship between chick daily survival rates and invertebrate numbers or biomass, Coleoptera biomass, or biomass consumption index (Spearman rank correlation, invertebrate no.: r_s = -0.09, P = 0.4, n = 90; invertebrate biomass: r_s = -0.15, P = 0.16, n = 90; Coleoptera biomass: r_s = -0.05, P = 0.61, n = 90; biomass consumption index: r_s = 0.1, P = 0.4, n = 67).		We studied plovers on 3 stretches of the Missouri River (2 riverine and one reservoir) and in the alkali wetlands of North Dakota from 2001 to 2003.	**Table 4.** Piping plover mean chick daily survival rates ± standard error in North and South Dakota, USA, 2001–2003. Results of 2-way analyses of variance comparing daily survival rates among sites.[a] Sample size is the number of broods. Site — n — $\hat{x}$ — SE Alkali — 23 — 0.932 AB[b] — 0.024 Hypolimnetic — 22 — 0.985 A — 0.005 Epilimnetic — 29 — 0.853 B — 0.042 Reservoir — 20 — 0.923 AB — 0.025 [a] Site: $F_{3,94} = 3.25$, $P = 0.03$; yr: $F_{2,94} = 0.01$, $P = 0.99$; site × yr: $F_{6,94} = 0.97$, $P = 0.45$. [b] Sites with the same letter are not significantly different ($\alpha = 0.05$).
Murphy et al. 2003b	Great Plains and Great Lakes			**Table 1. Predation on adult Piping Plovers at predator exclosure cages at alkali lakes in five areas across the northern Great Plains during 1993-2002: number of applications of cages at which plovers were killed (N_d) and total number of applications of cages (N_{tot}).** Year — Hanna N_d — Hanna N_{tot} — Reflex Lakes N_d — Reflex Lakes N_{tot} — Stateline N_d — Stateline N_{tot} — Lostwood N_d — Lostwood N_{tot} — Williams Preserve N_d — Williams Preserve N_{tot} 1993 — — — — — — — — 0 — 8 — — — 1994 — — — — — — — — 0 — 9 — — — 1995 — — — — 1 — 20 — 0 — 2 — 0 — 30 — 0 — 9 1996 — — — — 0 — 17 — 0 — 21 — 10 — 51 — — — 1997 — — — — 1 — 15 — 0 — 35 — 0 — 46 — 3 — 15 1998 — 0 — 12 — 3 — 28 — 0 — 44 — 0 — 114 — 6 — 34 1999 — 13 — 27 — 14 — 41 — 0 — 103 — 11 — 125 — 0 — 23 2000 — 0 — 4 — 4 — 35 — 0 — 40 — 0 — 56 — 0 — 25 2001 — — — — — — — — 0 — 75 — 1 — 78 — 0 — 44 2002 — — — — — — — — 0 — 73 — 1 — 56 — 0 — 40 Total — 13 — 43 — 23 — 156 — 0 — 393 — 23 — 573 — 9 — 190 % — 30 — — 15 — — 0 — — 4 — — 5 — —
Roche et al. 2010a	Great Lakes				Banded birds from 1992-2008. All nests have predator exclosures. In 2008, the detection probability for breeding individuals was > 0.90 during each 10-day period from mid-May through early July.	
Roche et al. 2008	Great Lakes				Model based estimates: 	Banded birds from 1992-2008
Sidle et al. 1992	Platte River	Extremely high flows between modal nest initiation dates and modal fledging dates (6 weeks later) are likely to cause severe mortality.			**Table 1. The number of monitored Least Tern (LT) and Piping Plover (PP) nests inundated along the Platte River between the Loup River confluence and the Missouri River, 1986-90.** Yr — No. Nests Before First Flood LT/PP — No. Flooded LT/PP — % Flooded LT/PP — No. Renests[a] LT/PP — No. Renests Flooded LT/PP — Total % Flooded LT/PP — Maximum Daily Flow (cms) NB/LN[b] 1986[c] — 136/28 — 18/6 — 13/21 — 0/0 — 0/0 — 13/21 — 421.7/795.0 1987[c] — 69/17 — 5/0 — 7/0 — 0/0 — 0/0 — 7/0 — 392.8/484.5 1988[c] — 77/52 — 3/1 — 4/3 — 0/0 — 0/0 — 4/3 — 143.2/162.6 1989[c] — 61/13 — 21/7 — 34/54 — 24/5 — 0/0 — 39/59 — 351.1/320.0 1990[c] — 94/28 — 94/28 — 100/100 — 93/13 — 4/6 — 52/85 — 911.5/1713.3 [a] Least terns and piping plovers renested after losing eggs or chicks from other causes in 1986 and 1988 but initiated no new nests on the river after flooding. [b] Flows measured at U.S. Geological Survey gauging stations at North Bend and Louisville, Nebraska. [c] Flooding occurred in late May-early June before birds had initiated many nests. [d] Flooding occurred in late May-early June before birds had initiated many nests. [e] The first flood inundated sandbars along the entire river and occurred during peak nesting. The second flood inundated sandbars below Salt Creek (U.S. Army Corps of Engineers river mile 29) and occurred in late July after most tern but no plover renests had hatched. The second flood killed all chicks on sandbars below river mile 29.	
Gaines and Ryan 1988	North Dakota			We used limited data on band resightings (n = 64) from this study and from Saskatchewan (Whyte 1985) and Minnesota (Wiens 1986) to calculate a minimum adult annual survival rate of 0.634. Juvenile piping plovers disperse widely, and band sightings (and therefore survival rate data) are lacking. Therefore, we used values of piping plover juvenile survival ranging from 80 to 100% of adult annual survival (0.507-0.634) in our calculations.		

Reference	Location	Nest Survival	1st Winter Survival	Adult Survival	Study Design	Sample Size
Barber et al. 2010	Northeast Canada			Significantly more exclosed (36/183) than nonexclosed nests (19/301) were abandoned (19.7% vs 6.3%, respectively; Chi-square, df = 1, P < 0.0001) and they had significantly more adult mortality, including predation on adults, associated with them (16/183 vs 2/301; 8.7% vs 0.7% respectively, Fisher's Exact Test: P < 0.0001).		
Catlin et al. 2011b	Upper Missouri River	Nests on engineered sandbars had a 2.5-fold higher DSR than those on natural sandbars (log-odds ratio: 2.50, 95% CI: 1.05–5.94), but nests on managed sandbars showed no difference from natural sandbars (log-odds ratio: 0.82, 95% CI 0.42–1.64). The model-averaged predicted DSR for natural sandbars on the median date in the dataset was 0.98 (95% CI: 0.80–1.00) and for engineered sandbars was 0.99 (95% CI: 0.88–1.00), leading to mean nest successes of 47% and 66%, respectively. Table 2. Raw nest success (% total) for piping plover nests on the Missouri River during the 2005–2007 nesting seasons. Known successful nests had ≥1 chick associated with the nest or the parents of the nest. Possibly successful nests did not have chicks associated with the nest or adults but eggs disappeared within 2 days of the projected hatch date. Failed nests did not have chicks associated with the nest or adults and disappeared outside the window of reasonable hatch dates (excluding abandoned nests). [illegible]				
Elias et al. 2000	East coast US	Estimated survival from hatching to fledging for three different habitats.			[illegible]	
Wemmer et al. 2001	Great Lakes		Table 1 Survival estimates for piping plovers Survival rate / Adult / Fledgling–age 1 Empirical for Michigan[a] / 0.73 / 0.24–0.312 Average other sources[b] / 0.74 / 0.32–0.34 Min all sources / 0.64 / 0.17 Max all sources / 0.94 / 0.48 [a] Wemmer, unpublished data. [b] Loegering (1992), Root et al. (1992), Wiens (1986), Cross, MacIvor, in USFWS (1996).		Habitat based population model	14 years (1984-1997) of monitoring data and 6 years (1993-1998) of banding data
Cohen et al. 2009	East coast US	[illegible]	[illegible]			
Catlin 2009	Upper Missouri River	Juvenile survival was negatively related to nesting density on relatively densely populated, newly created sandbars, and positively related to density on the less dense, natural sandbars. We found no evidence of density-dependent adult survival within our study area.	[illegible] Page 85/106, see nest density for definition	[illegible] Page 82/106	Site selected for specific characteristics. Band/re-siting data	Banded 357 adults and 685 chicks
Johnson et al. 1997	Great Plains and Great Lakes		immature (0.60) survival rates	Root et al. (1992) estimated adult survivorship for piping plovers in North Dakota to be 0.66.		
Hunt et al. 2013	Upper Missouri River		Estimates of daily survival rate (ϕ) for chicks in the two treatments did not differ (ϕsingle-capture = 0.984 ± 0.006, ϕmultiple-capture = 0.985 ± 0.006). Similar to previous studies, we found little evidence of increased prefledge mortality associated with frequent handling of Piping Plover chicks.		Lewis and Clark Lake	We evaluated the potential for increased mortality related to frequent handling of chicks with an experiment that compared the survival of chicks handled a single time for banding (N = 48) to chicks handled repeatedly from hatch to fledge (N = 50) during the 2010 breeding season.
McGowan et al. 2014	Great Plains		Overall mean juvenile survival was set at 0.52 (SE = 0.12) and was based on unpublished analyses of mark recapture data (D. Catlin, unpublished data) and is similar to published values from Saskatchewan at 0.57 (SE 0.05; Cohen and Gratto-Trevor, 2011).	Overall mean adult survival was set at 0.78 (SE = 0.03) and was based on unpublished (D. Catlin, unpublished data) and published estimates (Larson et al., 2000; Cohen and Gratto-Trevor, 2011; Roche et al., 2010)	Modeled probabilities of extinction.	
Brudney et al. 2013	Great Lakes	Average daily survival rates of chicks increased from hatch until fledging (~23 days of age), but this pattern exhibited substantial variation among breeding sites and years. Average (± SE) survival to fledging age was 0.556 ± 0.011. Survival declined with hatching date (28 May–25 July). Survival of chicks was reduced by heavy rain (>1 cm) during the first 3 days of life, but <10% of broods were affected. Chick survival decreased with proximity of nests to trees. Because broods that hatched earlier in the			We used a 20-year database (1992–2011) of brood monitoring histories to examine factors that affect Piping Plover chick survival in the Great Lakes population. Our objectives were to (1) estimate age- specific survival of Piping Plover chicks from hatching to fledging age (~23 days old); (2) document annual and spatial variation in survival among years and breeding locations; and (3) examine the importance of weather, hatch date, and nest-site characteristics on chick survival. We evaluated rates and potential causes of mortality for 2,143 chicks from 597 broods at 31 separate breeding sites.	

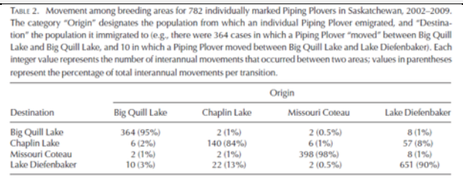
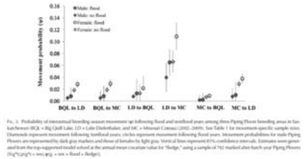
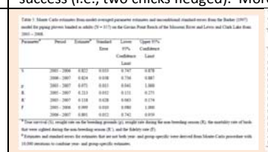
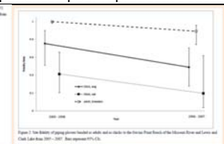
Reference	Location	Nest Survival	1st Winter Survival	Adult Survival	Study Design	Sample Size
		season had higher survival, management efforts should be focused on finding and protecting nests of early breeding pairs.  FIG. 2. Daily survival rates of Piping Plover chicks from 1 to 23 days of age in the Great Lakes region, 1992–2011. Individual points denoted by "×" represent daily survival rates under the fully age-dependent model $S_{(1,2,3,\ldots,23)}$ with their associated 85% confidence intervals (CIs), filled squares are from the three-stage model $S_{(1-3,4-10,11-23)}$, and the curvilinear line represents the logistic regression model: logit(S) = $\beta_0 + \beta_1$*age, where β_0 = 2.948 (85% CI: 2.868–3.028) and β_1 = 0.067 (85% CI: 0.059–0.075).				

Loegering, J. P. 1992. Piping Plover breeding biology, foraging ecology and behavior on Assateague Island National Seashore, Maryland. M.S. thesis, Virginia Polytechnic Institute and State University, Blacksburg, VA.
U. S. Fish and Wildlife Service (USFWS). 1996 Piping Plover (*Charadrius melodus*), Atlantic Coast Population, Revised Recovery Plan, Hadley, MA.
Whyte, A. J. 1985. Breeding ecology of the piping plover (*Charadrius melodus*) in central Saskatchewan. M.S. Thesis, Univ. Saskatchewan, Saskatoon. 126pp.
Wiens,T. P. 1986. Nest-site tenacity and mate retention in the piping plover (Charadriusm elodus).Thesis, University of Minnesota, St. Paul, USA.

Site Fidelity, Immigration, and Emigration

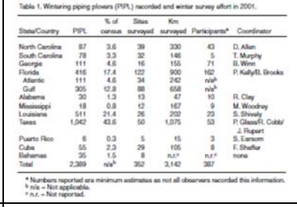
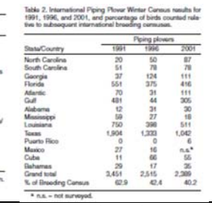
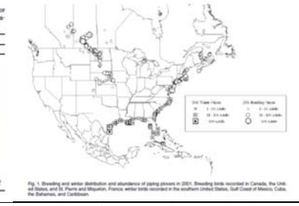
Reference	Location	Findings
Gratto-Trevor et al. 2012	Gulf coast	Resightings of uniquely marked birds from 2001 to 2008 were used to determine winter distributions of 4 breeding populations of piping plover. Very few large-scale movements of individuals in winter were reported either within or between years. This study highlights the significance of geographic regions for eastern Canada, the United States Great Lakes, the United States Great Plains, and Prairie Canada populations, and demonstrates relatively high winter site fidelity. The vast majority (97%) of birds remained in the same region, often at the same beach. Within some regions, however, there was local movement (up to 10 km). For example, marked plovers were recorded on bay and gulf shorelines in southern Texas, depending on the suitability of each habitat, where water levels are affected by tidal cycle as well as by wind direction and intensity. For birds originating from Prairie Canada, of 160 birds observed more than once in a single winter, only 4 cases of movements between regions were noted (2%). Approximately 14 movements (9%) were noted within regions, usually among sites in northern Texas. Considering birds from the Great Plains, 2 of 83 birds (2%) seen more than once in a single winter moved between winter regions, and both probably represented migration. For birds originating in eastern Canada, 16 individuals were seen on more than 1 occasion during a single winter, and none moved among regions. The majority were resighted at the same beach throughout the winter or <10 km away, whereas 4 birds (25%) moved 18–52 km. Of 150 Prairie Canada individuals observed in 2 or more years, movements among regions were reported only 6 times (4%). Of 59 birds marked in the Great Plains that were seen in different winters, 2 were observed in different regions (3%). **Table 1.** Banding locations and numbers of piping plovers uniquely color marked, for each breeding population. **Table 2.** Percentage of individually marked piping plovers from each breeding population observed in different wintering areas along the east coast of the United States and into Mexico up to December 2008. Each individual was counted only once.
Haig and Oring 1988a	Great plains Canada	Individually marked Piping Plovers were studied from 1981-1987 in Manitoba and Minnesota relative to dispersal patterns of age and sex classes. Dispersal distances did not differ between the sexes. Table 1. Breeding-site fidelity and natal philopatry in Piping Plovers. Table 2. Dispersal of breeding Piping Plovers between years. Table 3. Dispersal of Piping Plovers from natal sites to breeding sites. Fig. 3. Dispersal of Piping Plovers from breeding sites to wintering sites, based on 91 sightings of marked birds.
Haig and Oring 1988b	Great plains Canada	Breeding-site fidelity, territory retention, and mate fidelity were examined in a color-banded population of piping plovers breeding at five focal sites in southern Manitoba from 1981 to 1986. Between years approximately 70% of surviving adults were site faithful. Males did not return significantly more often than females, and both sexes returned regardless of previous reproductive success. Although former mates were present in subsequent years, 30 of 37 birds changed mates. Birds that changed mates from the previous year and whose mates were present in subsequent years had experienced poorer hatching success the previous season than those that retained mates. Birds that retained mates did not improve their reproductive success over the previous year. Five pairs that switched mates took approximately 13 ±6.4 days to initiate a new clutch; 11 pairs that remained together took 8.3 ± 1.5 days. Hatching success in second nests where birds retained mates (n = 26) was 42.3%, slightly higher than success for birds that changed mates (35.7% nests produced hatchlings, n = 16). Birds that retained mates were significantly more successful in fledging chicks than birds that chose new mates (X^2 = 11.6, 1 df, P < 0.005). TABLE 1. Annual individual reproductive effort of Piping Plovers breeding in southern Manitoba, 1982–1985. TABLE 2. Population reproductive success of Piping Plovers at Lake Manitoba and West Shoal Lake, Manitoba, 1982–1985. TABLE 3. Breeding-site fidelity of Piping Plovers in southern Manitoba by previous year's reproductive success. Sample sizes (in parentheses) are the number of individuals monitored. TABLE 4. Interyear mate fidelity and territory retention of Piping Plovers breeding in southern Manitoba, 1982–1985. Sample sizes (in parentheses) are the number of individuals monitored. Asterisks indicate significant differences between males and females (χ^2, P < 0.01). TABLE 5. Comparison of previous hatching success among Piping Plovers that retained territories in southern Manitoba with those that changed territories. Sample sizes (in parentheses) are the number of individuals monitored. TABLE 6. Interayear mate and territory fidelity of Piping Plovers breeding in southern Manitoba, 1982–1985.* Sample sizes (in parentheses) are number of individuals monitored. TABLE 7. Intrayear pair-bond retention following destruction of Piping Plover nests in southern Manitoba, 1982–1985.*

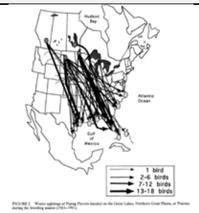
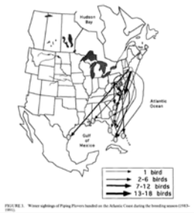
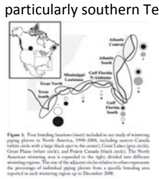
Reference	Location	Findings
Hillman et al. 2012	Great Lakes	More than 7,000 Piping Plovers have been banded and monitored since 1982, yet no individual marked as a hatchling or breeding adult in the range of one subspecies has been reported breeding in the range of the other. Recent molecular genetic analyses further support subspecific taxonomic classification of Atlantic and Interior breeding populations. On 22 May 2011, a banded Piping Plover and unbanded mate were observed nesting on North Core Banks, Cape Lookout National Seashore, North Carolina, USA. The uniquely-banded adult was a female captured as a chick at Wasaga Beach, Georgian Bay, Ontario, Canada; the female nested unsuccessfully in its first breeding year at Tawas Point State Park, Michigan, USA and then successfully fledged two chicks in its second breeding year at North Core Banks, North Carolina, USA. The observation is the first confirmed record of a Piping Plover dispersing from its subspecies' range and successfully breeding in the range of the other subspecies.
Knetter et al. 2001	North Dakota	During July 1998, 4 juvenile plovers fitted with radio-transmitters departed their natal areas, on alkali lakes, in northwestern North Dakota when 24-25 days old. Only 2 were observed again and had moved 50 km southeast and south respectively. They moved again by the following day and could not be relocated. Juveniles may migrate significant distances from natal areas within 3-4 days of attaining flight.
Stucker et al. 2010	Great Lakes	Between 1995 and fall 2005, we obtained 434 observations of banded plovers sighted during migration or winter. We were highly confident of the identity of 154 individuals (x– = 3.1 ± 3.9 observations (±SD), range 1-31 observations). Among these 154 birds, we also confirmed 51 pairs and 45 parent-offspring combinations. Observations were categorized into three seasons as follows: fall migration (n = 152), winter (n = 219) and spring migration (n = 63). Eighty-five plovers were observed during one non-breeding season (e.g. fall 1998 through spring 1999) and the remaining 69 individuals were observed ≥2 non-breeding seasons (mean = 2.6 ± 0.97 nonbreeding seasons (SD)). Of 63 individuals observed recurrently, 33 (52%) exhibited intra-seasonal winter site fidelity, with two or more sightings reported within 3.5 km of initial and subsequent geographic locations within a winter season. Among those 63 individuals, 53 (84%) were documented in subsequent winter seasons, with 30 individuals (62%) identified as inter-annually site faithful in 2-7 winter seasons following their initial observation (x bar = 2.6 ± 0.97 winters (SD)). Moreover, 91% of intra-seasonal site faithful birds were also interannually site faithful.
Cohen et al. 2006	East coast US	We captured and marked 66 nesting adult Piping Plovers from 2001 to 2005. In addition, 43 banded chicks fledged and 5 of those entered our sample of breeding adults in a subsequent year. Assuming that the probability of observing an animal before it died (R') of 0.025, apparent survival rate (S), site fidelity (F), and resighting rates of adult Piping Plovers at West Hampton Dunes and Westhampton Beach for 2002–2004 were (mean ±SE) 0.70 ± 0.03, 0.83 ± 0.07, and 0.97 ± 0.01, respectively. Table 1. Number of adult Piping Plovers captured and number of resightings at West Hampton Dunes and Westhampton Beach, Long Island, NY, from 2001 to 2005. Table 3. Model-averaged estimates and unconditional confidence intervals on apparent survival rate (S), resighting rate (R), and apparent site fidelity (F) of adult Piping Plovers, West Hampton Dunes and Westhampton Beach, Long Island, NY 2001–2005 (N = 71 adults). Site fidelity during the period of territory establishment averaged 0.827 ± 0.069 (SE). Given this high site fidelity, ensuring the survival of adults is important for maintaining local populations. Although the 2002–2004 average was high, site fidelity was apparently lower in 2003 than in 2002. This decrease in site fidelity may have been related to several years of habitat loss and increasing predation of nests or chicks, as reported in a partly concurrent long-term study at our site. The species-level effect of local dispersal due to anthropogenic habitat loss depends on the fitness of dispersers, and is currently unknown for Piping Plovers.
Cohen and Gratto-Trevor 2011	Great plains Canada	Average adult and subadult site fidelity were 0.86 and 0.46, respectively. Adult site fidelity declined during our study at two study sites, most strongly at the flooded site. Males had greater site fidelity than females in some years. Table 2. Mean Saskatchewan Piping Plover true survival (S), breeding/natal site fidelity (F), breeding site recapture probability (p), wintering grounds resighting probability (R = birds that survived to breed, R' = birds that did not), and re-immigration rate of emigrants (F'), 2002–2009. Fig. 3. Model-averaged (mod ave) estimates (with SE bars) and global model estimates (dots) of annual adult site fidelity of Piping Plovers in Saskatchewan, 2002–2009. The global model, S(g+), was not well supported based on information-theoretic criteria, probably because sample sizes were not adequate to estimate survival for each site × year combination precisely. However, we provide the estimates for reference. LD/LC = Lake Diefenbaker/Lake Chaplin and BQL = Big Quill Lake. Fig. 4. Model-averaged estimates (with SE bars) of annual site fidelity of adult male and female Piping Plovers in Saskatchewan, 2002–2007.
Ledee et al. 2010	Great Lakes	We obtained encounter histories for 150 (82 females, 68 males) AHY Piping Plovers uniquely marked at the Great Lakes from the breeding season of 1998 through the winter of 2007–2008. These 150 birds were resighted 324 times in the breeding season (x¯ = 2.2, SD = 2.1 detections per bird) and 147 times in the nonbreeding season (x¯ = 1.0, SD = 1.2 detections per bird). Although resightings represent at least one live encounter on the occasion i, or in the interval i to I + 1, all breeding birds and >40% of all resighted wintering birds were encountered multiple times during each detection period. The majority of winter resightings were from the southeastern Atlantic coast from Carteret County, North Carolina, to Miami-Dade County, Florida or the eastern coast of the Gulf of Mexico from Collier County, Florida, to Mobile County, Alabama. The birds' fidelity to the area of the Great Lakes in which they were monitored was high (x¯ = 0.91, 95% CI 0.66–0.98), with evidence of an increase through time that we attribute to increased intensity of monitoring. Annual return of emigrants (F') was 0.198 (SE = 0.118). FIGURE 1. Model-averaged parameter estimates and 95% confidence intervals from Barker model for fidelity (F), survival (S), and nonbreeding detection probability (R) of Piping Plovers (n = 150) at the Great Lakes from 1998 to 2006.
Wiens and Cuthbert 1988	Minnesota	Distances between nests in successive years were not normally distributed with extremes of 0 to over 15,000 m; the median was 41 m (N = 31). Most (84%) of the movements were less than 200m and occurred within one breeding locality (e.g., Morris Point). Large extremes occurred when birds nested at different localities in successive years. Comparing nest-site tenacity for birds that bred successfully to those that failed, the median year-to-year movement was 24 m and 297 m, respectively (Mann-Whiney U = 35, p = 0.12; excluding all birds that reunited).

Reference	Location	Findings
Roche et al. 2012	Great plains Canada	 TABLE 2. Movement among breeding areas for 782 individually marked Piping Plovers in Saskatchewan, 2002–2009. The category "Origin" designates the population from which an individual Piping Plover emigrated, and "Destination" the population it immigrated to (e.g., there were 364 cases in which a Piping Plover "moved" between Big Quill Lake and Big Quill Lake, and 10 in which a Piping Plover moved between Big Quill Lake and Lake Diefenbaker). Each integer value represents the number of interannual movements that occurred between two areas: values in parentheses represent the percentage of total interannual movements per transition. Origin Destination: Big Quill Lake / Chaplin Lake / Missouri Coteau / Lake Diefenbaker Big Quill Lake: 364 (95%) / 2 (1%) / 2 (0.5%) / 8 (1%) Chaplin Lake: 6 (2%) / 140 (84%) / 6 (1%) / 57 (8%) Missouri Coteau: 2 (1%) / 2 (1%) / 398 (98%) / 8 (1%) Lake Diefenbaker: 10 (3%) / 22 (13%) / 2 (0.5%) / 651 (90%)  Chaplin Lake is ~80 km south of Lake Diefenbaker Our results support this "individual success" effect, suggesting that Piping Plovers were far more likely to move among the study areas following years of poor reproductive success (i.e., no chicks fledged) than following years of moderate reproductive success (i.e., two chicks fledged). Moreover, Piping Plovers that experienced poor reproductive success during a flood year were even more likely to disperse in a subsequent year than following reproductive failure in a nonflood year.
Catlin 2009	Upper Missouri River	 
Johnson et al. 1997	Great Plains and Great Lakes	Following nest failure, most plover pairs remained intact but moved their territories (Haig and Oring 1988b). Breeding site fidelity for piping plovers ranges from 15% in Nova Scotia (Cairns 1977), to 70% in southern Manitoba (Haig and Oring 1988a), to 92% in Minnesota (Haig and Oring 1987). Philopatry does not differ significantly between males and females (Haig and Dring 1988a), although males may return to previously used breeding sites slightly more often than females (Haig and Oring 1988a). Natal site-fidelity is lower than adult philopatry. The percentage of one-year-old plovers returning to natal sites ranged from 4.7% in New York to 20.2% in Minnesota (Wilcox 1959, Wiens 1986). On the Platte River, 18% of plover chicks returned to their natal colonies to breed (Lingle 1993c).

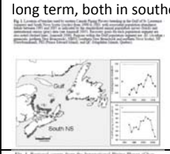
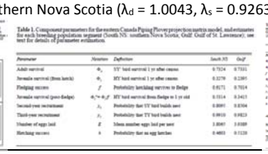
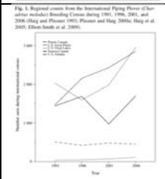
Cairns, W.E. 1977. Breeding biology and behavior of the piping plover (Charadrius melodus) in southern Nova Scotia. MS Thesis, Dalhousie University.
Felio et al. 2010: Two birds banded in Nebraska (Light Blue Flag) were found nesting in the study area in June and July.
Haig, S.M. and L.W. Oring. 1987. Population studies of piping plovers at Lake of the Woods, Minnesota, 1982-87. The Loon 59:113-117.
Lingle. G.R. 1993c. Site fidelity and movements of least terns and piping plovers along the Platte River. Nebraska. Pp. 189-191 in K.F. Higgins and M.R. Brashier, eds., Proceedings. The Missouri River and its tributaries: Piping plover and least tern symposium. South Dakota State University, Brookings.
Wiens, T.P. 1986. Nest-site tenacity and mate retention in the piping plover. M.S. Thesis, U. of Minnesota, Duluth MN.

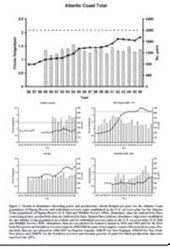
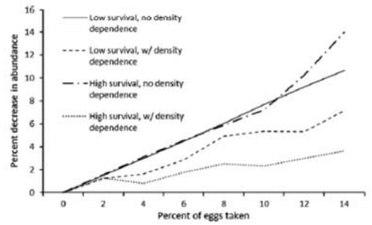
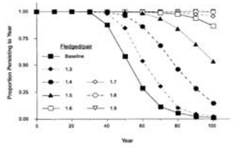
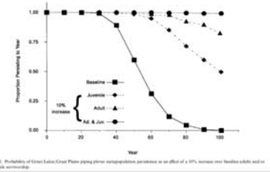
Overwintering Habitat

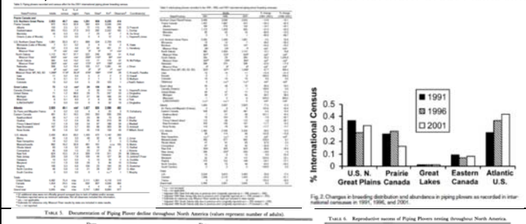
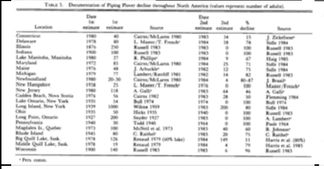
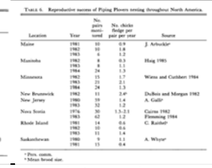
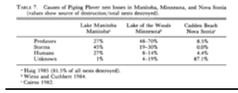
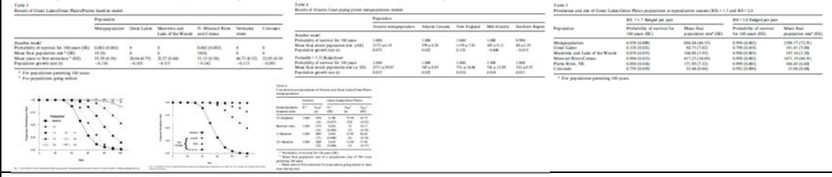
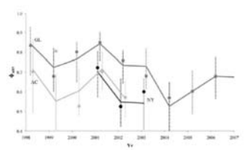
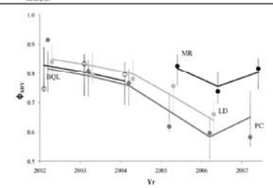
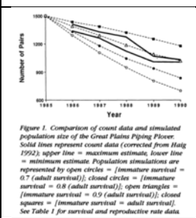
Reference	Location	Findings
Haig et al. 2005	North America	During the winter 2001 census, 2,389 piping plovers were observed at 33.5% of potentially occupied sites (n = 352). Of these, 56.8% had ≤10 birds present. More than 80% of sites with piping plovers present had ≤10 birds.  Table 1. Wintering piping plovers (PIPL) recorded and winter survey effort in 2001. State/Country: PIPL / % of census / Sites surveyed / Km surveyed / Participants[a] / Coordinator North Carolina: 87 / 3.6 / 39 / 330 / 43 / D. Allen South Carolina: 78 / 3.3 / 32 / 146 / 5 / T. Murphy Georgia: 111 / 4.6 / 16 / 155 / 71 / B. Winn Florida: 416 / 17.4 / 122 / 900 / 162 / P. Kelly/B. Brooks Atlantic: 111 / 4.6 / 34 / 242 / n/a[b] Gulf: 305 / 12.8 / 88 / 658 / n/a[b] Alabama: 30 / 1.3 / 13 / 47 / 10 / R. Clay Mississippi: 18 / 0.8 / 12 / 167 / 9 / M. Woodrey Louisiana: 511 / 21.4 / 26 / 202 / 23 / S. Shively Texas: 1,042 / 43.6 / 50 / 1,075 / 53 / F. Glass/R. Cobb/ J. Rupert Puerto Rico: 6 / 0.3 / 5 / 15 / 3 / S. Earsom Cuba: 55 / 2.3 / 29 / 105 / 8 / F. Shaffer Bahamas: 35 / 1.5 / 8 / n.r.[c] / n.r.[c] / none Total: 2,389 / n/a[b] / 352 / 3,142 / 387 [a] Numbers reported are minimum estimates as not all observers recorded this information. [b] n/a = Not applicable. [c] n.r. = Not reported.  Table 2. International Piping Plover Winter Census results for 1991, 1996, and 2001, and percentage of birds counted relative to subsequent international breeding censuses. State/Country: Piping plovers 1991 / 1996 / 2001 North Carolina: 20 / 50 / 87 South Carolina: 51 / 78 / 78 Georgia: 37 / 124 / 111 Florida: 551 / 375 / 416 Atlantic: 70 / 31 / 111 Gulf: 481 / 44 / 305 Alabama: 12 / 31 / 30 Mississippi: 59 / 27 / 18 Louisiana: 750 / 398 / 511 Texas: 1,904 / 1,333 / 1,042 Puerto Rico: 0 / 0 / 6 Mexico: 27 / 16 / n.s.[a] Cuba: 11 / 66 / 55 Bahamas: 29 / 17 / 35 Grand total: 3,414 / 2,515 / 2,389 % of Breeding Census: 62.9 / 42.4 / 40.2 [a] n.s. = not surveyed. 
Haig and Oring 1985	North America	Distributional data were gathered via direct censuses by the author and coordination of census data collected throughout North America between 1982 and 1984. In January 1983, all Texas coastal beaches from Corpus Christi to Galveston were surveyed for potential winter sites. From January to April 1984, 1228 km of Gulf of Mexico beaches, sandflats, and mudflats were ground-surveyed from Campeche(Mexico) to the Florida Keys for presence of Piping Plovers. An additional 960 km of coastal habitat, from Tampico to Matamoros(Mexico), were air-surveyed (for proper habitat) and later ground-censused where access proved feasible. A census of all shorebirds was carried out in 742 km of the surveyed habitat deemed most productive for Piping Plovers. In November 1984, additional surveys and censuses were carried out on beaches and tidal flats from Rio Lagartos to Campeche (Yucatan) and from Corpus Christi to Brownsville (Texas). TABLE 2. Winter distribution and population estimate of Piping Plovers in the United States. State: Estimate: Year Alabama: 80: 1984 Florida: 135: 1984 Georgia: 15: 1983 Louisiana: 33: 1984 Mississippi: 51: 1984 North Carolina: 100: 1983[a] South Carolina: 20: 1984 Texas: 400+: 1984 Total: 834+: 1984 [a] J. Fussell, pers. comm. TABLE 3. Occurrence of Piping Plovers throughout their winter range. Location: Estimate: Year: Source Bahamas: Rare: 1981: R. Norton[a] Barbados: Rare: 1984: M. Hutt[a] Bermuda: 6–8: 1983: D. Wingate[a] Dominican Republic: Rare: 1931: Wetmore and Swales 1931 Cuba: 10+: 1984: O. Garrido[a] Ecuador: 1 seen: 1956: Marchant 1956 Haiti: 2 seen: 1929: U.S. National Museum[b] Jamaica: 0–2: 1983: I. Goodbody, R. Sutton[a] Mexico: 20+: 1984: This study Netherlands-Antilles: 1: 1983: Voous 1983 Puerto Rico: Rare: 1983: Raffaele 1983 United States: 834+: 1984: This study Virgin Islands: 0–5: 1983: J. Yoesema[a] West Indies: Rare: 1982: R. Norton[a] [a] Pers. comm. [b] Museum skin. FIGURE 2. Breeding and winter distribution of Piping Plovers in 1984.
Haig and Plissner 1993	North America	Over 1,000 biologists and volunteers from 10 nations collaborated in the 1991 International Piping Plover Census. Approximately 2,099 sites were censused yielding the highest number of breeding (5,482 adults) and wintering (3,45 1 birds) Piping Plovers ever recorded Most winter birds occurred in Texas (55%) and along other United States Gulf Coast sites (93%). Among winter birds, 5 1% used ocean beaches, 43% used sand or algal flats in protected bays, and 6% used areas where protected bays met ocean beaches. Breeding birds were widely distributed in small populations in the Northern Great Plains/Prairie (63.2%) and on the Atlantic Coast (36%). Few birds (N = 39) remain on the Great Lakes. Habitat use among breeding birds varied considerably across the species range. While most Atlantic (93.9%) and Great Lakes (100%) birds used sandy beaches, 59.6% of Northern Great Plains/Prairie birds used shorelines around small alkaline lakes, 18.2% used large reservoir beaches, 19.9% used river islands and adjacent sand pits, 2% used beaches on large lakes, and 0.4% used industrial pond shorelines.

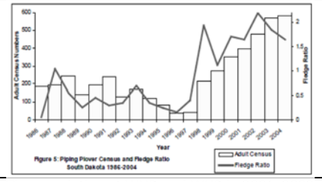
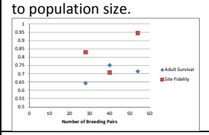
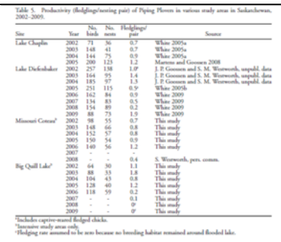
Reference	Location	Findings
		TABLE 1. Numbers of wintering Piping Plovers and winter sites censused in 1991. TABLE 2. Major Piping Plover winter sites in 1991.   TABLE 4. Major Piping Plover breeding sites in 1991.
Ledee et al. 2008	Gulf coast	The purpose of this study was to investigate the relationship between the distribution and abundance of the piping plover and its nonbreeding habitat along the Gulf of Mexico coastline. We identified sites consistently used by wintering piping plovers and quantified landscape and anthropogenic features within a subset of those locations. Using published literature from 1871-2003, we documented that piping plovers consistently winter at 49 locations on the U.S. Gulf of Mexico coast (Marco Island, Florida to Padre Island, Texas). At 31 sites, we conducted a remote analysis of aerial photos for 11 features (e.g., intertidal area, urban area, beach). Linear regression was used to test correlations between plover abundance and landscape characteristics. We found that certain shoreline features (e.g., landform, inter-tidal area, total area) and measures of urbanization (e.g., urban area, length of roads) were significantly correlated, positively and negatively, respectively, with piping plover abundance across the Gulf of Mexico coastline. Table 1. *Results from simple linear regression. From the set of 11 parameters, the five parameters in bold were found to be significant* predictors of piping plover abundance on the U.S. Gulf of Mexico coast.* *$p < 0.05$.
Drake et al. 2001	Gulf coast	We radio-marked Piping Plovers wintering along the coast of southern Texas to determine seasonal variation in habitat use, movement patterns, amount of area used by individuals (home-range size and core area), and survival rates during the nonbreeding period. We conducted our study on South Padre Island and on adjacent mainland tidal flats from Port Mansfield south to the Rio Grande River. South Padre Island is bordered to the west by the hypersaline Laguna Madre and to the east by the Gulf of Mexico. Plovers remained within the southern Laguna Madre during fall, winter, or spring.
Gratto-Trevor et al. 2012	Gulf coast	Resightings of uniquely marked birds from 2001 to 2008 were used to determine winter distributions of 4 breeding populations of piping plover. Although considerable overlap exists, a distinct pattern in winter distributions was evident. Birds originating from eastern Canada wintered farthest north compared to other populations. Most individuals from the United States Great Lakes were found in South Carolina and Georgia in winter, whereas birds from eastern Canada were found primarily in North Carolina. Although the great majority of birds marked in Prairie Canada were observed wintering in Texas, particularly southern Texas, breeding plovers from the United States Great Plains were more widely distributed on the gulf coast from Florida to Texas.  **Table 3.** Percentage of marked individual piping plovers from each breeding population observed at specific locations along the eastern coast of the United States in the winters of 2006–2007 and 2008–2009. [a] Corrected for the number of banded birds estimated alive in each breeding population. [b] By 2008 few, if any, eastern Canada plovers were individually recognizable due to removal of color bands and abrasion of codes on remaining bands.
Stucker et al. 2010	Great Lakes	Between 1995 and fall 2005, we obtained 434 observations of banded plovers sighted during migration or winter. We were highly confident of the identity of 154 individuals (x– = 3.1 ± 3.9 observations (±SD), range 1-31 observations).

Population Trends through Time

Reference	Location	Observed Trend	Study Design
Calvert et al. 2006	Northeast Canada	Annual population censuses suggested moderate growth in abundance between 1998–2003, but vital rate estimates indicated that this temporary growth may be replaced by declines in the long term, both in southern Nova Scotia (λ_d = 1.0043, λ_s = 0.9263) and in the Gulf of St. Lawrence (λ_d = 0.9651, λ_s = 0.8214).  	The endangered eastern Canadian breeding population of Piping Plover was estimated at only 444 adults in 2005, and extensive effort has been invested in conservation activities, reproductive monitoring, and marking of individual birds, providing a comprehensive data set on population dynamics since 1998. We used these data to build a matrix projection model for two Piping Plover population segments that nest in eastern Canada in order to estimate both deterministic and stochastic rates of population growth (λ_d and λ_s, respectively).
Gratto-Trevor and Abbott 2011	North America	 As noted, most regions measure the effectiveness of their conservations actions by achievement of productivity goals (number of chicks fledged per pair). However, fledging rates are subject to several types of inaccuracies, including determining exactly what constitutes a "pair" (e.g., must it have a nest that has been found; whether renests are known and excluded from pair calculations or not). Accurately assessing the number of chicks that have fledged from each nest may also be a problem, depending on vegetation, changes in brood habitat suitability, and timing of brood visits versus fledging. For example, a recent study in Saskatchewan using individually marked chicks, showed that even with experienced personnel and in both large and small alkali wetlands, chick fledging rates were underestimated by over 20%, on average, based on observations of "unfledged" chicks in subsequent years	Regional counts from the International Piping Plover Breeding Census during 1991, 1996, 2001, and 2006.
Haig and Plissner 1993	North America	Change in status from previous censuses was difficult to determine. New populations were found in Montana, Colorado, and Saskatchewan, Canada; however, the distribution gap between Atlantic and Northern Great Plains/Prairie Piping Plover distribution grows as numbers decline in Minnesota; Manitoba, Canada; and the Great Lakes. TABLE 5. Changes in numbers of Piping Plovers at specific breeding areas. Atlantic Coast Newfoundland 1968 30 1984 4 7 −72 +75 Cadden Beach, Nova Scotia 1976 56 1983 28 20 −64 −29 Maine 1976 48 1982 12 38 −21 +217 Rhode Island 1945 80 1983 20 47 −41 +135 Connecticut 1980 40 1983 34 67 +68 +97 Long Island, New York 1939 1,000 1983 200 338 −66 +69 New Jersey 1980 118 1983 64 280 +137 +338 Delaware 1978 80 1984 18 10 −88 −44 Maryland 1972 85 1984 25 35 −59 +40 Great Lakes Michigan 1979 77 1982 14 39 −49 +179 Wisconsin 1900 140 1983 6 1 −99 −83 Northern Great Plains/Prairie Big Quill Lake, Saskatchewan 1978 210 1984 186 151 −28 −19 Chain Lakes, Alberta 1976 50 n.a. n.a. 9 −72 n.a. Lake Manitoba, Manitoba 1980 27 1984 9 3 −89 −67 Lake of the Woods, Minnesota 1982 44 1986 32 13 −70 −59 Niobrara River, Nebraska 1981 92 1985 100 110 +20 +10 [1] Sources are listed in Haig and Oring (1985). [2] Sources are listed in Table 2.	Over 1,000 biologists and volunteers from 10 nations collaborated in the 1991 International Piping Plover Census. Approximately 2,099 sites were censused yielding the highest number of breeding (5,482 adults) and wintering (3,45 1 birds) Piping Plovers ever recorded
Hecht and Melvin 2009a	Atlantic coast	Table 1. Number of nonfederal cooperators, total number of sites occupied by United States Atlantic Coast breeding piping plovers, number of sites with ≤5 pairs, and percent breeding pairs at sites with ≤5 pairs, 2002. No. nonfederal cooperators[a]: State agency / Local government / Private; No. occupied breeding sites: Total / With ≤5 pairs; % breeding pairs at sites with ≤5 pairs ME 2 2 7 20 17 71 NH 1 0 0 2 2 100 MA 2 10 6 109 84 28 RI 1 1 2 13 7 14 CT 1 0 1 8 7 77 NY 2 7 4 84 64 35 NJ 2 3 1 28 18 29 DE 3 0 0 2 2 100 MD 1 0 0 1 0 0 VA 2 0 2 7 1 1 NC 2 1 2 7 6 35 All states 19 24 25 281 208 30 [a] State and local government agencies, nongovernmental organizations, or private landowners that 1) paid full- or part-time staff to monitor or protect piping plovers, or 2) contributed funds, equipment, or supplies to on-site protection efforts.	We calculated expenditures and hours of staff and volunteer time dedicated to monitoring and managing the United States breeding population of Atlantic Coast piping plovers (*Charadrius melodus*) in 1993 and 2002 and considered implications for recovery of this management-dependent species. Using state reports to the USFWS, we tallied the total number of tabulated sites that supported breeding pairs in 2002, as well as the proportion of each state's population that nested at sites with M5 breeding pairs. We counted the number of cooperators, defined as governmental agencies, nongovernmental organizations, or private landowners that 1) paid full- or part-time staff to monitor or protect piping plovers, or 2) contributed funds, equipment, or supplies to on-site protection efforts in 2002.

Reference	Location	Observed Trend	Study Design
Hecht and Melvin 2009b	Atlantic coast	Census totals more than doubled, from 790 pairs in 1986 to 1,749 pairs in 2006, concomitant with sustained intensive management. Population growth was greatest and most rapid in the New England and New York-New Jersey recovery units, while increases were more modest in the Southern and Eastern Canada units. Periodic rapid declines in the Southern and Eastern Canada units raise concerns about long-term risks of extirpation. The Atlantic Coast population became less evenly distributed between 1989 and 2006, with the percentage of the population breeding in New England increasing from 21.5% to 36.2% while declining proportionately in the other three recovery units. Overall productivity for the Atlantic Coast population 1989-2006 was 1.35 chicks fledged per pair (annual range = 1.16-1.54), and overall productivity within recovery units decreased with decreasing latitude: Eastern Canada = 1.61, New England = 1.44, New York-New Jersey = 1.18, and Southern = 1.19. Within recovery units, annual productivity was variable and showed no sustained trends. There were significant, positive relationships between productivity and population growth in the subsequent year for each of the three U.S. recovery units, but not for Eastern Canada. There was a latitudinal trend in predictions of annual productivity needed to support stationary populations (λ = 1.0) within recovery units, increasing from 0.93 chicks fledged per pair in the Southern unit to 1.44 in Eastern Canada. **Table 1. Distribution, population growth (λ) 1989-2006, and productivity (chicks fledged per pair), by state and recovery unit, of breeding pairs of Atlantic Coast Piping Plovers.** [illegible] [illegible] Atlantic Coast Total  Figure 1. Trends in abundance (breeding pairs and productivity (chicks fledged per pair) for the Atlantic Coast population of Piping Plovers (U.S. East and Middle Atlantic Coasts, 1986–2006) and for the three U.S. recovery units, and for Eastern Canada, 1986–2006. Abundance (bars) and productivity (lines) are shown for each recovery unit, and for the Atlantic Coast population (Atlantic Coast Total). Dashed lines indicate attempts to discontinue management in the recovery unit. Arrows indicate the year in which management was discontinued in the recovery unit. Productivity data are not available for Eastern Canada in 1986–1988 and for the Southern Region in 1986–1989. Productivity data are not available for the Atlantic Coast Total in 1986–1989. Abundance data are not available for Eastern Canada in 1986–1988. Abundance data are not available for the Southern Region in 1986–1989. Productivity data are plotted for 1986–1987 for Eastern Canada, 1988–87 for New England, 1988–89 for New York-New Jersey, and 1990–91 for the Southern Region. Productivity data are not available for the Atlantic Coast Total in 1986–1989. Abundance data are plotted for 1986–1987 for Eastern Canada, 1988–87 for New England, 1988–89 for New York-New Jersey, and 1990–91 for the Southern Region. Productivity data are not available for the Atlantic Coast Total in 1986–1989. Abundance data are plotted for 1986–1987 for Eastern Canada, 1988–87 for New England, 1988–89 for New York-New Jersey, and 1990–91 for the Southern Region. Productivity data are not available for the Atlantic Coast Total in 1986–1989.	Breeding pairs in the U.S. and Eastern Canada were censused annually from 1986 through 2006, and productivity (chicks fledged per breeding pair) was reported annually for varying proportions of the population.
McGowan et al. 2011	Missouri River	 **Fig. 2.** The simulated percent decline in median abundance at 30 years, relative to the no-take scenario, under 0% (baseline), 2%, 4%, 6%, 8%, 10%, 12%, and 14% of eggs incidentally taken for piping plovers in the Great Plains, with low or high adult survival, and with or without density-dependent juvenile survival.	Using a two-stage population model (that annually split the population into three habitat types in order to isolate the habitat where take was permitted) for piping plovers in the Great Plains, we found that when the population is in rapid decline or when there is no density dependence, the probability of quasi-extinction increased linearly with increasing take. However, when the population is near stability and subject to density-dependent survival, there was no relationship between quasi-extinction probability and take rates. We note however, that a brief examination of piping plover demography and annual cycles suggests little room for compensatory capacity. We argue that a population's capacity for demographic compensation of incidental take should be evaluated when considering incidental allowances because compensation is the only mechanism whereby a population can absorb the negative effects of take without incurring a reduction in the probability of survival in the wild. With many endangered species there is probably little known about density dependence and compensatory capacity.
Plissner and Haig 2000b	General	Table 3 Results of Great Lakes/Great Plains/Prairie baseline model [illegible] Table 4 Results of Atlantic Coast piping plover metapopulation models [illegible]  Fig. 2. Probability of Great Lakes/Great Plains piping plover metapopulation persistence relative to mean number of offspring fledged per pair. Table 5 Persistence and size of Great Lakes/Great Plains populations at reproductive success (RS) = 1.7 and RS = 2.0 [illegible]  Fig. 3. Probability of Great Lakes/Great Plains piping plover metapopulation persistence as an effect of a 10% increase over baseline adult and/or juvenile mortality.	The metapopulation viability analysis package, vortex, was used to examine viability and recovery objectives for piping plovers.

Reference	Location	Observed Trend	Study Design
Haig et al. 2005	North America	Results indicated an 8.4% increase from 1991 but only a 0.2% increase since 1996. Regional trends suggest that since 1991, number of breeding birds increased on the Atlantic Coast by 78% (2,920 birds; 12.4% increase since 1996) and by 80% in the Great Lakes (72 birds; 50% increase since 1996). However, plovers declined 15% (2,953 birds; 10% decline since 1996) in Prairie Canada/U.S. northern Great Plains. Subregional trends since 1991 reflect a 32.4% decline in Prairie Canada (972 birds; 42.4% decline since 1996), a 2.5% decline in the U.S. northern Great Plains (1,981 birds; 24% increase since 1996), 5.5% decline in eastern Canada (481 birds; 14% increase since 1996), although a 66.2% increase on the U.S. Atlantic Coast (2,430 birds; 12% since 1996). While numbers were down in much of the U.S. northern Great Plains since 1996, an increase (460%, 1,048 birds; 67.7% increase since 1991) was detected on the Missouri River. 	Censuses were conducted during 2 periods, corresponding to mid-winter and mid-breeding seasons. The breeding census was conducted from 3 to 16 June 2001 for all regions except the U.S. Atlantic, which was completed from 26 May through 3 June. The breeding census recorded 5,945 adults at 777 of 1,892 sites surveyed.
Haig and Oring 1985	North America	  	Distributional data were gathered via direct censuses by the author and coordination of census data collected throughout North America between 1982 and 1984. Since there are no population estimates for the species prior to this paper, it is impossible to quantify a comprehensive increase or decrease in numbers of Piping Plovers.
Plissner and Haig 2000a	General	Baseline models indicate that while Atlantic Coast populations, under current management practices, are at little risk of near-term extinction, Great Plains and Great Lakes populations require 36% higher mean fecundity for a significant probability of persisting for the next 100 years. 	The metapopulation viability analysis package, vortex, was used to examine viability and recovery objectives for piping plovers. Reproductive success was operationally defined as the number of fledged young per pair and was derived for each population from annual productivity estimates (Atlantic Coast, 1991-1995, Table 1) and various local studies (Great Lakes and Great Plains, Table 2). Mean overall baseline productivity for the Atlantic metapopulation was 1.37 fledglings per pair and 1.25 fledglings per pair for Great Lakes and Great Plains piping plovers. Based on similarity of habitat types, environmental variance in reproductive success on the Great Lakes and Atlantic coast was based on a coefficient of variation of 0.4, used for earlier Atlantic Coast models. Slightly higher variance, CV of 0.5 was used for the Great Plains populations, where habitats are more variable.
Roche et al. 2010c	General	  Figure 2. Annual adult apparent survival (Φ_{AHY}) of piping plover populations in North America (1998–2008) using Atlantic Coast nonbreeding habitat based on real parameter estimates generated by model [Φ {[BQL,PC,LD,MR]^t},{[AC,GL,NY]^t}, p(reduced)] is represented by thick lines. Vertical dashed lines represent the associated 95% confidence intervals. Dots represent raw survival estimates generated from the fully temporal model [Φ (breeding^t), p(top)]. Populations using Atlantic Coast nonbreeding habitat were Atlantic Canada (AC), Great Lakes (GL), and Long Island, New York (NY). Populations using Gulf Coast nonbreeding habitat were Big Quill Lake, Prairie Coteau, Lake Diefenbaker, and Missouri River. Figure 3. Annual adult apparent survival (Φ_{AHY}) of piping plover populations in North America (1998–2008) using Gulf Coast nonbreeding habitat based on real parameter estimates generated by model [Φ {[BQL,PC,LD,MR]^t},{[AC,GL,NY]^t}, p(reduced)] is represented by thick lines. Vertical dashed lines represent the associated 95% confidence intervals. Dots represent raw survival estimates generated from the fully temporal model [Φ (breeding^t), p(reduced)]. Populations using Atlantic Coast nonbreeding habitat were Atlantic Canada, Great Lakes, and Long Island, New York. Populations using Gulf Coast nonbreeding habitat were Big Quill Lake (BQL), Prairie Coteau (PC), Lake Diefenbaker (LD), Missouri River (MR).	We used long-term (1998–2008) mark–recapture data on piping plovers collected from 7 separate studies located throughout North America to conduct a range-wide analysis of after hatch year apparent survival (ϕ_{AHY}). Our objectives were to compare concurrent survival estimates from disparate breeding sites and determine whether estimates followed similar trends or were correlated among breeding populations with shared wintering grounds. Our analysis included resighting data from 2,040 AHY (after hatch year) piping plovers, including 451 from the Missouri River, 249 from Big Quill Lake, 216 from the Prairie Coteau, 333 from Lake Diefenbaker, 286 from the Great Lakes, 435 from Atlantic Canada, and 70 from New York.
Root et al. 1992	North Dakota	Since then the Great Plains population has continued to decline, from an estimated 1500 pairs in 1985 to < 1100 pairs in 1990 (Great Lakes & Northern Great Plains Piping Plover Recovery Team, unpub. data).	We estimated survival of Piping Plovers in 1984-1990 at the Chain-of-Lakes wetlands in the John E. Williams Memorial Nature Preserve (JWMNP), McLean County, North Dakota. Mean population size was 90.4 pairs (range = 46-152) during our study (Mayer 1990), making the site one of the largest concentrations of Piping plovers in the Great Plains.
Ryan et al. 1993	North Dakota	 Figure 1. Comparison of count data and simulated population size of the Great Plains Piping Plover. Solid lines represent count data (corrected from Haig 1992); upper line = maximum estimate, lower line = minimum estimate. Population simulations are represented by open circles = [immature survival = 0.7 (adult survival)]; closed circles = [immature survival = 0.8 (adult survival)]; open triangles = [immature survival = 0.9 (adult survival)]; closed squares = [immature survival = adult survival]. See Table 1 for survival and reproductive rate data. **Table 3. The effect of population growth rate and management delays on Piping Plover population growth to recovery levels.** Management initiation year: Immediate; Immediate; Delay 1 year; Delay 5 years Projected population size[a]: 1500; 1500; 1386; 1010 Population growth rate: 2%; 1%; 1%; 1% Years to recovery (95% confidence limit): 30 (26–34); 51 (37–61); 64 (47–94); 118 (88–144) [a] Based on simulations under recent demographic conditions (with immature survival = 0.60), see Table 1 for data.	

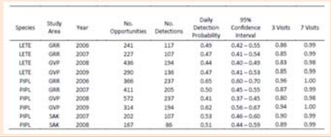
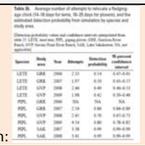
Reference	Location	Observed Trend	Study Design
Aron 2005	Upper Missouri River		USCOE Monitoring Program
Cohen et al. 2006	East coast US	The population at WHD and WHB in 2001, 2002, 2003, and 2004 included 53, 54, 40, and 28 pairs, respectively. The data do not suggest that survival or site fidelity are a consistent response to population size. 	Our objectives were to estimate adult survival and site fidelity of a declining Piping Plover population on Long Island, New York, to determine if survival or site fidelity changed over time as the population declined, and to document dispersal within and among breeding seasons.
Cohen and Gratto-Trevor 2011	Great plains Canada	We estimated that a productivity of 0.75 chicks fledged/pair would be required to maintain a stationary population in our study area, given our model-averaged estimates for adult and subadult survival averaged across the entire study period. In stochastic simulations using a CV of 0.09 for the survival rates, the productivity needed for a stationary population increased to 0.86 (95% confidence bounds 0.29–1.82) and, in simulations using a CV of 0.2 for survival rates, the necessary productivity increased to 1.16 (95% confidence bounds 0.09 to 4.62). Mean productivity from 2002 to 2009 at Big Quill Lake, Lake Diefenbaker, Lake Chaplin, and the Coteau was 0.87, 0.96, 0.88, and 0.88, respectively.	 Table 5. Productivity (fledglings/nesting pair) of Piping Plovers in various study areas in Saskatchewan, 2002–2009. Site / Year / No. birds / No. nests / Fledglings/pair / Source Lake Chaplin / 2002 / 71 / 36 / 0.7 / Wiltse 2005a 2003 / 148 / 41 / 0.7 / Wiltse 2005a 2004 / 144 / 75 / 0.9 / Wiltse 2005a 2005 / 200 / 123 / 1.2 / Martens and Goossen 2008 Lake Diefenbaker / 2002 / 257 / 138 / 1.0[a] / J. P. Goossen and S. M. Wiseworth, unpubl. data 2003 / 164 / 95 / 1.4 / J. P. Goossen and S. M. Wiseworth, unpubl. data 2004 / 185 / 97 / 1.3 / J. P. Goossen and S. M. Wiseworth, unpubl. data 2005 / 251 / 115 / 0.9[a] / Wiltse 2005b 2006 / 162 / 84 / 0.9 / Wiltse 2009 2007 / 134 / 83 / 0.5 / Wiltse 2009 2008 / 154 / 89 / 0.2 / Wiltse 2009 2009 / 88 / 73 / 1.9 / Wiltse 2009 Missouri Coteau[b] / 2002 / 148 / 73 / 0.7 / This study 2003 / 148 / 66 / 0.8 / This study 2004 / 152 / 57 / 0.8 / This study 2005 / 150 / 54 / 0.9 / This study 2006 / 140 / 56 / 1.2 / This study 2007 / - / - / - / - 2008 / - / - / 0.4 / S. Wiseworth, pers. comm. Big Quill Lake[c] / 2002 / 64 / 30 / 1.1 / This study 2003 / 88 / 33 / 1.8 / This study 2004 / 104 / 43 / 0.8 / This study 2005 / 128 / 40 / 1.2 / This study 2006 / 118 / 59 / 0.2 / This study 2007 / - / - / 0.3 / This study 2008 / - / - / 0.7 / This study 2009 / - / - / 0[a] / This study [a]Includes captive-reared fledged chicks. [b]Intensive study areas only. [c]Fledging rate assumed to be zero because no breeding habitat remained around flooded lake. We used mark-resight data from 2002 to 2009.
Maxson and Haws 2000	Minnesota		
Sherfy et al. 2008	Missouri River	This work is focused on developing standard protocols for monitoring biological changes in response to ESH management actions. The monitoring program is intended to provide a basis for understanding positive and negative outcomes of management actions, inform future decisions about implementation of ESH projects, and provide for adaptive management through feedback to subsequent management decisions.	
Sherfy et al. 2009a	Missouri River		
Sherfy et al. 2009b	Missouri River		
McGowan et al. 2014	Great Plains	We simulated extinction probabilities of piping plovers in the Great Plains and estimated the relationship between extinction probability and various demographic parameters. We tested the fit of regression models linking initial abundance, productivity, or population growth rate to extinction risk, and then, using the regression parameter estimates, determined the conditions required to reduce extinction probability to some pre-defined acceptable threshold. An initial abundance of approximately 2400 females with an expected mean population growth rate of 1.0 will limit extinction risk for piping plovers in the Great Plains to less than 0.048.	

Genetics

Reference	Location	Findings
Miller et al. 2010	General	Analyses included individuals from 23 U.S. states and Canadian provinces and were based on mitochondrial DNA sequences (580 base pairs, n = 245) and 8 nuclear microsatellite loci (n = 229). Our findings provide support for separate Atlantic and Interior subspecies (*C. m. melodus* and *C. m. circumcinctus*, respectively). Birds from the Great Lakes region were allied with the Interior subspecies and should be referred to as *C. m. circumcinctus*. Population genetic analyses illustrated stronger genetic structure among Atlantic than among Interior birds, which may reflect reduced natal- and breeding-site fidelity of Interior individuals. Furthermore, analyses suggested that Interior birds previously experienced genetic bottlenecks, whereas there was no evidence of such patterns in the Atlantic subspecies.
Haig et al. 2011	General	Funk et al. (2007b) measured adequate gene flow among Great Basin and Pacific Coast Snowy Plovers (*Charadrius alexandrinus*), but banding information suggested quite minimal movement between these inland and coastal areas. As a result, a distinct population segments was defined under the ESA on the basis of demographic isolation rather than genetics. A "subspecies" is generally defined as a breeding population that has measurably distinguishable genotypes or phenotypes (or both) and occupies a distinct geographic area within its species range.

Funk, W. C., T. D. Mullins, and S. M. Haig. 2007b. Conservation genetics of Snowy Plovers (*Charadrius alexandrinus*) in the Western Hemisphere: Population genetic structure and delineation of subspecies. Conservation Genetics 8:1287-1309.

Detection Probabilities

Reference	Location	Findings
Shaffer et al. 2013	Upper Missouri River	Nest Detection:  Chick detection: 
Brudney et al. 2013	Great Lakes	Each brood (n = 126) was treated as a sample unit, and 29 daily encounter occasions were included to estimate daily survival (ϕi) and detection probabilities (pi) from hatch until 6 days past our definition of fledging. Model-averaged daily detection probabilities ranged from 0.709 to 0.962, depending on age and year, with an overall mean (± SE) of 0.908 ± 0.016.
Roche et al. 2014	Upper Missouri River	We individually color-marked, recaptured, and re-sighted 1,635 interior least tern chicks and 1,318 piping plover chicks from 2006 to 2009 at 4 study areas along the Missouri River and investigated effects of observer-, subject-, and site-level covariates suspected of influencing detection. Increasing the time spent searching and crew size increased the probability of detecting both species regardless of study area and detection methods were not associated with decreased survival. However, associations between detection probability and the investigated covariates were highly variable by study area and species combinations, indicating that a universal mark-recapture design may not be appropriate. We conducted mark-recapture of piping plovers at SAK in 2006–2009, at 37 sandbars at GRR in 2007, and at 31 sandbars at GVP in 2008–2009. Mark-recapture of least terns occurred at 37 sandbars at GRR in 2006–2007, at 31 sandbars at GVP in 2006–2009, and at 4 sandbars at LCL in 2007–2008. Following chick banding, we revisited each sandbar or shoreline site every 2–3 days and resighted or recaptured chicks. Searches were conducted systematically in a grid formation. The location of vegetation on a sandbar (or lakeshore) along with the shape of the sandbar itself, can have serious consequences on the ease at which resighting or recapture occurs, with large sandbars punctuated by several discrete vegetated areas offering some of the most logistically difficult conditions, and narrow unvegetated relatively small sandbars some of the easiest. Given that 1) increasing search times is probably more cost effective than adding additional team members to a crew, and 2) increasing search times increased the probability of detection for both species at all study areas, the most efficient means by which to increase overall daily detection probability would be to increase the length of site visits.

Not Used

Reference	Location	Findings
Abbott 1916	North Dakota	Piping plovers observed in Nebraska "potholes" in 1916.
Wiltermuth et al. 2009	Lake Sakakawea	Observed eight occurrences of piping plovers moving their nest and eggs during incubation.

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Reproductive Success

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Appendix B

Literature Review Table for Terns

Appendix B

Literature Review Table for Terns

Tabulated results from the literature review for terns are presented by the following categories:

- Reproductive Success
- Prey
- Nest Site Selection
- Nest Density
- Habitat Quality and Quantity
- Predation
- Stage Survival
- Site Fidelity, Immigration, and Emigration
- Overwinter Habitat
- Population Trends through Time
- Genetics
- Detection Probabilities.

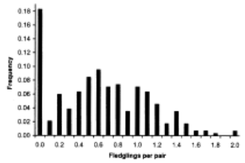
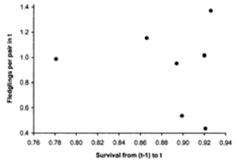
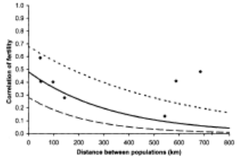
For convenience, the images inserted in the category tables have been placed in an accompanying appendix (Appendix G) so that they may be viewed in full size.

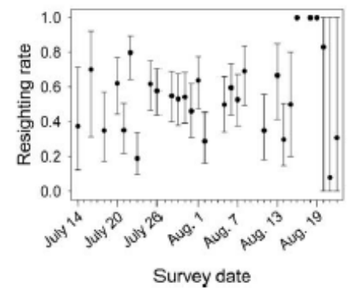
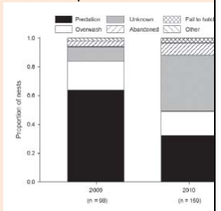
References are listed as “not used” or by the category in which they appeared in the table.

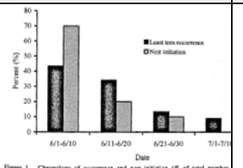
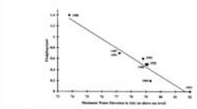
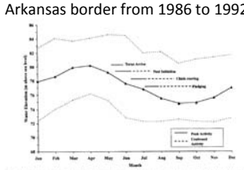
Tables

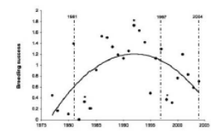
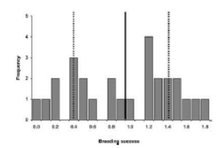
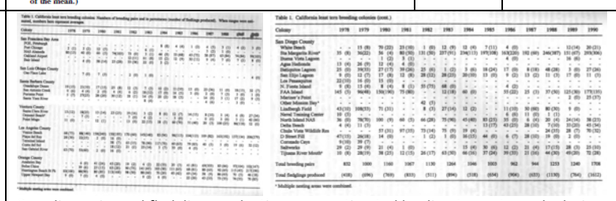
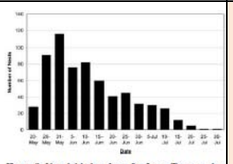
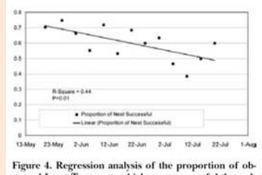
Information within the light orange cells was used in the parameter table.

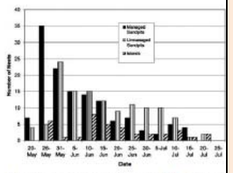
Reproductive Success

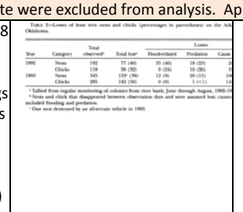
Citation	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
Akcakaya et al. 2003	California	Massey et al. (1992; Table 2) reported the age that 186 banded adults were first known to breed: age 2 for 58 adults, age 3 for 75 adults, and age >4 for the remaining 53 adults. Based on these numbers, we calculated P2 = 31.2% (58/186), P3 = 71.5% ([58 + 75]/186), and P4+= 100.0%.			 Fig. 3. Frequency distribution of fledglings per pair for all populations of California least tern in California, USA, and Baja California, Mexico, from 1971 to 1998 combined. Each x-value is the upper limit of the class represented by the vertical bar for that value; thus, all data in the "0.0" class is exactly zero.  Fig. 4. Number of fledglings per pair of California least tern in year *t*, in the North San Diego population (California, USA), as a function of the adult survival rate from year *t* – 1 to *t* in Camp Pendleton (which is part of this population) for years 1988–1994. The coefficient of correlation is –0.12 (SE = 0.44).  Fig. 7. Correlation in the number of fledglings per pair between populations of California least tern. The points show the observed correlation among several populations with complete data sets. The curves show the low (dashed), medium (solid), and high (dotted) correlation-distance functions used in our model.	We examined a metapopulation model for the California least tern that can be used to predict persistence of populations along the Pacific coast and the effects of various management actions. We used the model to estimate the effect of reducing predation impact, an important source of reduced fecundity, in various populations. Apart from restricting human access to nesting sites, most management efforts have concentrated on predation. In the model, each cluster of nearby colonies is defined as a population. Within each population, the model includes age-structure, year-to-year changes in survival and fecundity, regional catastrophes (strong El Nifio/Southern Oscillation [ENSO] events), and local catastrophes (reproductive failure due to predation).			
Aron 2005	Upper Missouri River		58.7%; 55.1%; 34.2%; and 52.7%	488 (11%) lost to Predation 200 (4%) abandoned 757 (16%) unknown	0.93; 0.65; 0.81; and 0.88	4645 nests; 11055 eggs	1986-2004	USCOE Monitoring Program	Lake Oahe; Fort Randall River; Lewis and Clark Res.; and Gavins Point River
Bailey and Servello 2008	Northeast				Fledgling residence times decreased relative to fledging date at all colonies, and the trend for maximum values of FRT, which may be an indicator of the maximum time to departure, varied from approximately 30 d in mid-July to 15 d in early to mid-August. The highest one-d counts of fledglings underestimated fledgling production by 21-30% for the two sites with high chick survival. Methods based on multiple counts also tended to underestimate fledgling production. Fledgling residency patterns limit the utility of count methods for estimating fledgling production and fledging success.	The objectives of the present study were to measure chick survival and fledgling residence time (FRT) at three Least Tern colonies in Maine and examine the accuracy of single or multiple counts of fledglings for estimating fledgling production and fledging success. Chicks were color-banded at age zero to one d in 2002-2003 to estimate chick survival using mark-resighting methods and to examine relationships between fledging date and FRT. Colony sites were searched for banded chicks and fledglings every one to three d from just prior to fledging until their disappearance from the vicinity of nesting areas at three colonies. Data on numbers of banded fledglings known alive on each survey date were used to simulate single or periodic counts of banded fledglings for the purpose of comparing estimates of fledgling production and success based on count methods with known values. **Table 1. Chick survival estimates for three Least Tern breeding colonies in Maine, 2002-2003.** Colony: Crescent Surf-Lautholm Beach; Higgins Beach; Crescent Surf Beach Number of chicks banded: 96; 62; 53 Proportion of banded chicks resighted as fledglings[1]: 0.74; 0.71; 0.15 Chick survival (Φ) based on mark-resighting analyses — Daily (Φ ± SE): 0.9864 ± 0.0069; 0.9855 ± 0.0042; 0.9098 ± 0.0148 Chick survival (Φ) based on mark-resighting analyses — 21-d chick period: 0.75; 0.74; 0.14 [1]Chicks were classified as fledglings when age was ≥21 d.			

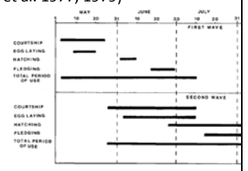
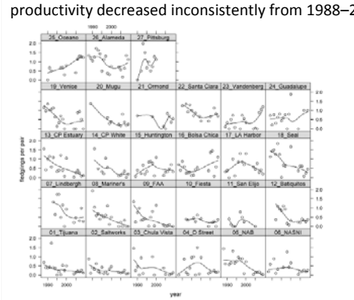
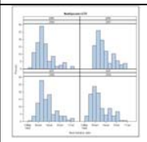
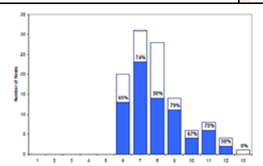
Citation	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
					 Figure 1. Resighting probabilities (p and 95% CI) for banded Least Tern fledglings for 30 survey dates (29 resighting intervals) at Crescent Surf-Laudholm Beach in 2002.				
Brooks et al. 2013	South coast		Measures of nest success (n = 257 nests) ranged from 0-93% among colony sites. The DSR of nests was primarily related to colony site, but year and estimates of predation risk also were related to DSR.	Predation was the principal cause of identifiable nest loss, accounting for 47% of nest failures when the two years of data were pooled.  Figure 2. Cause of nest loss for Least Terns (*antillarum*) in Cape Romain National Wildlife South Carolina, 2009-2010.		Daily survival rate (DSR) of nests and chicks was examined at four natural nesting sites in Cape Romain National Wildlife Refuge, South Carolina, 2009-2010.			
Brunton 1997	Northeast		Breeding success dropped from 0.56 chicks per nesting attempt during 1987 to 0.08 chicks per nesting attempt during 1988, primarily due to increased predation. Abandonment levels were proportionally equal for center and edge. Abandonment decreased slightly during 1988 when Black-crowned Night-Heron predation was most intense. Pledging success was significantly lower in the center compared to the edge during both years (ML x2, = 45.8, P < 0.001, Table 1). Hatching success did not differ between center and edge nests across years (ML x2, = 3.0, P > 0.08, Table 1). TABLE 1. Hatching and fledging success, and causes of nest failure within regions of the colony during 1987 and 1988. 1987 Edge / 1987 Center / 1987 Total / 1988 Edge / 1988 Center / 1988 Total Hatching success # Eggs hatched/nest $\bar{x}$ 1.39 1.28 1.31 0.77 0.80 0.79 SD 0.08 0.07 0.08 0.08 0.07 0.07 Fledging success # Chicks hatched/nest $\bar{x}$ 0.94 0.41 0.57 0.12 0.01 0.06 SD 0.08 0.09 0.08 0.07 0.09 0.08 95% CI* 0.85–0.95 0.37–0.45 0.55–0.57 0.09–0.17 0.00–0.02 0.04–0.08 (Total # chicks) (136–152) (142–174) (298–310) (21–42) (1–9) (24–49) Causes of failure Herons 1 46 47 75 204 279 (%) (32.9) (73.8) Crows 7 0 7 39 2 41 (%) (4.9) (10.8) Human 2 10 12 0 0 0 (%) (8.4) (0.0) Tides 7 0 7 1 0 1 (%) (4.9) (0.3) Abandoned 24 22 46 11 16 27 (%) (32.2) (7.1) Unknown 5 19 24 13 17 30 (%) (16.8) (7.9) Total failed nests 46 97 143 139 239 378 (% of each region) (28.6) (25.3) (26.2) (57.2) (65.8) (62.4) Number of nests 161 384 545 243 363 606 * Confidence limits for proportions (Fisher and Yates 1963).	606 nests	1987-1988	A permanent, alphanumeric grid was established in the Sandy Point Least Tern colony, West Haven, Connecticut and consisted of 10 X 10 m quadrats, with a central marker (brick with the grid coordinates). This grid made it possible to record the presence and location of all nests.	Edge and center of colony

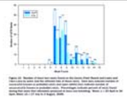
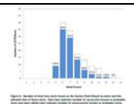
Citation	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
Conway et al. 2003	Texas	 Figure 1. Chronology of occurrence and nest initiation (% of total number (category) for interior least terns nesting in the Prairie Dog Town Fork of the R in Childress County, Texas, 1998.	Table 1. Nesting parameters of interior least terns nesting in the Prairie Dog Tow the Red River in Childress County, Texas, 1998. Nesting parameters — Least tern nest Total number of nests (n) — 20 Successful nests (n) — 13 Abandoned nests (n) — 2 Nests with unknown status (n) — 5 Raw nest success (%) — 65 Mayfield estimate (%) — 70.7 Clutch ($\bar{x}$) — 2.25 Range — 1—3 Nests located on gravel substrate (n) — 5 Nests located on sand substrate (n) — 15 Nest dimensions (north to south) (cm) ($\bar{x}$) — 9.7 Nest dimensions (east to west) (cm) ($\bar{x}$) — 10.2 Nest depth (from nest rim to nest bowl bottom) (cm) ($\bar{x}$) — 1.9			13 successful and 7 uncessessful nests	1998	Approximately 4.5 km of the Prairie Dog Town Fork of the Red River in Childress County, Texas, were surveyed for nesting terns on June 2, 3, 4, 9; 10, 11, 17, 22, 29, 30 and July 2 and 8, 1998. All adults were counted on each visit. Nest searches were executed by locating adult terns performing distraction displays and by intensively searching suitable non-vegetated habitats for nests.	Habitat of successful vs. unsuccessful nests.
Dugger et al. 2002	Mississippi River				High spring flows (April and May) were not associated with increased tern reproductive success (Table II), but the number of FLP decreased with increased mean and maximum July water levels (Table II and Figure 3). None of the tern reproductive parameters were significantly associated with the hydrologic variables we developed to reflect flood timing or duration (Table III). Table I. Estimates of least tern reproduction on the LMR from colonies adjacent to Missouri during 1986–1993 (1986–1989; data from Smith and Renken, 1993) Year — No. pairs — No. fledglings — Chicks/pair — Fledglings/pair 1986 — 275 — 141 — 1.01 — 0.5 1987 — 298 — 210 — 1.70 — 0.7 1988 — 277 — 392 — 1.48 — 1.4 1989 — 499 — 96 — 0.73 — 0.2 1990 — 706 — 334 — 0.73 — 0.5 1991 — 1135 — 836 — 1.07 — 0.7 1992 — 587 — 342 — 0.61 — 0.6 1993 — ** — ** — ** — 0.0 Mean (S.E.) — 540 (117.9) — 336 (93.2) — 1.09 (0.17) — 0.58 (0.13)  Figure 3. The relationship between July maximum river elevation from the LMR at Cardwelville, MO and estimates of the number of fledglings/pair for tern colonies between River 1125 and 1334 during 1986–1993.	Our objective was to determine whether least tern reproductive success was directly associated with annual hydrology on the LMR. We monitored least tern nest colonies along the LMR adjacent to Missouri at river km (Rkm) 1521–1334 between Cairo, IL and the Missouri–Arkansas border from 1986 to 1992.  Figure 1. The relationship between tern reproductive chronology and the long-term annual hydrograph (1987–1993) (see Ellis and Gray, 1996) with monthly mean, minimum and maximum river elevations for the LMR at Cardwelville, MO. River 1361.				

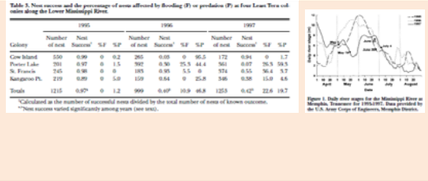
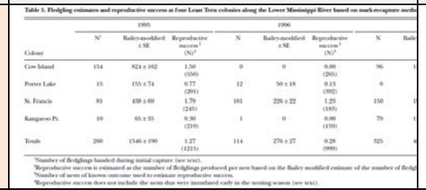
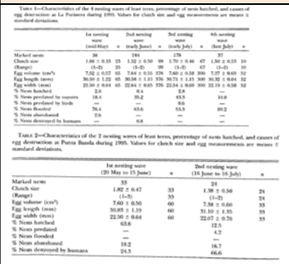
Citation	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
Dugger et al. 2000	Mississippi River				Based on survival rate estimates and number of known pairs producing chicks we estimated 0.72 and 1.0 fledglings/pair were produced at Rkm 1431 and Rkm 1481, respectively. TABLE 3. Fledging success of Interior Least Terns reported as per breeding pair (FL/PR). Site / FL/PR / Range (n)[a] Missouri River (Montana) 0.7 — Fort Peck Reservoir 1.3 — Yellowstone Reservoir 0.2 — North Dakota 0.6 0.4–0.9 (4) Missouri River[b] (South Dakota) 0.4 0.2–0.7 (6) Missouri River[c] (South Dakota) 0.4 0.1–0.6 (6) Cimarron River (Kansas) 0.5 0.0–1.4 (23) Lower Mississippi River (Missouri) 0.7 0.2–1.4 (7) Lower Mississippi River (Missouri) 0.85 0.7–1.0 (2) [a] Refers to number of years or sites in average. [b] Below Gavins Point Dam. [c] Below Fort Randall.	Our purpose was to estimate chick survival from hatching to fledging, for Least Terns nesting at two sites on the Lower Mississippi River in Missouri, using mark-recapture methodology. We banded 110 Least Tern chicks during 1995 on sand island nesting colonies situated at river kilometers (Rkm) 1431 and 1481 on the Lower Mississippi River.			
Elliott et al. 2007	California	Table 1. Mean nest initiation dates Point Least Tern colony from 2000-20 Year / Mean nest initiation date ± 2000 5 June ± 13 2001 22 May ± 10 2002 24 May ± 10 2003 28 May ± 15 2004 26 May ± 15	Table 2. Clutch sizes and egg fates for the Alameda Point Least Tern colony from 2000-2004. Year: 2000 / 2001 / 2002 / 2003 / 2004 (A) Clutch size (% nests) 1-egg 55 (17.6) / 17 (6.2) / 40 (12) / 68 (18.5) / 148 (33.6) 2-eggs 251 (80.4) / 247 (89.8) / 267 (82) / 287 (78.0) / 281 (63.9) 3-eggs 6 (2.0) / 11 (4.0) / 18 (5) / 13 (3.5) / 10 (2.3) 4-eggs 0 / 0 / 0 / 0 / 1 (0.2) 5-eggs 0 / 0 / 1 (1) / 0 / 0 Mean ± SD (N) 1.84 ± 0.41 (312) / 1.98 ± 0.32 (275) / 1.93 ± 0.42 (325)[a] / 1.85 ± 0.45 (368) / 1.69 ± 0.52 (440) (B) Egg fates (% eggs) hatched 360 (62.6) / 489 (89.9) / 435 (68.7) / 516 (75.8) / 554 (74.6) failed to hatch 105 (18.3) / 28 (5.1) / 56 (8.8) / 75 (11.0) / 63 (8.5) abandoned 19 (3.3) / 7 (1.3) / 50 (4.7) / 63 (9.2) / 78 (10.5) depredated 11 (1.9) / 1 (0.2) / 44 (7.0) / 0 / 1 (0.1) died while hatching 1 (0.2) / 0 / 0 / 0 / 1 (0.1) unknown 79 (13.7) / 19 (3.5) / 64 (10.1) / 25 (3.7) / 46 (6.2) unknown if attended[b] 0 / 0 / 4 (0.6) / 0 / 0 incidental take 0 / 0 / 0 / 2 (0.3) / 0 TOTAL 575 / 544 / 633 / 681 / 743 Mean hatching success ± SD (N) 0.67 ± 0.39 (284) / 0.89 ± 0.26 (275) / 0.67 ± 0.42 (326) / 0.70 ± 0.45 (368) / 0.70 ± 0.45 (440) [a]Results do not include the 5-egg clutch. [b]Undetermined if nest was attended during 21 day incubation period; egg did not hatch. clutch size = 1.9 (S.E. = 0.05)		 Figure 3. Breeding success (fledglings / breeding pair) of the Alameda Point Least Tern colony from 1976-2004. (* = El Niño years. Years of important site changes are noted: 1981 = electric fence was erected; 1997 = Naval Air Station closure; 2004 = chain link fence erected.)  Figure 4. Frequency distribution of breeding success (fledglings/breeding pair) at the Alameda Point Least Tern colony from 1976-2004. (Note: the solid line represents the mean, and the dashed lines represent ± 1 SD of the mean.)		Results on breeding chronology, clutch size, egg fates, and hatching success were only compared for years 2000-2004. When possible, mean clutch size and mean hatching success estimates from previous years were taken from CDFG annual monitoring reports and compared to the 2000-2004 results.	
Fancher 1992	California				 Breeding pairs and fledgling production were estimated by direct counts made during several colony visits per year. However, these methods have varied between census takers, sites, and years. However, since the late 1970's, relatively standardized techniques and comprehensive coverage have produced more reliable estimates of breeding pairs. Estimates of fledgling production are likely to have been less accurate than the breeding pair estimates due to complications of timing and frequency of visits by the census takers.				
Jenniges and Plettner 2008	Platte River	 Figure 2. Nest initiation dates for Least Terns at sites monitored along the central Platte River, Nebraska 1991 to 2005.	During the study period a total of 647 Least Tern nests were observed; 473 at managed sandpits, 125 at unmanaged sandpits, and 49 at constructed river islands. Hatching success for those nests was 38% on unmanaged sandpits, 65% on managed sandpits and 71% on constructed riverine islands.  Figure 4. Regression analysis of the proportion of observed Least Tern nests which were successful through the nesting at human created habitat along the central Platte River, Nebraska.		A total of 639 Least Tern chicks were observed to have fledged from these nests. Production of fledged chicks per nest for each type of nesting site were unmanaged sandpits 0.56 fledglings/nest, managed sandpits 1.13 chicks/nest and islands 1.04 chicks/nest. Overall, 647 nests produced an estimated 639 fledglings for a ratio of 0.99 fledglings per nest. Using the definition that pairs are equal to the maximum number of active nests (incubating adults) and broods (adults with chicks) present there were 553 pairs total for a fledgling per pair ratio of 1.15. Nest success and fledgling survival was significantly greater at managed sandpits than unmanaged sandpits in the 1994 to 1997 time period. Table 4. Least Tern reproductive output at nesting sites monitored. Successful nests are those th one egg, successful broods are those broods where at least one chick fledged. Management type / Total nests / Successful nests / Estimated successful broods / Number of initial chicks observed / Number of chicks fledged Unmanaged sandpits 125 48 32 101 54 Managed sandpits 473 309 254 679 534 Managed river islands 49 35 28 74 51 Total 647 392 314 854 639	Monitoring of nests occurred at three constructed riverine islands in the central Platte River and at seven sandpits along the central Platte River Management activities at the managed sandpits and created islands included vegetation control measures to maintain the bare sand nesting substrate, people management and predator management. Predator control measures included electric fences at all sites with nesting birds. Of the nests observed 125 were on unmanaged sandpits, 473 were on managed sandpits and 49 were on islands.	Managed and unmanaged sandpits

Citation	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
		 Figure 5. Nest initiation dates for Least Tern nests as managed sandpits, unmanaged sandpits and created islands 1994-1997.	Table 3. Nest success for Least Terns nesting at sites monitored 1991-2005. Success shown as ap the Mayfield estimate with 95% confidence intervals calculated. Management type / Number of nests / Apparent nest success / Mayfield nest success / Mayfield lower 95% C.I. Unmanaged sandpits 125 38% 36% 29% Managed sandpits 473 65% 64% 59% Managed riverine islands 49 71% 69% 56% Total 647 63% 61% 57%		Table 5. Fledge ratios for Least Terns at sites monitored along the central Platte River. Results pre per nest and fledglings per pair. A pair is defined as the maximum number of nest and broods pre gle visit. Management type / Number nests / Number of pairs / Fledglings per nest / Fle Unmanaged sandpits 125 97 0.43 Managed sandpits 473 409 1.15 Managed river islands 49 47 1.04 Total 647 553 0.99				
Kirsch 1996	Platte River	The distribution of nest initiation dates and rates of colony-site turnover were similar on both habitats.	Productivity did not differ between habitats but varied significantly among sites. Nest success, fledging success, and fledglings per pair averaged 0.54, 0.28, and 0.47, respectively. Key factor analysis revealed that chick survival had a greater influence on production of fledglings (on both sandbars and sandpits) than did failure to produce a maximum clutch size or egg mortality. Table 13. Nest and fledging success, and fledglings per pair for least terns on sandbars and sandpits along the lower Platte River, Nebraska, 1987–90. Year / No. of study colonies: Sandbars, Sandpits / Nest success: Sandbars f, SE; Sandpits f, SE / Fledging success: Sandbars f, SE; Sandpits f, SE / Fledglings per pair: Sandbars f, SE; Sandpits f, SE 1987 3 5 0.49 0.25 0.46 0.18 0.14 0.04 0.37 0.21 0.21 0.12 0.28 0.08 1988 4 3 0.72 0.14 0.72 0.12 0.27 0.02 0.24 0.13 0.73 0.15 0.55 0.23 1989 6 6 0.41 0.08 0.54 0.06 0.55 0.10 0.38 0.11 0.57 0.17 0.50 0.06 1990 6 7 0.44 0.05 0.64 0.12 0.12 0.08 0.33 0.05 0.25 0.16 0.64 0.14			I examined habitat selection and productivity of least terns to determine if terns selected habitat according to suitability (as indicated by productivity), what factors affected habitat selection and productivity, and if estimated productivity could support this population. Natural sandbar habitat and sandpit sites created by gravel dredging adjacent to the river were compared with census data from 1987-1990.	
Kirsch and Sidle 1999	Central US				Fledging success ranged from 0.00 to 2.33 fledglings/pair, and was <0.51 in 9 areas. Fledging success estimates were available for interior least terns in 22 local areas for -1 year. Fledging success was extremely variable, and least terns failed to produce fledglings in >1 year in several of the areas. Mean fledging success for 9 local areas was less than the 0.51 (0.06-0.49) fledglings/pair needed for population maintenance. Fledging success was marginally higher than 0.51 (0.52-0.54) in 3 areas and exceeded 1 (1.22-1.60) fledgling/pair in 5 areas. Based on available fledging success estimates, there is no evidence that productivity within the interior range caused recent increases in tern numbers.	We compiled data from published and unpublished surveys of interior least tem abundance and productivity for 1984-1995. Census information included dates, numbers of adults and nests observed, locations of nesting areas, and census method. Productivity information included numbers of pairs and nests monitored, frequency of visits to colonies, fledglings per pair or nest, estimation methods, and definition used for "fledged".		
Koenen et al. 1996	Oklahoma		Least Terns (P < 0.001) selected nest sites on ridges more than expected in the fenced exclosure with ridges from 1991 to 1994. During this study, nest success for Least Terns on ridges ranged from 0.23 to 1.00 (Table 1). There were no differences (P > 0.05) in nest success between Least Terns nesting on or off ridges from 1991 to 1994. Least Tern nests in the fenced exclosure had higher nest success (P < 0.05) in 1991 than nests outside of fenced exclosure. From 1992 to 1994, there were no significant differences in nest success inside and outside of the exclosures.			See Table 1	1991-1994	We evaluated two management methods to increase nesting success of interior LeastTerns at Salt Plains National Wildlife Refuge, OK from 1991-1994. Nest ridges were constructed from existing substrate and designed to provide elevated habitat safe from sheet flooding. Solar-powered electric fences were built to reduce predation by mammals. Least Tern nests on and off ridges were located inside and outside fenced areas and revisited every 3-4 d to quantify nest success using the Mayfield Method and determine causes of nest failure.	

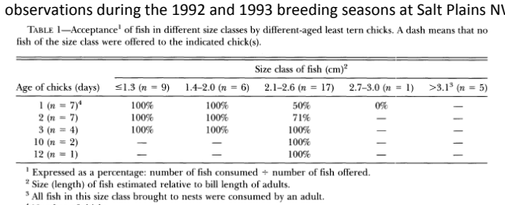
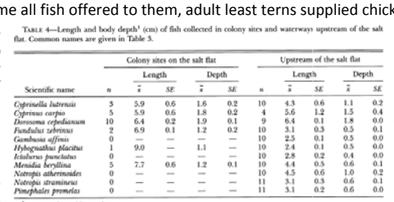
Citation	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
Kruse et al. 2001	Upper Missouri River		TABLE 1. Productivity summary for Piping Plovers and Least Terns nesting on the Gavins Point and Fort Randall river reaches of the Missouri River, South Dakota, 1991 and 1992. Piping Plover / Least Tern Total Nests: 238 / 354 Total Destroyed: 128 (53.8%) / 179 (50.6%) Predation: 61 (25.6%) / 80 (22.6%) Chicks Hatched: 368 / 330 Chicks Fledged: 57 (15.5%) / 81 (24.5%) Apparent nesting success of caged nests (n = 86, 62%) was higher ($P < 0.001$) than for control nests (n = 122, 34%). Of 33 caged nests that were unsuccessful, 6 (18%) were inundated, 16 (48%) were lost to predation, 2 (6%) were abandoned for unknown reasons, and 9 (27%) were lost due to human disturbance, sandbar erosion, weather, or unknown causes. 44 least tern nest with unknown fate were excluded from analysis. Apparent nest success was 49.4%, fledge rate was 0.32.			May-August 1991 and 1992	Nest were visited every 5-7 days	
Leslie et al. 2000	Oklahoma		Least terns produced 192 nests, 118 chicks, and 80 fledglings in 1992 at 12 colonies compared with 345 nests, 285 chicks, and 143 fledglings in 1993 at 20 colonies. Loss of nests to flooding was greater in 1992 (46% of 77 nests lost) than 1993 (9% of 133 nests lost). Overall loss of chicks was greater in 1993 (50%) than 1992 (32%).		Estimates of fledglings per breeding pair from the four census methods were 0.50-0.66 in 1992 and 0.48-0.70 in 1993. TABLE 1—Numbers of adult breeding pairs and fledglings per breeding pair of least terns of successful nests, median-date method, probable renests subtracted from the total number airboat surveys (see methods for description) on the Arkansas River, Oklahoma. Year / Category / Census methods: Successful nests / Median date / Probable renests 1992 Adult pairs: 140[a] / 166 / 159 Fledglings/breeding pair: 0.59 / 0.50 / 0.52 1993 Adult pairs: 212[a] / 299 / 296 Fledglings/breeding pair: 0.67 / 0.48 / 0.48 [a] Considered a minimal estimate of adult pairs because it did not account for unsuccessf		We monitored breeding activities of least terns to determine effects of hydropower and flood-control operations of Keystone Dam from 1992-1993. Flows in 1993 scoured and elevated existing sandbars and created new sandbars, resulting in an expansion of nesting habitat that was associated with an increase from 12 breeding colonies in 1992 to 20 colonies in 1993. Estimates of adult least terns increased from 125-166 breeding pairs in 1992 to 212-299 in 1993.	
Lombard et al. 2010	South coast		Overall daily nest survival (±SE) excluding managed colonies was 0.92 ± 0.03 (period survival = 0.18). Daily nest survival of managed colonies (fenced) was significantly higher (0.97 ± 0.02; period survival = 0.51). Nest success, the probability that at least one egg hatched per nest, was positively and significantly influenced by colony size. As reported by Birkhead (1997), nest success was highest in large colonies. TABLE 1. Daily and period nest survival for Least Terns at St. Croix, U.S. Virgin Islands, 2003–2006. Period nest survival is the daily nest survival extrapolated to 21 days. Estimates summarized by habitat (substrate), year, and colony size. Managed colonies = exclosure or platform. Small = 10–30, medium = 40–90, large ≥110 nests. Standard errors were adjusted for design effect. Category / Daily nest survival / Period nest survival Habitat Salt flats: 0.910 (0.017) / 0.138 Sandy beach: 0.936 (0.036) / 0.252 Offshore cay: 0.929 (0.005) / 0.215 Industrial: 0.929 (0.016) / 0.195 Managed: 0.968 (0.023) / 0.511 Year 2003: 0.902 (0.025) / 0.115 2004: 0.934 (0.025) / 0.242 2005: 0.923 (0.012) / 0.188 2006: 0.906 (0.023) / 0.127 Colony size Small: 0.876 (0.041) / 0.062 Medium: 0.883 (0.023) / 0.074 Large: 0.927 (0.001) / 0.204	The period survival of chicks was 0.30 ± 0.11.	From estimates of nest and chick survival we estimated that 269 fledglings were produced during this study, equivalent to 0.06 chicks fledged per nesting attempt (n = 4640 nests) and 0.15 chicks per breeding pair (n = 1744 pairs).		We report nest and chick survival and colony-site dynamics of the Least Tern. These results are the first for the Caribbean and were derived with likelihood-based approaches from 4640 nests and 44 chicks fitted with transmitters monitored in 52 colonies at St. Croix, U.S. Virgin Islands, 2003–2006. All nesting sites were surveyed every four days from mid-April to mid-August.	
Massey and Atwood 1981	California	Usually egg laying begins the first or second week in May after the terns have been in the area for 1-4 weeks, and the first wave of laying is complete in 10-14 days.	All of the eggs in the first nesting wave at Anaheim Bay hatched successfully, but only 82% (44/54) in the second wave hatched. Four nests were apparently abandoned after laying, as no bird was ever seen incubating the eggs. Two others were deserted during the incubation period. One was abandoned early; the other, a renest		The minimum number of fledglings per maximum number of breeding pairs is ascertained for each colony whenever possible. If a colony has produced 1.0-1.5 fledglings/nesting pair, success is considered good,0 .5-1.0/pair is a moderate success, and 0-0.5/pair is poor.	A large-scale, long-term program of banding Least Tern chicks in southern California was begun in 1976 and has continued	In 1980, a group of late-nesting birds, many of them banded and color-banded, presented a unique opportunity for study. Most of the banded birds were identified as 2-yr-old terns nesting for the first time; several were older, renesting pairs that had failed earlier at a colony close by. The chronology of the second wave, age composition, clutch size, behavior of 2-yr-old parents, fledging success, and postfledging dispersion were documented.

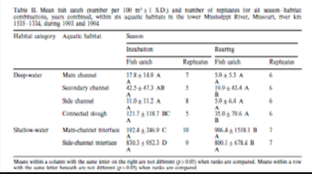
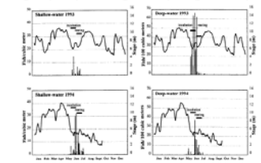
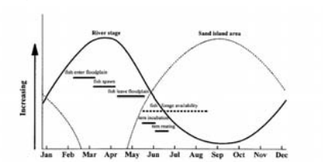
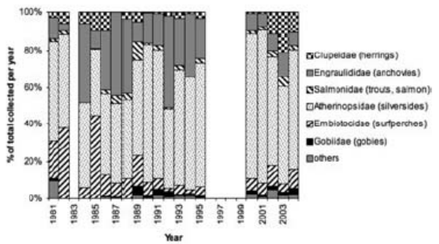
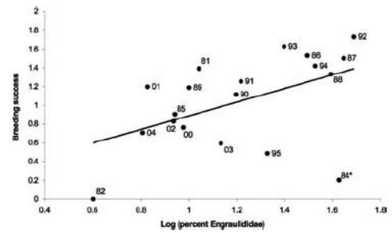
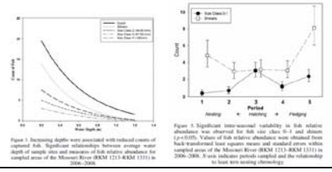
Citation	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
		By mid-June, when most of the chicks have hatched, a second and usually smaller group of courting and nesting terns is often observed. The second period of egg laying is more prolonged than the first, and hatching may continue throughout July and occasionally into August (Atwood et al. 1977, 1979)  Fig. 1. Chronology of the California Least Tern nesting colony in Anaheim B TABLE 2. Egg-laying period—second nesting wave, Anaheim Bay 1980. Date of laying of first egg June 1 4 8 17 18 20 24 28 1 4 Total number of nests (n = 29)* 1 2 1 2 2 1 2 2 1 4 2 1 Nests of 2-yr-old terns (n = 10)* 1 1 1 1 1 1 1 2 Nests of terns 3+ yr of age (n = 2)* 1 * The total number of nests in the second wave was 33; laying dates of four abandoned nests could not be determ * Twelve 2-yr-old terns nested in the second wave at 10 nests; in two instances both pair members were banded (n * Three birds 4+ yr of age nested in the second wave at two nests; both pair members were banded and were 4+	containing the last eggs laid in the colony, was abandoned after 25 days, the eggs still unstarred. Two eggs failed to hatch at nests where the other egg hatched: one of them was infertile; the other did not survive pipping TABLE 3. Clutch sizes of first- and second-wave nests at several California Least Tern colonies in 1980. Number of eggs Colony 1 2 3 $\bar{x}$ ± SE First wave Venice Beach (30/150)* 3 (8%) 31 (86%) 2 (6%) 1.97 ± 0.063 Huntington Beach (29/65) 3 (10%) 25 (86%) 1 (3%) 1.93 ± 0.069 Anaheim Bay (6/10) 1 (17%) 2 (33%) 3 (50%) 2.33 ± 0.333 Second wave Anaheim Bay (33/33) 11 (33%) 21 (64%) 1 (3%) 1.70 ± 0.067 Anaheim Bay—nests of 2-yr-old birds (10/10) 3 (30%) 7 (70%) 0 1.70 ± 0.153 * Sample size = 31 nests, colony size = 150 nests. The $t \times 3$ table of clutch distributions was subjected to a χ^2 test and revealed a significant difference in clutch size between the first and second waves of nesting ($P < 0.05$)	The number of chicks fledged from the initial nesting group at Anaheim Bay in 1980 could be estimated only roughly; approximately 5-10 youngsters probably fledged from 8-10 nests, with success characterized as moderate. Fledging success could be gauged with greater accuracy for the second wave. All chicks were banded with red anodized service bands and were distinguishable from first wave chicks. We were able to document a minimum of 19 fledglings and possibly as many as 35. Thus, the second wave at Anaheim Bay showed moderate success with a minimum of 0.58 fledglings/pair. The 10 nests in the second wave where at least one parent was a 2-yr-old bird produced a minimum of 8 fledglings. The ratio was 0.8, considered a moderate success. TABLE 4. Fledging success at several California Least Tern colonies of varying sizes i Colony / Number of nesting pairs / Minimum number of fledglings / Fledglings/nesting pairs* Venice Beach 150–165 240 1.45 Huntington Beach 70–90 85 0.94 Los Cerritos 7 6 0.86 Bolsa Chica 20–26 15 0.58 Anaheim Bay (total) 38–43 24 0.56 First wave 8–10 5 0.50 Second wave 33 19 0.58 Second wave (nests of 2-yr-old birds) 10 8 0.80 * Minimum number of fledglings/maximum number of nesting pairs.	each year since, with 259 chicks banded in 1976, 297 in 1977, 407 in 1978, and 612 in 1979. Since 1978, chicks at major colonies in Los Angeles and Orange counties have been color-banded with a year and colony code, so that many returning birds are now identifiable in the field as to age and natal colony. In 1980 we began to color-band adults individually as they were trapped. TABLE 1. Ages of banded, breeding California Least Terns. Age (yr) 1 2 >2 Unknown Birds nesting at Anaheim Bay in 1980 (n = 23) First nesting wave 0 1 2 2 Second nesting wave 0 12 (63%) 7 (36%) 2 (10%) Birds nesting in Los Angeles and Orange Counties, 1976–1980 (n = 11) First nesting wave 0 1 (9%) 7 (64%) 0 Second nesting wave 0 0 (0%) 3* (27%) 0 * All nesting after initial failure.			
Schuetz 2011	California				Fledgling productivity was positively associated with clutch size ($p < 0.1$). At the mean clutch size (1.79 eggs), California least terns fledged 0.50 young. At two standard deviations above and below the mean (1.38 and 2.20 eggs), terns fledged 0.39 and 0.65 young. Fledging productivity decreased inconsistently from 1988–2009.  Figure 3. Plots of fledging probability by year conditional on site. Sites are ordered by latitude with Tijuana Estuary being the southernmost site and Pittsburg Power Plant being the northernmost. Dots indicate estimates of fledglings per year in a single season and lines are smoothers used to reveal patterns in the data. doi:10.1371/journal.pone.0019489.g003		1988-2009	I assembled data on California least tern population size, clutch size, and fledgling production at 27 breeding sites throughout California using California Department of Fish and Game (CDFG) breeding season surveys (1988-2009).
Shaffer et al. 2013	Upper Missouri River		% Nest Success Species / Study Area / Year / USGS Estimate / USGS 95% Confidence Interval / COE Reported Apparent / COE Reported Mayfield LETE GRR 2006 65 54 – 73 77 76 LETE GRR 2007 78 68 – 85 89 87 LETE GVP 2008 66 59 – 73 75 69 LETE GVP 2009 73 64 – 80 79 76 PIPL GRR 2006 21 16 – 27 54 46 PIPL GRR 2007 47 39 – 54 73 65 PIPL GVP 2006 42 35 – 48 67 53 PIPL GVP 2009 41 34 – 49 54 46 PIPL SAK 2007 25 17 – 33 41 12 PIPL SAK 2008 28 19 – 37 44 16	Tables 9 – 21 (pages 130 - 134)	Minimum Species / Study Area / Year / Reported by COE / Low Renesting: USGS Expanded Estimate / USGS 95% Confidence Interval / Relative Bias (%) / High Renesting: USGS Expanded Estimate / USGS 95% Confidence Interval / Relative Bias (%) LETE GRR 2006 0.81 0.78 0.51–1.17 4 0.90 0.59 – 1.35 -10 LETE GRR 2007 1.06 0.80 0.46–1.32 33 0.89 0.52 – 1.48 18 LETE GVP 2008 1.14 1.23 0.98–1.50 -7 1.37 1.09 – 1.68 -17 LETE GVP 2009 1 1.43 1.16 – 1.75 -30 1.61 1.30 – 1.97 -38 PIPL GRR 2006 NA NA PIPL GRR 2007 0.97 0.96 0.46 – 1.77 1 1.12 0.54 – 2.06 -14 PIPL GVP 2008 1.37 0.89 0.76 – 1.05 54 1.03 0.88 – 1.21 33 PIPL GVP 2009 1.09 0.60 0.57 – 0.63 83 0.68 0.65 – 0.72 60 PIPL SAK 2007 0.7 0.34 0.04 – 1.11 107 0.42 0.06 – 1.37 68 PIPL SAK 2008 0.68 0.27 0.05 – 0.83 155 0.32 0.06 – 0.98 116	Gavins Point, Garrison Reach, and Lake Sakakawea	2006-2008	Stratafied Random Habitat searched every 2-3 days from mid-April-early Aug	USCOE % of population over time
Sherfy et al. 2008	Upper Missouri River	 Figure 16. Number of least tern nests found on the Garrison Reach by week and the ultimate fate of those nests. Dark bars indicate number of successful (known or probable) nests and open (white) bars indicate number of unsuccessful (known or probable) nests. Percentages indicate percent of nests found during that week that ultimately produced at least 1 hatchling. Week 1 is 29 April - 5 May, Week 13 = 22 July - 28 July, 2007.	Nests With One Radiomarked Adult # Nests % Successful 37 52 Probable successful 23 32 Probable failure 1 1 Failure 10 14 Unknown 1 1 Total 72		Gavins Point, Garrison Reach, and Lewis and Clark Lake	2007	Stratified sample	High or low use based on USACE data

Citation	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
Sherfy et al. 2009a	Upper Missouri River		Table 12. Numbers and fates of least tern and piping plover nests on created and natural sandbars on the Gavins Point Reach (GVP) and Lewis and Clark Lake (L-CL) in 2008. Species: Least Tern; Piping Plover; Total Study Area: GVP; LCL Sandbar Type: Mechanically Created; Natural; Mechanically Created; Both Total Nests: 155; 41; 140; 237; 537 Successful: 80; 15; 58; 94; 247 Nest Fate – Probable Successful: 25; 5; 38; 13; 81 Nest Fate – Probable Failure: 0; 0; 0; 0; 0 Failure: 38; 19; 31; 108; 196 Unknown: 12; 2; 13; 22; 49			Gavins Point and Lewis and Clark Lake	2008	Stratified sample	High or low use based on USACE data
Sherfy et al. 2009b	Upper Missouri River		Table 7. Numbers and fates of least tern and piping plover nests on the Gavins Point Reach in 2009. Total Nests: 136; 194; 330 Successful: 91; 72; 163 Nest Fate – Probable Successful: 6; 16; 22 Nest Fate – Probable Failure: 1; 0; 1 Failure: 33; 98; 131 Unknown: 5; 8; 13			Gavins Point	2009	Stratified sample	High or low use based on USACE data
Sidle and Harrison 1990	Central US		The interior least tern's annual reproductive success varies greatly along a given river or shoreline (Table 6). Because tern's use ephemeral habitats, they are susceptible to frequent nest and chick loss. Consequently there are great local differences in productivity. Table 6. Some examples of the productivity of interior least terns. Locations: Missouri River North Dakota; Missouri River South Dakota; Missouri River South Dakota; Lower Platte River Nebraska; Cimarron River Kansas; Salt Plains NWR, Oklahoma Year: 1988, 1989; 1986, 1987; 1988, 1989; 1987, 1988, 1989; 1982-83; 1987 Nest Success: 0.62, 0.56; 0.36, 0.51; 0.57, 0.67, 0.43; 0.18; 0.44-0.33 Fledglings per Pair: 0.42, 0.21; 0.20, 0.64; 0.44, 0.55; 0.29, 0.71, 0.47; 1.09-0.56; 0.44-0.15 Frequency of Visits: 7-10 days; 7-10 days; 7-10 days; 2-3 days; --; 1-3 days % Population Monitored: 100%; 100%; 100%; 39%, 44%, 42%; -- Source: Mayer and Dryer 1989; Schwalbach 1988; Dirks 1990; Kirsch 1987-89; Schulenberg and Ptacek 1984; Hill 1987 1921, Hardy 1957, Stiles 1939). Thompson (1982) presented the following longevity data for coastal least terns revealed by band recoveries: Percentage of Recoveries Age (years): 0-5; 5-10; 10-15; 15-20; >20 Known and Assumed Dead (N): 74 percent (58); 9 percent (7); 10 percent (8); 4 percent (3); 3 percent			1985-1988	This plan summarizes available biological data, details various actions to stabilize and/or restore the interior least tern, and establishes criteria to remove it from the federal list of endangered species.
Smith and Renken 1993	Mississippi River		Average clutch size among years ranged from 2.09 to 2.48 eggs. Nest success, the probability a nest will hatch at least one egg, averaged 65% during 1986-1988; nest success in 1989 (51%) was significantly lower than previous years because of a flood during the peak of nesting. Average chick production was 1.5 chicks hatched/pair during 1986-1988, but only 0.7 chicks hatched/pair in 1989.	Untimely floods had the greatest impact upon Least Tern reproduction. We observed that rising water levels destroyed 41% of 92 existing nests on 2 July and 13 July 1986, and 40% of 436 nests on 19 June 1989. Coyote predation on eggs was documented in 1987, 30 of 31 nests on one island were destroyed. Nest predation by coyotes also was verified at six other colonies during 1988 and 1989. All-terrain vehicle disturbance of tern nests and young also hampered successful tern reproduction. In 1988, ATVs passed through or near 25 nests, 18 of which failed. We also observed two instances in which chicks or adults were killed by ATVs. In 1988, ATV users killed two chicks that had recently hatched in a nest.	Average indices of fledging success were 0.86 fledglings/pair during 1986-1988 and 0.2 fledglings/pair in 1989. If we assumed that production of 0.5-1.0 fledglings/pair was necessary to maintain tern populations, our estimates of production appeared to be adequate to maintain population stability. Table 1. Estimates of reproductive parameters for interior Least Terns nesting in the Mississippi River valley during 1986 - 1989. Reproductive parameter: Nest survival rate (s.e.); Egg survival rate (s.e.); Clutch size (s.e.); Chick production[a] (s.e.); N[b]; Fledging success[a]; Total number of nests per year[d] Year 1986: 0.63 A[c] (0.029); 0.94 AB (0.025); 2.39 A (0.036); 1.01 A (0.102); 165; 0.5; 381 1987: 0.68 A (0.033); 0.98 A (0.011); 2.57 A (0.038); 1.70 B (0.095); 189; 0.7; 397 1988: 0.64 A (0.033); 0.88 B (0.024); 2.48 B (0.036); 1.68 B (0.079); 257; 1.4; 455 1989: 0.51 B (0.023); 0.74 C (0.038); 2.09 C (0.026); 0.73 C (0.056); 375; 0.2; 912 [a]Estimates with the same letter are not significantly different at P=0.05, using LSD tests (SAS1988). [b]No. of chicks hatched per pair. Yearly comparisons made using Chi-square tests. [c]No. of known-fate nests. [d]No. of fledglings per pair. [e]Includes all nesting attempts.	Reproductive parameters of interior Least Terns were estimated during the 1986-1989 breeding seasons at Mississippi River colonies in Missouri, Illinois, Kentucky, and Tennessee. Population indices ranged from 275-499 breeding pairs during the 1986-1989 breeding seasons.			
Stucker et al. 2013	Upper Missouri River		Influence of habitat features on nest fate.—We determined the fates of 95% of nests. Nest success was greater on constructed sandbars (70%, n = 675) than on natural sandbars (37%, n = 149). Nest failure was ~25% more likely in the presence of any wet substrate in the surrounding area. TABLE 2. Counts of Least Tern nests and surrounding-area habitat assessments on natural and constructed sandbars on the Missouri River, by year and point type (nest or surrounding area). Natural: Nest, Surrounding area; Constructed: Nest, Surrounding area; Nest total 2006: 64, 64, 208, 208, 272 2007: 52, 52, 212, 212, 264 2008: 40, 40, 293, 286, 333 Total: 156, 156, 713, 706, 869 Constructed, nest; Natural, nest - wet substrate; Natural, nest - dry substrate; Probability of nest success; Percent pebble FIG. 2. Increasing percentage of pebble at Least Tern nesting sites was associated with increased nest success on constructed and natural sandbars on the Missouri River (Gavins Point Reach and Lewis and Clark Lake), 2006–2008. Substrate moisture differences in surrounding area (3 m from nest) for natural sandbar sites are indicated by dashed (dry) and dotted (wet) lines.		We documented habitat characteristics and fate for 869 Least Tern nests during 2006–2008; 71% were discovered before clutch completion. Eighteen percent of nests were located on natural sandbar habitats and 82% on constructed sandbar habitats. We categorized nest fate as successful (at least 1 chick), unsuccessful, or unknown.			
Szell and Woodrey 2003	Mississippi River	Table 2. Clutch initiation dates at four Least Tern colonies along the Lower Mississippi River. Dates are Julian date. Colony: Cow Island; Porter Lake; St. Francis; Kangaroo Pt.; Totals 1995 Range (N)[a]: 176-201 (146); 178-202 (108); 177-205 (255); 175-206 (240); 175-206 (1298) 1995 Median: 181[b]; 185[b]; 187; 179[b]; 182[b] 1995 Peak[c]: 176-185; 178-189; 177-195; 175-184; 175-188 1996 Range (N)[a]: 181-198 (270); 182-199 (521); 183-205 (196); 177-199 (161); 177-205 (1090) 1996 Median: 184[b]; 187[b]; 187; 182[b]; 187 1996 Peak[c]: 181-196; 182-191; 185-192; 177-195; 181-192 [a]Number of nests for which clutch initiation could be determined (includes nests of known-fate eggs). [b]Peak clutch initiation period represents the first 75% of active clutches laid (see text). [b]Includes 95 clutches laid 29 May 1997 which became inundated 2 June 1997. These clutches were included in the calculation of peak initiation. [b]Includes 136 clutches laid 30 May 1997 which became inundated 2 Jun 1997. These clutches were included in the calculation of peak initiation. [b]**Median clutch initiation date varied significantly among colonies within a given year (see text). [c]Median clutch initiation date varied significantly among years (see text).	Both nest success, the proportion of nests from which at minimum one egg hatched, and hatching success, the proportion of eggs that hatched, were significantly higher in 1995 (97% and 94% respectively) as compared to 1996 and 1997 (~40%). Predation and untimely flooding were the largest negative impacts upon successful tern reproduction.		Reproductive success averaged between 0.28-1.27 fledglings/clutch.				The reproductive ecology of the Least Tern was examined at four nesting colonies along the Lower Mississippi River during the 1995-1997 breeding seasons. Nest success, hatching success, and reproductive success were calculated in order to estimate productivity and evaluate reproductive status. Nesting colonies varied in size from 172 to 550 nests, with average clutch sizes ranging from 1.9-2.7 eggs/nest.

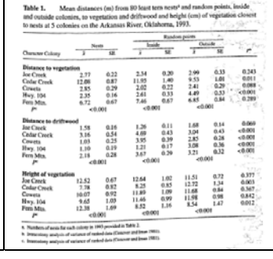
Citation	Location	Nest Initiation	Nest Success	Nest Fate	#fledge/nest or min # adult pairs (1.1 for stable pop)	Sample Size	Years	Design	Strata
									
Zuria and Mellink 2002	Northwestern Mexico	The colony flooded almost entirely during the highest tides, forcing the terns tore-nest several times. We counted 4 nesting waves: mid-May, early June, early July, and late July. The first was composed of early breeders. The second corresponded to the peak laying period and may have had newcomers as well as re-nesters. The third and fourth waves probably were composed primarily of re-nesters, because the colony flooded almost entirely in mid-June and mid-July.						We studied two colonies, La Purinera and Punta Banda, during the breeding season of 1995 (April to September). We located nests by finding incubating adults from a distance and by walking through the colony early in the mornings.	
Dirks 1990	Missouri River	Least terns arrived in the Gavins Point area on May 14, 1988 and nests were initiated from May 22 to July 30, 90% of which were initiated between May 28 and July 14. In 1989 least tern nests were initiated from May 17 to July 17 with 90% of nest initiations occurring between May 26 and July 5. Least terns averaged 1.8 nest initiations per pair in 1988 and 1989.	Least tern hatching success increased from 36% in 1988 to 51% in 1989.		Sixty-seven least terns fledged in 1989 compared to 20, 93 and 65 in 1986, 1987, and 1988, respectively. The number of least tern chicks fledged per pair increased slightly from 0.44 in 1988 to 0.55 in 1989.			Surveys were conducted in 1988 and 1989 to determine the distribution and reproductive success of least terns and piping plovers on the Missouri and Cheyenne rivers in South Dakota. All suitable habitat was surveyed, with research concentrated on two important areas: 1) the Missouri River from Fort Randall Dam to Springfield, SD, and 2) the Missouri River from Gavins Point Dam to Ponca State Park, NE. All colony sites were revisited every 7 to 10 days to determine nest fates and to search for new nests. Colony sites with successful nests were monitored to determine chick survival and fledging success.	

Prey

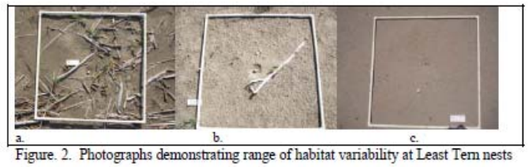
Citation	Location	Findings
Carreker 1985	North America	This model was developed with information obtained from the published literature and communications with professional biologists familiar with the species and its habitat requirements. This model does not directly measure the availability of fish. As an alternative, it is assumed that a suitable nesting habitat in close proximity to abundant and/or diverse aquatic habitat is a desirable nesting site for least terns. The stomach contents of 49 1least terns co11 ected in New Jersey, Florida, and Louisiana consisted of 43.1% silver anchovy (Engraulis eurystole), 6.8% menhaden (Brevoortia tyrannus), 6.3% mummichogs (Fundulus heteroclitus), 5.0% Crustacea, 1.1% silversides (Menidia spp.),' and 37.7% unidentified items (McAtee and Beal 1912). In California, northern anchovy (Engraulis mordax) usually was the most commonly dropped fish followed by topsmelt (Atherinops affinis), jacksmelt (Atherinopsis californiensis), and deepbody or slough anchovies (Anchoa compressa,~. delicatissima) (Atwood and Kelly 1984). In Mississippi, Hays (1980) found Gulf menhaden (Brevoortia patronus) and bay anchovy (Anchoa mitchilli) most frequently dropped at a least tern colony. In addition to these two species, emerald sleeper (Erotelis smaragdus) and rough silverside (Membras martinica) were dropped at colony sites in Texas (B. C. Thompson, Texas Parks and Wildlife Department, Austin; letter dated 9 August 1984). Burroughs (1966) found sand lance (Ammodytes spp.), herring (Clupea spp.), and hake (Urophycis spp.) dropped in least tern colonies in Massachusetts. Hardy (1957) found dropped river shiner (Notropis blennius) on the lower Ohio River and determined this species to be the dominant food of least terns in that area. Schulenberg et al. (1980) collected plains killifish (Fundulus kansae) most often at colony sites in Kansas.
Schweitzer and Leslie 1996	Oklahoma	We examined foraging patterns and success of the least tern nesting on a salt flat in north-central Oklahoma during the 1992 and 1993 breeding seasons. Our objectives were to 1) estimate whether there were limited quantities of forage available to breeding least terns, 2) examine least tern use of bodies of water adjacent to the salt flat, and 3) identify fish species brought to colony sites and in bodies of water adjacent to the salt flat. Mean feeding rate was 2.0 ± 0.3 fish/h, and overall time between feedings was 24.5 ± 2.2 min. Although not statistically different, fish were brought to nests containing two chicks more frequently than to nests with one chick (2.3 + 1.0 fish/h vs. 1.9 + 0.2 fish/h; P = 0.51). Mean estimated length of fish brought to nests was 2.9 ± 0.4 cm. Young chicks (<2 days old) were offered and consumed a greater percentage of small fish than older chicks (>3 days old). Fish in the greatest size class (>3.1 cm long) were only offered to, and consumed by, adult least terns. Our observations during the 1992 and 1993 breeding seasons at Salt Plains NWR suggested that because chicks did not consume all fish offered to them, adult least terns supplied chicks with enough small (<4 cm) fish at suitable intervals (2 fish/h) to meet their energy requirements.  
Stucker 2012	Upper Missouri	As a plunge-diving piscivore, Least Terns also rely on ready access to appropriately sized slender-bodied fish: <52 mm total length for adults and <34 mm total length for young chicks. This study evaluates 1) the shallow water (<1.5 m) fish community near river and mechanically created

Citation	Location	Findings
	River	emergent sandbars during three nesting seasons (2006-2008), 2) nest-scale habitat selection to determine if nest habitats differ between constructed and natural sandbars, and it evaluates the consequences of this selection on nest success, and 3) the potential associations between specific habitat features, at differing spatial scales, associated with airborne and foraging Least Terns. Effort focused on the 95 kilometers of Gavins Point Reach of the Missouri River, between Yankton, SD and Ponca, NE during the Least Tern nesting season in 2006-2008. We sampled fish within 15-16 areas every two weeks from late May - July to document the relative abundance, species richness, and size classes of fish. Using systematic surveys on sandbars every 2-3 days, we detected and tracked 869 Least Tern nests until eggs hatched or failed, on constructed and natural sandbars. Successful foraging sites (N=416) were compared to a paired nearby random location to evaluate evidence of habitat selection. Fish relative abundance was negatively related to depth. Catches were dominated by schooling species. Significant differences in the fish communities between the sandbar types were not detected. See Stucker et al. (2012) below for figures.
Tibbs and Galat 1998	Mississippi River	Spatial and temporal availability of small fishes, as forage for interior least terns was compared to the least tern reproductive period during 1993 and 1994 in the lower Mississippi River, Missouri. Timing of forage availability and least tern reproduction was also related to river stage and temperature regimes. Four deep-water habitats (main channel, secondary channel, side channel, and connected slough) and two shallow-water habitats (main-channel interface and side-channel interface) were sampled. A total of 67245 fish 510 cm TL were collected in the six aquatic habitats, 34882 during 1993 and 32363 during 1994. Mean ranked fish catch differed between seasons (p≤0.005) and among habitats (p≤0.0001). Catch-per-volume and richness were highest in shallow-water habitats in both years. Highest catches in deep- and shallow-water habitats occurred between 56 and 64 days after peak spring flows in both years. Water temperature was increasing rapidly during the spring flood peak each year.  Table III. Mean taxa richness of collected fish (±1 S.D.) and number of replicates within six aquatic habitats, years, and seasons combined, in the lower Mississippi River, Missouri, river km 1535–1334, during 1993 and 1994 Habitat category; Aquatic habitat; Total richness; Replicates Deep-water; Main channel; 4.6 ± 2.0 A; 13 Secondary channel; 4.4 ± 3.1 AB; 11 Side channel; 3.9 ± 1.9 A; 14 Connected slough; 2.6 ± 1.4 B; 11 Shallow-water; Main-channel interface; 6.8 ± 2.9 C; 17 Side-channel interface; 8.3 ± 3.2 C; 16 Means within a column with the same letter on the right are not different ($p < 0.05$) when ranks are compared.  
Elliott et al. 2007	California	 Figure 5. Least Tern dropped prey collected at Alameda Point from 1981-1982, 1984-1995, and 2000-2004. (Years 1981-1995 from Collins (1995).)  Figure 6. Least Tern breeding success and the percentage of Engraulididae in the dropped prey. (Each point is labeled with the two-digit year. * = possible outlier.)
Stucker et al. 2012	Upper Missouri River	Least terns rely on ready access to appropriately sized slender-bodied fish: <52mm total length for adults and <34mm total length for young chicks. We studied the shallow-water (<1.5 m) fish community near river and mechanically created emergent sandbars during three nesting seasons (2006–2008). We sampled every 2 weeks each year from late May to July within 15–16 areas to document the relative abundance, species richness and size classes of fish. Fish relative abundance was negatively related to depth. Catches were dominated by schooling species, including emerald shiner, sand shiner, spotfin shiner and bigmouth buffalo. Significant inter-annual differences in relative abundance were observed, with generally increasing trends in intra-seasonal relative abundance of shiners and the smallest size classes of fish (<34 mm). Significant differences in the fish communities between the sandbar types were not detected in this study. Results suggest that mechanical sandbar habitats host comparable fish communities at similar levels of relative abundance. Table V. Measures of fish species and relative abundance during fish sampling in areas of shallow-water habitat near emergent sandbars of the Missouri River (RKM 1213–RKM 1331), 2006–2008 Mechanically created; River created; $F_{1,16}$; p Species; 3.3 (0.5); 3.4 (0.3); 0.03; 0.86 Count of fish[a]; 2.1 (0.3) [7.0]; 2.2 (0.2) [8.5]; 0.31; 0.59 Count of shiners[a]; 1.5 (0.3) [3.7]; 1.7 (0.2) [4.6]; 0.32; 0.58 Count of size class 0–1; 0.8 (0.2) [1.2]; 0.9 (0.1) [1.6]; 0.67; 0.43 Count of size class 2[a]; 1.1 (0.2) [1.9]; 1.1 (0.1) [1.9]; 0; 0.95 Count of size class 3[a]; 0.7 (0.2) [1.0]; 0.8 (0.1) [1.2]; 0.19; 0.67 Count of size class 4[a]; 1.2 (0.3) [2.3]; 1.4 (0.2) [3.2]; 0.59; 0.45 Results from least squares means (standard errors) [back-transformed means] are presented by emergent sandbar type, with associated F-test and level of significance (p). [a]Least square means presented are the means at the average of the included covariate, depth, 0.61 m. Table II. Fish size class, fish body length (TL) and relationship with least tern bill length Size Class; Fish length (TL, mm); Relationship to least tern bill length 0; <20; <1 bill 1; 21–33; ~1 bill 2; 34–46; >1 bill 3; 47–52; 2 bills 4; >52; >2 bills Least tern bill length is estimated as 26 mm (Thompson *et al.*, 1997). 

Nest Site Selection

Citation	Location	Findings
Leslie et al. 1997	Oklahoma	We investigated attributes of 5 colony sites of least terns nesting on the Arkansas River from Tulsa to Muskogee, Oklahoma, in 1992 and 1993. We tested hypotheses that least terns selected areas to nest with the lowest amount and height of vegetation and nested near driftwood or similar debris. River flows prior to the nesting season in May 1993 scoured all sandbars, removing vegetation and increasing abundance of driftwood.  Table 2. Mean distance (m) from nests (N) to the narrowest and widest water barriers from the shores of sandbars to the river bank at 5 least tern colonies on the Arkansas River, 1992–1993. Year; Colony; N; Distance to narrowest water barrier $\bar{x}$; Distance to narrowest water barrier SE; Distance to widest water barrier $\bar{x}$; Distance to widest water barrier SE; P[a] 1992; Anchor Stone; 6; 184.2; 5.42; 40.3; 5.16; <0.001 Joe Creek; 8; 104.3; 3.72; 35.4; 5.18; <0.001 Tuft; 10; 31.7; 4.61; 38.5; 6.00; 0.457 1993; Joe Creek; 28; 70.3; 2.61; 57.3; 4.78; <0.021 Cedar Creek; 9; 142.4; 6.39; 30.5; 5.68; <0.001 Coweta; 8; 103.7; 3.82; 35.8; 6.70; <0.001 Hwy. 104; 21; 102.4; 3.81; 33.0; 3.34; <0.001 Fern Mtn.; 14; 162.3; 13.00; 48.7; 8.83; <0.001 a. Intracolony ranked t-tests.

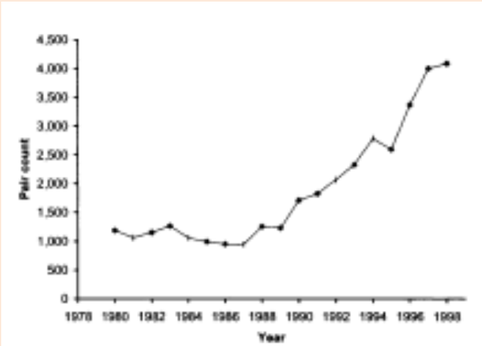
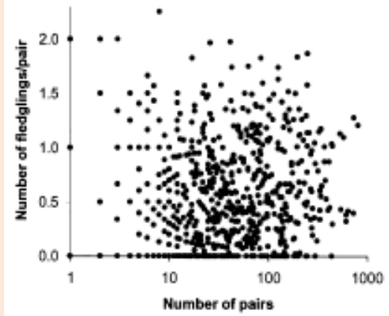
Citation	Location	Findings
Marcus et al. 2007	Platte River	**Table 3. Substrate composition and vegetation cover at Least Tern (LT) or Piping Plover (PP) nests (representing what the birds selected) and at random points within used plots (representing what was available to them in the area around their nest) at gravel mines along the Platte and Elkhorn rivers, Nebraska, 2000 and 2001. A used plot is defined as a plot with at least one nest initiated in it. Values with the same letter within a row are not significantly different (P > 0.05).** Attractant plots: Nests / Random; Control plots: Nests / Random Fine Sand LT 16.8 A 13.4 AB 16.1 A 8.5 B PP 18.7 A 17.2 A 19.8 A 12.5 B Coarse Sand LT 56.5 A 53.5 A 64.9 A 85.4 B PP 43.2 A 54.7 A 43.5 A 79.2 B Small Gravel LT 18.9 A 23.3 A 9.0 AB 3.0 B PP 28.4 A 20.4 A 29.6 A 17.6 A Large Gravel LT 7.9 A 10.0 A 9.5 A 3.0 B PP 10.0 A 6.2 A 11.6 A 1.6 A Vegetation LT 2.2 A 5.1 A 1.5 A 2.0 A PP 4.6 A 1.7 A 3.2 A 3.0 A The possibility of shifting nesting from active to inactive mining areas by using a deterrent (mylar flagging), an attractant (gravel and driftwood spread on bare sand), and a control (untreated sand) was evaluated. Experimental plots (mean 0.36 ha) were established at 18 different gravel mines, twelve in 2000 and seven (one repeat) in 2001 along the Platte and Elkhorn rivers prior to nesting season. Of 117 tern nests, 73% were in attractant, 2% in deterrent, and 26% in control plots. Of 23 plover nests, 61% were in attractant, 9% in deterrent, and 30% in control.
Schweitzer and Leslie 1999	Oklahoma	We measured habitat features in 12 active (≥3 nests) and four inactive (≤ 2 nests) colony sites on the alkaline flat at Salt Plains NWR, OK in 1992 and 1993.
Sherfy et al. 2012	Upper Missouri River	We evaluated nest-site habitat selection by least terns on constructed and natural sandbars by comparing vegetation, substrate, and debris variables at nest sites (n ¼ 798) and random points (n ¼ 1,113) in bare or sparsely vegetated habitats. Odds ratios for habitat variables indicated that avoidance of habitat features was the dominant nest-site selection process on both sandbar types, with nesting terns being attracted to nest-site habitat features (gravel and debris) and avoiding vegetation only on constructed sandbars, and avoiding silt and leaf litter on both sandbar types. We collected habitat data at 798 least tern nests (n ¼ 699 on constructed sandbars and n ¼ 99 on natural sandbars) and 1,113 random points (n ¼ 849 on constructed sandbars and n ¼ 264 on natural sandbars). Vegetation tended to be more abundant on natural than constructed sandbars. **Table 2.** Arithmetic means and 95% confidence limits for habitat variables at least tern nests and random points on natural and constructed sandbars on the Missouri River, 2006–2008. Constructed sandbars: Nest sites Mean 95% CL; Random points Mean 95% CL Natural sandbars: Nest sites Mean 95% CL; Random points Mean 95% CL WOODSTEM 1.3 0.9–1.7 1.1 0.8–1.2 4.9 3.3–6.7 3.4 2.6–4.1 WOODCOV 0.5 0.4–0.5 1.5 1.2–1.8 1.6 1.0–2.1 5.1 3.9–6.3 TERR_HERB 0.4 0.2–0.5 1.7 1.2–2.0 1.9 0.9–3.0 5.4 4.9–5.4 WPT_HERB 0.3 0.1–0.5 1.1 0.7–1.4 0.4 0.1–0.7 2.6 1.9–3.2 MAX_VEG_HT 0.2 0.1–0.2 0.3 0.2–0.3 0.4 0.3–0.4 0.6 0.5–0.6 MEAN_VEG_HT 0.2 0.1–0.2 0.2 0.2–0.2 0.3 0.2–0.3 0.5 0.4–0.5 SAND 72.1 70.4–73.7 66.2 62.1–66.1 78.5 75.9–79.3 59.6 55.7–63.4 SILT 6.1 4.7–7.4 16.0 14.7–18.7 1.6 −0.3–3.6 24.9 20.7–29.0 PEBBLE 11.9 10.7–12.9 11.4 10.0–12.6 14.1 11.3–17.9 9.7 7.0–12.2 GRAVEL 2.2 1.9–2.4 1.2 0.9–1.4 0.4 0.3–0.6 0.5 0.3–0.7 LEAF 0.2 0.1–0.3 1.1 0.9–1.7 0.3 −0.1–0.1 4.1 2.9–5.3 SM_DEBRIS 2.5 2.4–2.7 2.2 1.9–2.6 2.7 2.6–2.7 5.1 3.9–6.2 LG_DEBRIS 2.6 2.3–2.9 1.4 0.9–1.7 2.9 1.9–3.9 1.6 0.5–2.5 The 1east tern habitat model considers the ability of the habitat to meet the food and nesting needs of the species as an indication of overa11 breeding season habitat suitability. This model was designed to address the major habitat variables that affect the occupancy of potential nesting sites by least terns throughout their range, as indicated by the literature.
Sidle and Kirsch 1993	Platte River	**Table 2. Numbers of adult Least Terns (LT) and Piping Plovers (PP), percentages of all individual birds using sand pits, and the number of sandbar and sand pit nesting sites on the central and lower reaches of the Platte River, 1988-1991. Data were derived from Lingle (1993a), and Nebraska Game and Parks Commission (1988, 1989, and unpublished data), and Sidle *et al.* 1991.** Central Platte: Number of birds LT PP; % birds on pits LT PP; No. nesting sites Bar Pit Lower Platte: Number of birds LT PP; % birds on pits LT PP; No. nesting sites Bar Pit 1988 124 96 61 62 6 16 497 161 26 19 24 11 1989 194 94 94 64 6 16 409 188 14 13 38 10 1990 176 74 91 78 3 17 361 149 40 35 23 13 1991 158 58 87 76 3 14 295 73 30 37 35 11 Mean 163.0 80.5 83.2 70.0 4 15.8 390.5 142.8 27.5 26.0 30.0 11.2 SD 29.9 18.0 15.1 8.2 1.7 1.3 85.0 42.3 10.8 11.8 7.6 1.3 We found 225 sand pits of which 78 were suitable and 187 were unsuitable for nesting. Along the central Platte, where sandbar habitat is severely degraded, birds nested at 81% of the suitable sand pits (N = 32) at least once during 1988-1991, and most birds (61-94%) nested on sand pits. Along the lower Platte, where both sandbar and sand pit habitat are plentiful, birds nested at 60% of the suitable sand pits (N = 35) at least once during 1988-1991, and most birds (60-86%) nested on sandbars.
Carreker 1985	North America	
Conway et al. 2003	Texas	**Table 2. Habitat variables between interior least tern nests and random sites in the Prairie Dog Town Fork of the Red River, Childress County, Texas, 1998.** Habitat variable[1]: Nests (n = 20) x̄ SE; Random (n = 20) x̄ SE; F P Distance to upland (m) 285.7 28.9 257.1 24.5 0.64 0.43 Distance to vegetation (m) 10.9 2.1 10.5 2.7 0.09 0.77 Distance to mudflat (m) 106.1 25.6 97.2 26.8 0.01 0.97 Distance to water (m) 198.3 35.2 195.5 34.7 0.01 0.95 Vertical cover[2] 1.0 0 1.0 0 0.00 1.0 Relative elevation[3] 14.2 2.9 14.9 2.8 0.01 0.94 1. Means followed by the same letter in the same row did not vary (P > 0.05). 2. Vertical cover measure using Robel pole (Robel et al. 1970) from each of the cardinal directions, 4 m from the nest and/or random point. 3. Relative elevation measured as the difference between the nest/random sites and the mean elevation of the river bottom generated from random transects. **Table 3. Habitat variables between successful and non-successful interior least tern nests in the Prairie Dog Town Fork of the Red River, Childress County, Texas, 1998.** Habitat variable[1]: Success (n = 13) x̄ SE; Non-success (n = 7) x̄ SE; F P Distance to upland (m) 263.2 34.3 341.7 50.7 1.03 0.322 Distance to vegetation (m) 11.9 2.76 8.23 2.46 0.52 0.48 Distance to mudflat (m) 113.1 27.4 88.8 61.8 0.03 0.87 Distance to water (m) 196.4 45.1 203.0 55.2 0.05 0.83 Vertical cover[2] 1.0 0 1.0 0 0.00 1.0 Relative elevation[3] 13.7 3.97 15.5 2.12 0.34 0.57 1. Means followed by the same letter in the same row did not vary (P > 0.05). 2. Vertical cover measure using Robel pole (Robel et al. 1970) from each of the cardinal directions, 4 m from the nest and/or random point. 3. Relative elevation measured as the difference between the nest/random sites and the mean elevation of the river bottom generated from random transects.
Jenniges and Plettner 2008	Platte River	Most data from the central Platte and 80% of the nests reported in this paper were on areas managed specifically for the successful nesting of Least Terns and Piping Plovers. The number of nests on natural river formed islands without mechanical modification; 17 nests in 1979 all flooded before fledging (Faanes 1983), 36 nests between 1985 and 1990 of which 18 hatched and produced no known fledglings (Lingle 2004) and post-1990 all but two nests have been on human created habitats with no estimate of success. The only documented successful fledging of Least Terns from central Platte River islands comes from islands where the bare sand nesting substrate was created by some human activity such as vegetation clearing.

Citation	Location	Findings
		Table 2. Number of Least Tern nests at sites monitored along the central Platte River 1991 to 2005. Year	Managed Sandpits	Unmanaged Sandpits	Islands	Total 1991	7	NA	1	8 1992	31	NA	0	31 1993	29	NA	4	33 1994	34	46	0	80 1995	56	50	14	80 1996	50	19	13	62 1997	54	50	10	77 1998	51	NA	4	55 1999	24	NA	3	27 2000	21	NA	0	21 2001	25	NA	0	25 2002	37	NA	0	37 2003	47	NA	0	47 2004	43	NA	0	43 2005	44	NA	0	44 Total	475	125	49	647
Kirsch 1996	Platte River	Least terns did not strongly select siteswith distinct sets of features of either habitat, and they did not occupy most of the apparently available habitat of either type. **Table 2. Summary statistics for habitat variables derived from aerial videos of river habitat along the lower Platte River, Nebraska, 1989–90. Pairs of values from used and unused sites that differ significantly (ANOVA) are indicated with asterisks.** Nesting period and year	Date of video	Date of site-use survey	River flow m^3/sec*	Type of site	n	Habitat variables: Channel width (m)	Mean size sand areas (m^2)	Total area of sand (m^2) $\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE Initial 1989	3 Jun	9–13 Jun	69.5	Used	35	424	21	71*	8	12,348**	152 Unused	30	594	168	45	6	7,070	839 Renesting 1989	29 Jun	20–22 Jun	325.3	Used	16	452	19	48	10	1,553	353 Unused	33	594	14	32	9	785	250 Initial 1990	11 Jun	12–14 Jun	145.5	Used	16	474	17	538*	274	3,484**	599 Unused	33	445	15	33	9	1,363	457 Renesting 1990	4 Jul	2–3 Jul	57.2	Used	22	489**	12	510*	91	25,477	3,468 Unused	29	428	16	308	54	1,100	1,759 * Mean daily flow in cubic meters per second at the North Bend gauge river kilometer 116.8 (river mile 72.4) (Boehar et al. 1980, 1990). * Differences between used and unused within nesting periods, $P < 0.05$. ** Differences between used and unused within nesting periods, $P < 0.01$. **Table 3. Summary statistics for habitat variables (1,000 m^2) derived from aerial videos of sandpit sites along the lower Platte River, Nebraska, 1989–90. Pairs of values from used and unused sites that differ significantly (ANOVA) are indicated with asterisks.** Nesting period and year	Date of video*	Date of site-use survey	Type of site	n	Habitat variables: Area of usable sand[b]	Area of unusable sand[c]	Area of water	Area of sandpit site $\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE Initial 1989	4 Jul	9–13 Jun	Used	6	23.4**	6.1	14.7	6.4	87.6	35.2	158.5	57.2 Unused	33	11.9	1.4	6.7	1.5	51.1	11.2	89.2	14.2 Renesting 1989	4 Jul	20–22 Jun	Used	7	17.9	4.7	7.3	1.9	104.9*	44.0	145.4	48.9 Unused	32	12.8	1.7	8.1	2.0	46.2	8.9	89.9	14.8 Initial 1990	17 Jun	12–14 Jun	Used	12	31.4	4.5	3.6	12.4	135.8*	25.5	243.3**	26.9 Unused	23	22.1	7.7	22.7	4.8	66.5	13.8	131.2	19.7 Renesting 1990	17 Jun	2–3 Jul	Used	12	31.5	4.6	28.9	9.0	123.7	26.1	229.9*	24.8 Unused	23	22.1	7.7	26.7	6.6	71.8	14.5	138.2	21.9 * A single video was actually taken of sandpits in each year because sandpit habitat changes little during the breeding season. [b] Sand that was not disturbed by equipment or humans or covered with slurry run-off. [c] Sand that was frequently disturbed by equipment and humans or covered by slurry run-off. * Differences between used and unused within nesting periods, $P < 0.05$. ** Differences between used and unused within nesting periods, $P < 0.01$. **Table 10. Numbers of nesting sites and numbers of least terns counted on river and sandpit habitats between river kilometers 164.3 and 0 (river miles 102 and 0) of the lower Platte River, Nebraska, 1984–90. Data for 1984–88 were obtained from Nebraska Game and Parks Commission (unpub. data). Data for 1987–90 include Game and Parks Commission data as well as those collected in the present study.** Year	River: No. nest sites	No. pairs nesting	Sandpits: No. nest sites	No. pairs nesting 1984	0	0	10	137–144 1985	12	77	7	35 1986	7	154	8	77 1987	25	254	13	102 1988	24	256	13	118 1989	58	240	10	64 1990 early[b]	17	155	9	122 1990 revised[c]	23	144	11	74 * J. J. Dinan, Platte River interior least tern and piping plover nesting survey, Unpubl. reports to Nebr. Game and Parks Comm., Lincoln, Nebr. 1984–90. [b] Flooding inundated all sandbars during 14–19 June (Boehar et al. 1990). These 2 rows indicate the numbers counted before the flood and about 10 days after the flood had receded (20 June) when terns had begun to renest.	
Stucker 2012	Upper Missouri River	We found 866 of 869 nests on barren sandy substrates (86% dry sand, 14% wet sand) that were largely devoid of vegetation. Among these nests, the annual percentage on dry or wet sand substrates did not differ between natural and constructed habitats. We observed proportionally more nests among wet substrates on natural sandbars (24-51% wet sand) compared to constructed sandbars (7-17% wet substrate) in each year. Among nests, significant differences were observed in substrates, amount of debris, and measures of vegetation between natural and created habitats, and between microhabitat at nests and the surrounding area. In general, Least Tern nest sites had coarser and larger substrate materials at the nest, more debris, and shorter and less vegetation compared to areas within the 3 m surrounding area. Nests in constructed habitats had a greater proportion of coarse substrates and less vegetation or debris than naturally created habitats. We determined fate, success or failure, for 95% of nests. Nest success among constructed sandbars (70%, n = 675) was greater than that observed among nests on natural sandbars (37%, n=149). Table 4. Count of Least Tern nests and surrounding area habitat assessments included in study by point type (nest, surrounding area) on natural and constructed sandbars, for year of the study. Natural: Nest, Surrounding Area; Constructed: Nest, Surrounding Area; Nest Total 2006: 64, 64, 208, 208, 272 2007: 52, 52, 212, 212, 264 2008: 40, 40, 293, 286, 333 Total: 156, 156, 713, 706, 869  a. b. c. Figure. 2. Photographs demonstrating range of habitat variability at Least Tern nests on constructed sandbars at time of discovery. Each nest is surrounded by a 1-m2 quadrat frame. a). Nest in wrack line, with abundant small debris, nest furniture, and limited vegetation on wet sand and silt substrate (RM 827); b) Nest with nest furniture, small debris, in contact with the nest bowl, consolidated moist sand substrate with surficial pebbles (RM 770); c) Dry, unconsolidated sand substrate (no vegetation, no debris) (RM 755).	
Stucker et al. 2013	Upper Missouri River	We documented habitat characteristics and fate for 869 Least Tern nests during 2006–2008. Eighteen percent of nests were located on natural sandbar habitats and 82% on constructed sandbar habitats. Several habitat characteristics of nest sites and surrounding areas differed between natural and constructed sandbars (Table 3). We found 866 of 869 nests on barren sandy substrates (86% dry sand, 14% wet sand) that were largely devoid of vegetation. Among these nests, the annual percentage on dry or wet sand substrates did not differ between natural and constructed habitats, except in 2008 (X^2 = 46.3, df = 2, P < 0.001), when 46% of nests in natural habitats and 7% of nests in constructed habitats were located on wet sand. We observed proportionally more nests among wet substrates on natural sandbars (24–51% wet sand) than on constructed sandbars (7–17% wet substrate) in each year. In general, Least Tern nest sites in natural habitats had slightly more vegetation than created habitats, with nest sites exhibiting less vegetation than surrounding areas. We suspect that Least Terns appear to key in on areas of greater pebble concentration, extending >3 m from nest. Nests on constructed sandbars were in areas with more cobble, gravel, and pebble than the surrounding areas. Similarly, nests on natural sandbars were in areas with more sand and less silt than surrounding areas. Although management of debris to encourage nesting in higher areas is possible, manipulating wet substrates under current water-management strategies aimed at increasing discharge through the nesting period will likely be contrary to evolved nest-selection patterns.	

Faanes, C. A. 1983. Aspects of the nesting ecology of Least Terns and Piping Plovers in central Nebraska. Prairie Naturalist. 15(4): 145-15.
Lingle, G. R. 2004. Platte River Recovery Implementation Program Draft Environmental Impact Statement Response. Submitted to The Central Platte Natural Resource District. April 28, 2004. Grand Island, Nebraska.

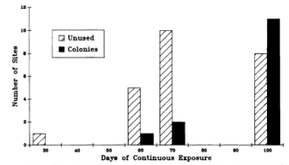
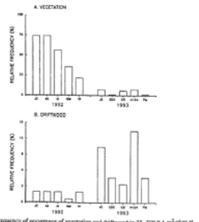
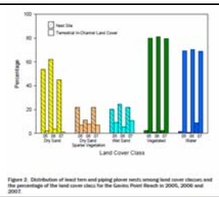
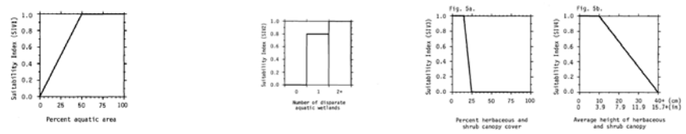
Nest Density

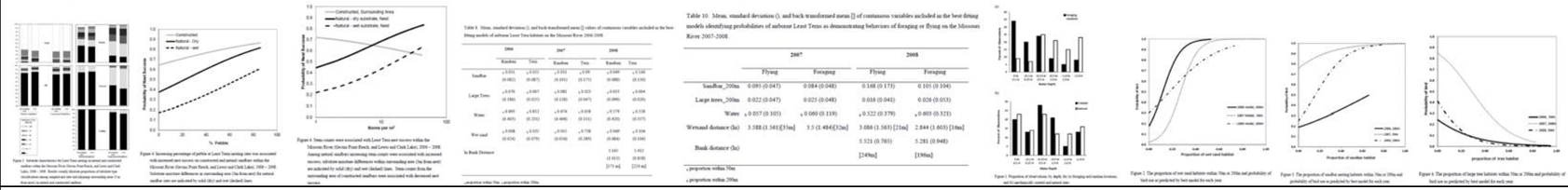
Citation	Location	Findings
Akcakaya et al. 2003	California	We examined a metapopulation model for the California least tern that can be used to predict persistence of populations along the Pacific coast and the effects of various management actions. In our analysis of fledgling data from several sites, we did not detect significant density dependence.

Citation	Location	Findings
		 Fig. 2. Total count of California least tern pairs from 1980 to 1998 (from Keane 2001) in California, USA, and Baja California, Mexico.  Fig. 5. The relationship between the size of a California least tern colony (in number of pairs) and the number of fledglings per pair in the colony, based on pooled data from all colonies in California, USA, and Baja California, Mexico, 1971–1998. The colony size explains <2% of the variation in the number of fledglings per pair, suggesting a lack of density-dependent effects on fecundity.
Schweitzer and Leslie 1999	Oklahoma	We measured habitat features in 12 active (≥3 nests) and four inactive (≤ 2 nests) colony sites on the alkaline flat at Salt Plains NWR, OK in 1992 and 1993. Pooled internest data from 1992 and 1993 indicated that least tern nests were distributed randomly ion active colony sites. TABLE 2.—Spatial distribution pattern of least tern nests from nearest-neighbor analyses (Krebs, 1989) in active colony sites Colony site / Number of nests / Area (ha) / Density (nests/ha) / Mean distance between nests (m) / Expected distance between nests (m) / Index of aggregation (r) / Standard deviation / Test statistic (z) / Spatial pattern 1992 F 11 40.0 0.26 94.0 95.4 1.0 15.0 −0.1 random G 12 16.0 0.75 71.2 57.7 1.2 8.7 1.5 random K 20 37.5 0.53 66.4 68.5 1.0 8.0 −0.3 random L 15 37.5 0.40 90.6 79.1 1.2 10.7 1.1 random M 5 7.5 0.67 70.5 61.2 1.2 14.3 0.6 random N 6 8.8 0.69 101.7 60.4 1.7 12.9 3.2 uniform 1993 DC 10 27.5 0.36 90.8 82.9 1.1 13.7 0.6 random F 25 10.9 0.23 56.0 33.0 1.7 3.4 6.7 uniform FC 14 27.0 0.52 89.5 69.4 1.3 9.7 2.1 uniform G 7 2.0 0.35 44.7 26.7 1.7 5.3 3.4 uniform KL 10 29.5 0.34 62.4 85.9 0.7 14.2 −1.7 random CH 11 1.1 0.10 21.4 15.8 1.4 2.5 2.2 uniform $\bar{x}_{overall}$ 12 20.4 0.60 71.6 64.8 1.1 9.7 0.7 random

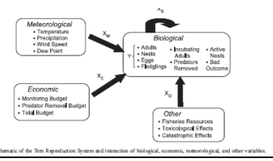
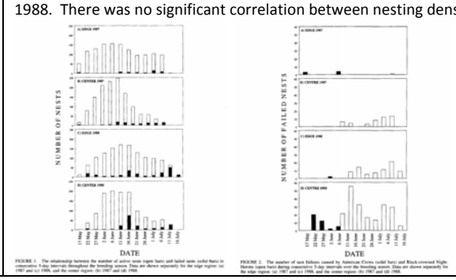
Habitat Quality and Quantity

Citation	Location	Findings
Sidle et al. 1992	Platte River	We observed extensive mortality (eggs and chicks) of the endangered interior population of the Least Tern and threatened Piping Plover caused by natural flooding during the 1990 breeding season along the Platte River, Nebraska. Aerial videography of the Platte River before and after the flood revealed a 78% reduction of perennial vegetation on sandbars. The flood scoured vegetation from sandbars and greatly increased the amount of barren sandbar habitat that nesting Least Terns and Piping Plovers use. **Table 1. The number of monitored Least Tern (LT) and Piping Plover (PP) nests inundated along the Platte River between the Loup River confluence and the Missouri River, 1986-90.** Yr / No. Nests Before First Flood (LT/PP) / No. Flooded (LT/PP) / % Flooded (LT/PP) / No. Renests[1] (LT/PP) / No. Renests Flooded (LT/PP) / Total % Flooded (LT/PP) / Maximum Daily Flow (cms) NB/LV[2] 1986[3] 156/28 18/6 15/21 0/0 0/0 15/21 421.7/795.0 1987[4] 69/17 5/0 7/0 0/0 0/0 7/0 502.8/484.5 1988[5] 77/52 5/1 4/5 0/0 0/0 4/5 143.2/102.6 1989[6] 61/13 21/7 34/54 24/5 0/0 28/39 551.1/520.0 1990[7] 94/28 94/28 100/100 95/15 4/6 52/85 911.5/1715.5 [1]Least terns and piping plovers renested after losing eggs or chicks from other causes in 1986 and 1988 but renested no new nests in 1987 and 1989 after flooding. [2]Flows measured at U.S. Geological Survey gauging stations at North Bend and Louisville, Nebraska. [3]Flooding occurred in late June after most nests had hatched. [4]Flooding occurred in late May-early June before birds had initiated many nests. [5]The first flood inundated sandbars along the entire river and occurred during peak nesting. The second flood inundated sandbars below Salt Creek (U.S. Army Corps of Engineers river mile 39) and occurred in late July after most tern but no plover renests had hatched. The second flood killed all chicks on sandbars below river mile 26.
Smith and Renken 1991	Mississippi River	Habitat characteristics of interior Least Tern colony sites in the Mississippi River valley adjacent to Missouri were examined during 1985-1989. During 1985, Least Tern nesting colonies were located on sand islands and sand bars that differed from unused sand islands by the length of time sites were continuously exposed above the river. Sites used by Least Terns were more likely than unused sites to be continuously exposed for at least 100 d during the period 15 May-31 August. Colony sites did not differ from unused sites in general habitat type (sand island, sandbar on wooded island or sand island accreted to the shore), nor in the amount of vegetation cover (≤ or > 10% vegetation cover). Within the study region, only one unused site that appeared to be suitable for tern nesting could be identified. The majority (87%) of 2027 nests located during 1986-1989 were on fine (52%) or coarse sand (35%) substrates. Fifty-two percent of 1690 nests located in 1986-1989 were ≤25 cm from a stick or other drift material.

Citation	Location	Findings
		 FIGURE 1. Distribution of interior Least Tern colonies and unused sites by category of days of continuous exposure, lower Mississippi River, 1985. TABLE 1. Mean ± SE of habitat characteristics of interior Least Tern colonies and unused islands or sandbars in the Mississippi River within the region where Least Terns nest (Lower River Mile 827 to Upper River Mile 30), and sites outside the breeding range (Upper River Mile 31 to Upper River Mile 361). Colonies (n = 14) / Unused islands (n = 24) / Outside breeding region (n = 48) Mean days of coninuous exposure during 15 May–31 August: 101.6 ± 4.0 / 83.7 ± 4.1 / 85.4 ± 3.8 Mean % vegetation cover: 8.9 ± 4.2 / 4.5 ± 2.2 / 16.8 ± 12.3
Leslie et al. 1997	Oklahoma	We investigated attributes of 5 colony sites of least terns nesting on the Arkansas River from Tulsa to Muskogee, Oklahoma, in 1992 and 1993. Nests at all colony sites surveyed were situated on sandbars at maximal distances to the narrowest water barrier from shores of sandbars to the river bank. Our results suggest that river flows that elevate sandbars before the nesting season ultimately will enhance recovery of endangered least terns on the Arkansas River by reducing losses of nests to flooding, retarding establishment of vegetation, and increasing abundance of driftwood.  Figure 1. Frequency of occurrence of vegetation and driftwood in 75–200 0.1-m² plots at 5 colony sites of least terns on the Arkansas River from Tulsa to Muskogee, Oklahoma, prior to (summer 1992) and after (summer 1993) scouring floods. Colony abbreviations: JC, Joe Creek; AS, Anchor Stone; IS, Indian Springs; ISW, Indian Springs West; TF, Taft; CDC, Cedar Creek; CS, Cowets; H104, Highway 104; PM, Pern Mountain.
Sherfy et al. 2008	Upper Missouri River	A focus of the Missouri River Research Program is the need to quantify acreage of emergent sandbar habitat as identified in the Biological Opinion. This Task focuses on least tern use of natural and newly created habitats. In 2007 we also recorded habitat characteristics associated with locations where successful foraging was observed and a nearby (≤200 m) randomly selected location. Characteristics collected at each of these sites included GPS location, water depth, temperature, turbidity, and fisheries landscape information. This information will help us describe and define least tern foraging habitat.
Sherfy et al. 2009a	Upper Missouri River	 Figure 2. Distribution of least tern and piping plover nests among land cover classes and the percentage of the land cover class for the Gavins Point Reach in 2005, 2006 and 2007. Table 9. Numbers (n) of least tern and piping plover nests where habitat characteristics were evaluated on the Missouri River in summer 2008. Study Area / River Miles / Species / n / Measurements Gavins Point / 754–795 / Least Tern / 192 / n, 3m Piping Plover / 223 / n Lewis & Clark Lake / 826–832 / Least tern / 140 / n, 3m Piping Plover / 61 / n Lake Sakakawea / 1390–1508 / Piping Plover / 93 / n, 3m, 10m, 2rand, s Least Tern / 29 / n, 3m, 10m, 2rand, s
Sherfy et al. 2009b	Upper Missouri River	This report describes the field methods that were used in 2009 and provides an overview of the data that were collected. Formal analyses and generation of research products are being conducted to adequately address each Task objective.
Carreker 1985	North America	A review of literature pertaining to the least tern indicated that habitat for the three subspecies can be characterized by the same environmental variables.  Figure 3. The relationship between the percent of the area within the average maximum flight distance from the potential nesting habitat that is aquatic habitat and the suitability index value for least tern food. Figure 4. The relationship between the number of disparate aquatic wetlands within the average maximum flight distance from the potential nesting habitats and the suitability index value for least tern food. Figure 5. The relationships between vegetation cover and the suitability index values for least tern reproduction.
Kirsch 1996	Platte River	Terns tended to use river sites with large midstream sandbars and a wide channel, and large sandpit sites with large surface areas of water relative to unused sites on both habitats. Terns apparently did not use all potentially available sandbar and sandpit sites because discriminant function factor scores for used and unused sites overlapped broadly for both habitats. Terns did not prefer one habitat over the other. Although proportions of available sites used were greater on sandpits than on the river, proportions of available sand used did not differ between habitats. Proportion of terns using each habitat was similar to proportion of available sand on each habitat. Table 20. Summary of habitat variables at least tern colony sites on sandbars along the lower Platte River, Nebraska, 1987–90. Year / No. sites / Distance to bank (m) / % of colony area[a] flooded 1987 / 3 / 5–25 / 0–5 1988 / 4 / 25–120 / 0–10 1989 / 6 / 85–165 / 80–95 1990 / 6 / 0–190 / 0–100 [a] That part of a nesting site that terns actually used for nesting, brooding and feeding chicks, and loafing. Table 21. Summary of habitat variables at least tern colony sites on sandpits along the lower Platte River, Nebraska, 1987–90. Year / No. sites / % of colony area[a] perimeter bordered by water / Distance to abundant vegetation[b] (m) 1987 / 5 / 25–75 / 5–200 1988 / 3 / 25–75 / 40–200 1989 / 6 / 25–75 / 10–200 1990 / 7 / 25–75 / 25–400 [a] That part of a colony site that terns actually used for nesting, brooding and feeding chicks, and loafing. [b] Abundant vegetation = >50% cover and >30 cm tall.
Stucker 2012	Upper Missouri River	During 2007-2008, successful foraging sites (N=416) were compared to a nearby random location (N=416) to evaluate evidence of habitat selection during successful foraging. We also used systematic surveys in 2006-2008 to identify Least Tern airborne locations within the river corridor (2006:966 sites, 2007:2940 sites, 2008:2003 sites). Observed nest success among constructed sandbars was 1.8x greater than that recorded among nests on natural sandbars, but a greater proportion of nests on natural bars were in or adjoining moist habitats where they were frequently destroyed. Historically, scoured habitats and the comparative abundance of debris within sites may have been a cue indicating safe habitats as river stage decreased. Water management regimes during this study favored survival of nests on higher elevation sandbars which included limited areas of wet substrates, typical of mechanically constructed sandbars. Within the greater landscape, Least Tern airborne locations within the river corridor were best explained by multi-scale logistic regressions; birds concentrated in areas with higher proportions of sandbar and wet sand habitats (200 m radius), while avoiding trees (50 m radius). When applied to a reserved sample of random locations, results suggest that in any given year, 40-97% of river corridor habitats were likely unsuitable for airborne terns. Despite the surficial visual uniformity of many aquatic habitats on the Missouri River, successful foraging (bird observed

Citation	Location	Findings
		retrieving fish from water) by Least Terns was associated with shallow, slack-water microhabitats adjacent to emergent sandbars and within areas with a greater proportion of sandbar habitat. 
Faanes 1983	Platte River	During 1979, sandbars within about 84 km of river channels were surveyed for tern and plover nests. Nest searches were initiated on 19 May and continued through 26 June. All nests, both hatched and active, were submerged by rising water on 21 June. Normal vegetational succession created some problems interpreting the impact of vegetative growth on nest site selection. When early arrivals began selecting nest sites the sandbars were virtually bare. By mid-June, nesting cover had changed from the date of nest initiation. Thus, later nesting birds were required to choose and evaluate habitat that was much different from that found earlier. The greater mean depth to moisture also suggested that terns preferred higher and drier habitat.

Predation

Citation	Location	Findings
Butchko and Small 1992	California	This paper documents the development and results of the predator control strategy in 1988 through 1991 in a major colony at Camp Pendleton Marine Corps Base near Oceanside, CA. It is offered as a case history which may be useful to resource managers as they plan a predator control program for a wildlife resource of their concern. The goal of the program was to make the colony free of all known and potential predators during the tern nesting season. The 1988, leghold and raptor traps, shooting, gas cartridges, and baited eggs were used 7 days a week, and tern nesting results were outstanding. An estimated 246 pairs fledged an estimated 365-409 birds. Virtually no nests or birds were lost to predators. In 1989, raptor control efforts were expanded to include pole traps for owls. An estimated 163 nests producing only 67 fledglings (Massey 1989). The primary reason for this was considered to be that ground squirrel and harrier control was done reactively rather than proactively. In 1990, 293 pairs fledged and an estimated 275 - 335 birds (Belluomini 1991a). In 1991, 328 pairs fledged an estimated 377 birds, which was the highest ever recorded at Camp Pendleton (Belluomini 1991b). While many factors contributed to this success, the reduction of predation is considered to be a major factor.
Lavers et al. 2010	General	We review predator removal studies focusing on observed demographic responses of bird populations to the removal campaigns. From the 800+ predator removal programs identified, a small fraction (n = 112) reported demographic responses of bird populations. Change in productivity was the most commonly reported response, which on average increased by 25.3% (2.5 SE) with predator removal. The predicted increase in productivity resulting from predator removal alone was insufficient to reverse the predicted population decline for 30–67% of bird species considered, suggesting that in many cases, removal of predators must be performed in combination with other conservation actions in order to ensure a stable or increasing population.
Shwiff et al. 2005	California	Approximately 17% of the state's breeding populations of the least terns remain at the Camp Pendleton Marine Corps Base near Oceanside, CA. This paper presents an empirical analysis of annual, fixed-cost budgets expended for reproduction-monitoring and predator-removal activities to protect this shorebird. The ex post study covered the inclusive 7-year period between 1995 and 2001. Mean net current annual reproduction-monitoring and predator-removal budgets were US$80,115 and US$78,178, respectively; annual fiscal data were converted to bproxyQ variables of personnel time (h) for analysis of economic effects. Mean time spent in reproduction-monitoring (3.12 h/day) and predator-removal activities (6.96 h/day) differed greatly. Expenditures for both reproduction-monitoring and predator-removal staff hours were associated with greater counts of tern eggs and adults, with increased monitoring hours predictive of finding more tern nests and fledglings and increased predator-removal hours linked with fewer fledgling counts. Economic variables showed that the greater monitoring dollars invested in hours used to measure tern reproduction, the more Fledglings counted. Regression analysis was used to obtain the estimates of the coefficients, and from this we were able to formulate equations to forecast the number of adults, nests, eggs, and fledglings, given fixed increases of 25%, 50%, and 100%, in the economic variables. Total nests and fledglings were influenced most by monitoring hours. Even 25% increased funds for this activity was forecasted to yield 105.6% and 38.6% more nests and fledglings counts, respectively (Table 6). New nests lack prey items so it is not necessary for predator removal staff to invest more time in protecting these nests until the presence of eggs. The difficulty in protecting fledglings is reflected by the negative sign on the coefficient for predator removal hours in the fledgling forecast. Fledglings are mobile, erratic, and vulnerable to a multitude of predators; this makes protection complex and time consuming. In many cases, predator-removal staff increases their daily work hours, but remove fewer predators. This explains the negative relationship between forecasts of greater funds for predator removal hours and the fledgling's variable. Table 3 Predator-removal and reproduction-monitoring budgets Year 2003 Dollars[a] \| Year \| Predator budget (US$) | Rank | Monitor budget (US$) \| Rank \| Total budget (US$) | Rank | | 1995 | 76,170 | 2 | 71,046 | 6 | 147,216 | 5 | | 1996 | 78,753 | 1 | 68,477 | 7 | 147,230 | 4 | | 1997 | 73,019 | 4 | 76,594 | 3 | 149,613 | 3 | | 1998 | 73,165 | 3 | 81,275 | 1 | 154,440 | 1 | | 1999 | 70,366 | 7 | 75,890 | 4 | 146,256 | 7 | | 2000 | 71,601 | 6 | 75,328 | 5 | 146,929 | 6 | | 2001 | 72,763 | 5 | 80,005 | 2 | 152,768 | 2 | [a] Adjusted for inflation.  Fig. 2. Schematic of the Tern Reproduction System and interaction of biological, economic, meteorological, and other variables. Table 6 Percent increase forecast for the dependent variables as a result of an increase in the independent variables | Development stage/ dependent variable | Scaled independent variable: Monitor hours (%) 25 | 50 | 100 | Predator-removal hours (%) 25 | 50 | 100 | Total hours (%) 25 | 50 | 100 | | Adults | 7.6 | 14.1 | 24.7 | 4.8 | 9.1 | 16.7 | 8.1 | 16.1 | 32.2 | | Total nests | 105.6 | 211.5 | 423.2 | 29.1 | 58.4 | 117.1 | 1.2 | 2.6 | 5.5 | | Eggs | 4.2 | 8.5 | 16.9 | 10.4 | 20.8 | 41.5 | 4.2 | 8.4 | 16.7 | | Fledglings | 38.6 | 77.2 | 154.3 | −42.9 | −85.8 | −171.7 | 24.9 | 49.8 | 99.5 |
Brooks et al. 2013	South coast	The DSR of nests was primarily related to colony site, but year and estimates of predation risk also were related to DSR. Predation was the principal cause of identifiable nest loss, accounting for 47% of nest failures when the two years of data were pooled (n = 257). For failed nests, cause of failure was classified as abandoned (eggs were cold and/or moisture was seen on the eggshell), depredated (signs of predation such as broken eggshells and yolk stains and/or evident predator tracks leading to the nest scrape), overwashed (cup next to a study nest contained saltwater, marked eggs found in wrack debris that was recently deposited), failure to hatch (hatching never occurred although parents continued to incubate through subsequent nest observation intervals), or unknown (empty scrapes were observed before estimated hatch date and no sign of predation or overwash were evident). All other causes of nest loss (i.e., nesting substrate such as a shell rake collapsed and human error) were defined as 'other'. Nest fate was recorded as undetermined for nests where there was no clear evidence of success or failure. Cape Romain NWR has trapped mammalian nest predators although to date these efforts have not occurred consistently across years or locations. Control of avian nest predators, if a desired approach of the Refuge, would prove far more difficult and likely require efforts to remove animals individually.
Brunton 1997	Northeast	The spatial and temporal patterns of breeding success of Least Terns were studied within a large colony at Sandy Point, Connecticut, an ocean beach on the East Coast of North America. Contrary to the 'selfish herd' hypothesis, nests located in the center of the colony suffered from significantly higher levels of predation and had correspondingly lower hatching and fledging success than nests located at the edge (particularly during 1988). Causes of chick mortality were difficult to determine once chicks reached a week old and left the immediate region of the nest. Chicks older than one week moved to the salt marsh area adjacent to the colony where long grasses presumably provided shelter. There were no significant differences between years in nesting densities for the two regions (edge: Mann-Whitney U = 5; center: U = 8, n, = 12, n2 = 12 in both cases). However, there was a significantly lower nesting density in the edge region compared to the center for both years (1987: Mann-Whitney U = 52; 1988: U = 65, n, = 12, n2 = 12 in both years, P < 0.01). Edge nests maintained a stable density throughout the nesting season, which may in part be due to the definition of edge. Center region densities peaked in early June during both years with slightly higher densities observed during 1988. There was no significant correlation between nesting densities and the proportion of nests that failed for either edge or center areas (edge: r = -0.31, P > 0.10; center: r = 0.25, P > 0.60), suggesting that other effects such as predation may be more important. 

Citation	Location	Findings
Koenen et al. 1996	Oklahoma	Coyotes are major egg predators of Least Tern and Snowy Plovers and were the only mammalian egg predator identified on the alkaline flats from 1991 to 1994. About 5-60% of monitored tern and plover nests have been lost to coyotes annually from 1977-1994 at Salt Plains NWR. Annual nest success for Least Terns was increased by electric fences only during 1991. However, pooling the four years of data effectively reduced the standard error and demonstrated an overall efficacy of electric fences to improve nest success of Least Terns. Although fences proved to be effective at reducing predation on Least Terns and Snowy Plovers, there were a number of problems associated with fences on the alkaline flats.
Kruse et al. 2001	Upper Missouri River	Predation accounted for nearly half of all nest loss, the remaining was due to inundation, weather, abandonment, human, and unknown. Chick shelter were not used and chicks ran past shelters to use vegetated areas or driftwood to escape predators.
Leslie et al. 2000	Oklahoma	Predation was confirmed at only one colony in 1992; 18 nests and 10 chicks (Table 2) were lost to canine (Canis sp.) predation between 21 and 24 July 1992. Five episodes of predation were confirmed in 1993. Seventeen nests (13% of 133 losses; Table 2) were lost to canine predation at three colonies, and three nests were lost to a domestic dog (C. familiaris) at another colony. Additional mammalian predation was indicated by the loss of 11 nests in a single episode at another colony. We observed a redtailed hawk (Buteo jamaicensis) capture one chick on 19 July 1993. Combined losses of nests and chicks to predators were greater in 1992 than 1993, but it is likely that many losses to predators went undetected, particularly in 1993 (Table 2). In all cases of confirmed mammalian predation in 1993, depths of water barriers between colonies and the vegetated river bank were <15 cm. However, presence, width, or depth of water barriers did not completely deter canids.
Szell and Woodrey 2003	Mississippi River	In addition to the direct flooding of nests, river level fluctuations also influence the degree of predation a colony site experiences. Sandbars remained isolated from the main shoreline for a greater portion of the incubation period during 1995 because river levels remained high. High river levels decrease the potential for mammalian predators gaining access to the colony sites via dikes or accreted shoreline. Raccoons gained access to the sandbar early in the season when river levels dropped sufficiently causing the sandbar's attachment to the riverbank shoreline. The loss of 154 nests (41%) on Porter Lake Dikes in 1997 was also due to shoreline accretion and subsequent Coyote (Canis latrans) predation. St. Francis Dikes was the only colony site that had consistently high nest and reproductive success across years. Such success may be attributed to the sandbar's "isolation" throughout the nesting season (water flowing around it throughout the summer), thus reducing the likelihood of mammalian predation. Barred Owls attacked incubating Least Tern adults during the evening, taking both adults and juveniles as indicated by foot tracks often leading to a pile of tern feathers and mandibles. Foot tracks were also found leading from one nest to another. Such "foot-hunting" behavior has been noted in this and other owl species (McMillian 1998). Because Barred Owls are considered opportunistic predators and are known to nest typically from March through April (Mazur and James 2000), it is likely that juvenile Barred Owls would be fed Least Tern chicks.
Kirsch 1996	Platte River	Most egg mortality was caused by predation on sandpits and by flooding on sandbars. Predation was suspected as the major cause of loss for chicks on both habitats. Path analysis revealed no strong or consistent correlations among mortality, numbers of nests and chicks, track trails of intruders into colonies, and habitat variables at colonies on either habitat.

Table 17. Mean frequencies of track trails at least tern colony sites on sandbars and sandpits on the lower Platte River, Nebraska, 1987–90. The nesting period ran from nest initiation to when 75% of the nests became inactive (hatched or destroyed). The chick rearing period followed the nesting period and ended when the last chick fledged.

Habitat	Year	n	Total season x̄	Total season SE	Nesting x̄	Nesting SE	Chick rearing x̄	Chick rearing SE	Unused sites[a] x̄	Unused sites[a] SE
Sandbar	1987	3	0.76	0.19	0.82	0.22	0.44	0.10		
	1988	4	0.68	0.16	0.88	0.23	0.30	0.06		
	1989	6	0.66	0.28	0.71	0.34	0.74	0.46		
	1990	6	1.34	0.38	0.57	0.26	2.08	1.10	2.50	0.50
Sandpit	1987	5	0.39	0.04	0.46	0.14	0.32	0.12		
	1988	3	0.47	0.07	0.37	0.03	0.70	0.25		
	1989	6	0.87	0.22	0.62	0.23	1.03	0.36		
	1990	7	1.66	0.40	1.45	0.54	1.73	0.37	3.04	0.87

[a] Data from unused sites were collected in 1990 from 3 sandbar and 4 sandpit sites.

Table 18. Correlations among number of nests, numbers of eggs lost due to predation and abandonment, and frequency of different types of track trails for least tern colonies along the lower Platte River, Nebraska, 1987–90. Correlations between track trail types and other variables were Spearman rank, and all other correlations were Pearson product-moment.

Habitat	Number	No. eggs lost to: Predation	No. eggs lost to: Abandonment	Frequency of track trails: Terrestrial predator	Human	Owl	ATV[a]	Heavy equipment	All trails
Sandbars	Nests	0.317	0.574**	0.107	0.303	0.233	0.113		0.316
	Depredated eggs		0.288	0.494**	0.170	0.399	−0.144		0.565
	Abandoned eggs			0.039	0.143	−0.027	−0.024		−0.037
Sandpits	Nests	0.049	0.631**	−0.306	0.286	0.182	0.013	0.220	−0.185
	Depredated eggs		0.148	−0.357	−0.018	0.284	−0.392	0.433*	0.235
	Abandoned eggs			−0.529*	0.459	0.247	−0.324	0.433*	−0.308

[a] All-terrain vehicle.
* Significant correlation, $P < 0.05$.
** Significant correlation, $P < 0.01$.

Table 19. Spearman rank correlations of proportions of least tern chicks known alive on each visit and numbers of chicks hatched at a colony, with frequency of different types of track trails on sandbar and sandpit colony sites on the lower Platte River, Nebraska, 1987–90.

Habitat	Chick variables	Frequency of track trails: Terrestrial predators	Humans on foot	Owl	ATV[a]	Heavy equipment	All track trails
Sandbars	Proportion of chicks alive (n = 156 visits)	−0.002	−0.010	0.007	−0.223*		−0.117
	Number of chicks hatched (n = 18 sites)	0.242	0.497	0.306*	0.561*		0.569
Sandpits	Proportion of chicks alive (n = 223 visits)	−0.234**	−0.023	−0.106	−0.093	0.016	−0.172
	Number of chicks hatched (n = 18 sites)	0.222	0.567*	0.226	0.386	0.030	0.413*

[a] All-terrain vehicle.
* Significant correlation, $P < 0.05$.
** Significant correlation, $P < 0.01$.

Stage Survival

Citation	Location	1st Winter Survival	2nd Winter Survival	Adult Survival	Study Design	Sample Size
Massey et al. 1992	California	The years that produced the smallest percentage of returning adults were 1982-1983. In 1982,208 chicks were banded but many did not survive to fledge. The productivity ratio was 0.4, and only 2% returned as adults to breed. In 1983, the productivity ratio was back up again (1.0) but very few fledglings of that year survived the two-year interval until age of first breeding.		This study encompasses the years 1983 to 1989, when we checked Venice nests systematically to find and color-band all banded, breeding adults. Table 2 shows the age at which 186 banded adults were first known to breed. While many bred at age two years, the largest age group was the three-year-olds. The few that were trapped at age equal to or greater than five years presumably began their breeding lives elsewhere and later emigrated to Venice. The overall survival rate for adults, once they had returned to breed, was 0.88. When "young breeders" were separated from "older breeders," survival rates were 0.81 and 0.92, respectively.		
Renken and Smith 1995a	Mississippi River	During the non-breeding season, Least Terns migrate to and winter in Central and South America (U.S. Fish and Wildlife Service 1990). It is not known what mortality factors affect adult Least Terns in that region, but incidental hunting and pesticides may pose threats to the survival of adults on the wintering grounds.	The estimated adult annual survival rate was 85% (SE = 0.057; 95% CI = 0.73-0.95). During our studies, adult Least Terns died from several different causes. At least eight adult terns were killed by Barred Owls at one colony during 1988. We identified the predators from foot tracks at kill sites. Another three adult terns were shot in 1991 and another adult was killed by an all-terrain vehicle in 1984.		During 1987-1990, 194 adult Least Terns were color-banded to estimate the adult annual survival rate for the population in the lower Mississippi River valley. A total of 103 resightings were compiled and 40% (78) of the 194 banded adults was resighted at least once during 1988-1992. Program JOLLY was used to calculate an adult annual survival rate. Model B, which assumes a constant survival rate and time-specific capture probabilities was the most appropriate model for the data.	

TABLE 1. Demography of the Least Tern population in California and at Venice Beach 1977–1989.

	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989
California													
Number of breeding pairs	775	832	1,000	1,160	1,067	1,130	1,264	1,046	1,003	960	945	1,253	1,240
Estimated number of fledglings	449	418	696	769	733	511	894	518	682	902	632	1,130	764
Productivity ratio[1]	0.58	0.50	0.70	0.66	0.69	0.45	0.71	0.50	0.66	0.94	0.67	0.90	0.62
Venice colony													
Number of breeding pairs	35	67	87	157	150	150	140	82	96	104	109	165	137
Estimated number of fledglings	37	75	145	240	200	60	140	94	113	113	82	192	134
Productivity ratio	1.1	1.1	1.7	1.5	1.3	0.4	1.0	1.2	1.2	1.1	0.8	1.2	1.0
Number of chicks banded	0	87	157	291	139	208	173	84	158	117	92	285	149
Percent of returns[2]	NA[3]	10	10	4	9	2	2	4	9	7	NA[4]	NA[4]	NA[4]

[1] Number of fledglings/pair of adults.
[2] Based on number of chicks banded each year that returned as adults to breed.
[3] Not applicable—no chicks banded in 1977.
[4] Not applicable—chicks banded 1987–1989 did not reach age of major first breeding during the study.

TABLE 4. Demographic analysis.

		Normal years: Estimate	Normal years: 95% Confidence Interval	ENSO years§: Estimate	ENSO years§: 95% Confidence Interval
Return rates					
Hatchling*	S_h	0.16	(0.13, 0.18)	0.03	(0.01, 0.05)
Young breeders†	S_y	0.81	(0.72, 0.87)	0.82	(0.66, 0.96)
Older breeders‡	S_o	0.92	(0.88, 0.95)	0.79	(0.68, 0.89)
Demographic parameters					
Hatchling productivity**	P_H	1.00	(0.84, 1.20)	0.98	(0.88, 1.08)
Breeding life††	L	9.63	(6.55, 14.48)	4.97	(3.46, 8.10)
Lifetime productivity‡‡	P_L	1.49	(0.93, 2.34)	0.15	(0.06, 0.30)

* Percentage of chicks hatched at Venice that return as adults to breed.
† Annual survival rate for birds breeding at ages two and three.
‡ Annual survival rate for birds breeding at ages four or older.
** Hatchlings produced per breeder per year.
†† Expected number of breeding years per adult: $L = 1 + S_y + S_o/(1 - S_o)$.
‡‡ Number of progeny per adult expected to survive to breeding age: $P_L = S_h \times P_H \times L$.
§ Estimates based on data from the 1982–1983 El Niño Southern Oscillation.

TABLE 5. Age-dependent survival rates.

Age	n	Survival estimate	95% Confidence Interval
2	31	0.82	(0.69, 0.93)
3	73	0.80	(0.72, 0.88)
4	64	0.90	(0.83, 0.96)
5	52	0.92	(0.85, 0.98)
6	43	0.93	(0.85, 0.98)
7	43	0.87	(0.78, 0.96)
8	31	0.87	(0.76, 0.97)
9	20	0.92	(0.80, 1.00)
10	10	0.76	(0.50, 1.00)
11	3	1.00	(1.00, 1.00)
12	2	0.60	(0.17, 1.00)

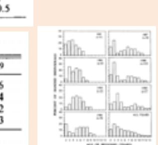
TABLE 2. Age of adults at Venice Beach when first trapped on nests (154) or identified by color code (32); n = 186.

Number of adults	Age (years): 2	3	4	5	6	7	8	9	10
Trapped	37	64	37	6	5	2	2	0	1
Identified by color code	21	11	0	0	0	0	0	0	0
Total	58	75	37	6	5	2	2	0	1
Percent	31	40	20	3.2	2.7	1.1	1.1	0	0.5

TABLE 3. Return rate of banded adults breeding at Venice (both known and unknown age).

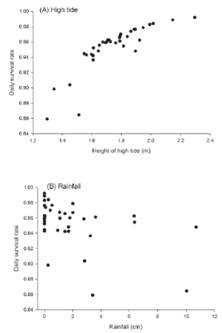
	1983	1984	1985	1986	1987	1988	1989
Total returns[1]	0	37	42	58	59	64	76
Total possible returns[2]	0	56	48	65	69	74	104
New birds	56	11	23	11	15	40	32
Return rate (%)	—	66	88	89	86	86	73

[1] Adults that nested at Venice the previous year.
[2] Banded adults trapped for the first time and unbanded mates of banded birds.



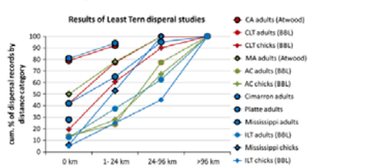
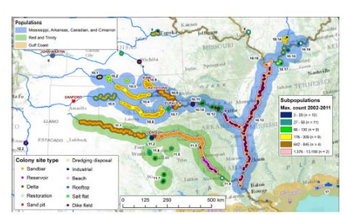
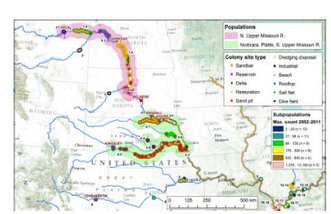
Citation	Location	1st Winter Survival	2nd Winter Survival	Adult Survival	Study Design	Sample Size
			TABLE 1. Capture-recapture summary statistics for adult Least Terns in the Mississippi River valley. Definitions of variables are taken from Pollock et al. (1990). n_i = Total number of animals captured in the ith sample (includes captured and resighted birds). m_i = Number of marked animals captured in the ith sample (resighted birds). R_i = Number of n_i that are released after the ith sample. r_i = Number of R_i released at t that are captured again. z_i = Number of animals captured before t_i not captured at t_i and captured again later. Period / Date / n_i / m_i / R_i / r_i / z_i 1 / 1987 / 37 / — / 37 / 16 / — 2 / 1988 / 37 / 3 / 37 / 19 / 13 3 / 1989 / 73 / 14 / 73 / 41 / 18 4 / 1990 / 104 / 40 / 104 / 22 / 19 5 / 1991 / 24 / 24 / 23 / 5 / 17 6 / 1992 / 22 / 22 / 22 / — / —			
Whittier and Leslie 2009	Oklahoma	A minimum estimate of chick survival over a 19 day period (assuming lost signals = mortality) was 5% of chicks surviving to the end of the study period and a high (assuming signal failor) of 27%. Five chicks fledged, six died, and fate of the remaining 14 was undetermined. Although chicks appeared to allocate a substantial part of their daily movement to nocturnal periods, average rates of diurnal (7.6 m/h) and nocturnal movement (12.5 m/h) did not differ. Chicks primarily moved toward stable sources of water. TABLE 1—Movement pattern and fate of chicks (%) of the least tern (*Sterna antillarum*) after they abandoned their nest at Salt Plains National Wildlife Refuge, Alfalfa Co., Oklahoma. Two chicks were excluded because they were tracked for <5 days and never moved >50 m from the nest. Movement pattern / All chicks / Fledged / Died / Unknown Traveler (travel in a definite direction; n = 5) / 24 / 20 / 20 / 60 Wanderer (wander randomly from the nest; n = 14) / 66 / 36 / 14 / 50 Sedentary (remain in the vicinity of nest; n = 2) / 10 / 50 / 50 / 0			Our objectives were to estimate survival of chicks and study their movement patterns. In 1999, we attached radio transmitters to 25 chicks and tracked them for 2–19 days depending on retention of transmitter. Because fate could not be determined always when signals were lost, a high and low estimate of survival was calculated. Kaplan-Meier estimates of survival were calculated first by assuming that all lost signals represented mortalities, and second that these transmitters failed and chicks were alive.	
Aron 2005	Upper Missouri River	16% (Massey et al. 1992)	85.3 – 94.1% (Thompson 1982)	85.3-94.1%; 88%; and 85% (SE = 5.7%, n = 194) (Thompson 1982, Massey et al. 1992, Renken and Smith 1995a)	Citing other studies	
Bailey and Servello 2008	Northeast	Chick survival estimates were 0.73, 0.74, and 0.14 for the three colonies. **Table 3. Estimates of total numbers of banded fledglings and fledgling success for three count methods compared to values based on estimates of fledgling numbers from mark-resighting analyses at two Least Tern breeding colonies in Maine, 2002-2003. Fledgling count data are presented in Table 2.** Crescent Surf-Laudholm Beach (No. chicks banded = 96) / Higgins Beach (No. chicks banded = 62) No. of fledglings / Fledging success[1] / No. of fledglings / Fledging success[1] Based on total banded fledglings[2] / 70 / 0.73 / 46 / 0.74 Based on highest daily number of banded fledglings (single high-count method)[3] / 56 / 0.58 / 32 / 0.52 Based on highest daily number of banded fledglings plus highest number 14 d before and after the highest count / 91 / 0.95 / 36 / 0.58 Based on Thompson and Slack (1984) method[4] / 60 / 0.63 / 29 / 0.48 [1]Fledging success is calculated as number of fledglings divided by the number of chicks hatched (i.e., no. of chicks banded for this analysis). [2]Total numbers of banded fledglings present during the nesting season were calculated from chick survival estimates for the interval hatch to fledging and known numbers of chicks banded. Note: estimates of total number of banded fledglings based on mark-resighting analyses and total numbers of individual banded fledglings identified in field surveys were nearly identical for the two colonies (70 vs. 71 and 46 vs. 44, respectively). [3]The number of banded fledglings potentially present on a single day on a resighting survey (Table 1). [4]$Count_2 + (Count_4 - 0.55\ [Count_2])$ where: $Count_2$ = the potential number banded fledglings in the second week of the fledgling period, $Count_4$ = the potential number of banded fledglings in the fourth week of the fledgling period. No fledglings were present in subsequent two-week periods.			Chick survival was estimated using mark-resighting data and methods for estimating survival when individuals cannot be observed for ≥ one period. In the present study, chicks generally could not be observed between day two and just prior to fledgling age. Consequently, resighting data were analyzed relative to chick age rather than survey date, and four age intervals were used in the analysis to produce an estimate of chick survival: 0-20 d (chick interval), 21-23 d, 24-26 d, and 27-29 d, except that the chick interval for the analysis was 1-20 d for Crescent Surf as noted earlier. **Table 1. Chick survival estimates for three Least Tern breeding colonies in Maine, 2002-2003.** Colony / Number of chicks banded / Proportion of banded chicks resighted as fledglings[1] / Chick survival (Φ) based on mark-resighting analyses: Daily (Φ ± SE) / 21-d chick period Crescent Surf-Laudholm Beach / 96 / 0.74 / 0.9864 ± 0.0069 / 0.73 Higgins Beach / 62 / 0.71 / 0.9855 ± 0.0042 / 0.74 Crescent Surf Beach / 55 / 0.15 / 0.9098 ± 0.0148 / 0.14 [1]Chicks were classified as fledglings when age was ≥21 d.	
Dugger et al. 2000	Mississippi River	The mean daily survival rate for Least Tern chicks at Rkm 1431 was 0.951 (SE 0.03) with 95% confidence intervals of 0.899–1.003. Mean daily survival rate for Least Tern chicks at Rkm 1481 was 0.972 (SE 0.03) with 95% confidence intervals of 0.911–1.034. Estimated survival of Least Tern chicks throughout the entire 17-d fledging interval was 0.43 at Rkm 1431 and 0.62 at Rkm 1481. TABLE 2. Laridae fledging success reported as a percentage of the number of hatched chicks that fledged. Species / Fledging success / Location source Great Skua (*Catharacta skua*) / 7.8%–95.3% / Hamer et al. (1991) Glaucous-winged Gull (*Larus glaucescens*) / 52%–74% / Hunt and Hunt (1976) Roseate Tern (*Sterna dougallii*) / 90%–97% / Nisbet et al. (1995) Common Tern (*Sterna hirundo*) / 59%–88% / Langham (1972) Common Tern / 68.3%–78.9% / LeCroy and LeCroy (1974) Common Tern / 80% / Nisbet (1978) Common Tern / 0% / Uttley et al. (1989) Arctic Tern (*Sterna paradisaea*) / 0% / Uttley et al. (1989) Least Tern / 43%–62% / This Study	Finally, although estimates of Least Tern survival from fledging to age at first reproduction are still not available for any tern population, estimates of chick survival can be used to decompose current estimates of sub-adult survival from hatching to age at first reproduction. For example, we can divide Massey et al.'s (1992) estimate of California Least Tern survival from hatching to 2–3 yr (0.16) by our estimates of chick survival to produce an estimate of Least Tern survival from fledging to age at first reproduction of 0.26–0.37.		Our purpose was to estimate chick survival from hatching to fledging, for Least Terns nesting at two sites on the Lower Mississippi River in Missouri, using mark-recapture methodology. We banded 110 Least Tern chicks during 1995 on sand island nesting colonies situated at river kilometers (Rkm) 1431 and 1481 on the Lower Mississippi River. We used Program JOLLY to compute survival point estimates, their associated variances, and goodness-of-fit tests for Jolly-Seber open population models.	
Lombard et al. 2010	South coast	This model yielded a daily survival probability of 0.95 ± 0.02 for a period survival estimate of 0.30 ± 0.11.			We report nest and chick survival and colony-site dynamics of the Least Tern. These results are the first for the Caribbean and were derived with likelihood-based approaches from 4640 nests and 44 chicks	

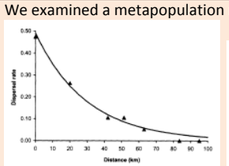
Citation	Location	1st Winter Survival	2nd Winter Survival	Adult Survival	Study Design	Sample Size
					fitted with transmitters monitored in 52 colonies at St. Croix, U.S. Virgin Islands, 2003–2006. All nesting sites were surveyed every four days from mid-April to mid-August.	
Sidle et al. 1992	Platte River	Extremely high flows between modal nest initiation dates and modal fledging dates (6 weeks later) are likely to cause severe mortality.			**Table 1. The number of monitored Least Tern (LT) and Piping Plover (PP) nests inundated along the Platte River between the Loup River confluence and the Missouri River, 1986-90.** Yr: 1986[a] 1987[b] 1988[c] 1989[d] 1990[e] No. Nests Before First Flood (LT/PP): 156/28 69/17 77/52 61/13 94/28 No. Flooded (LT/PP): 18/6 5/0 5/1 21/7 94/28 % Flooded (LT/PP): 13/21 7/0 4/5 34/54 100/100 No. Renests[f] (LT/PP): 0/0 0/0 0/0 24/5 95/15 No. Renests Flooded (LT/PP): 0/0 0/0 0/0 0/0 4/6 Total % Flooded (LT/PP): 13/21 7/0 4/5 28/39 52/83 Maximum Daily Flow (cms) NR/LY[g]: 421.7/795.0 302.8/484.5 145.2/102.6 351.1/520.0 911.3/1713.3 [a]Least terns and piping plovers renested after losing eggs or chicks from other causes in 1986 and 1988 but initiated no new nests on the river after flooding. [b]Flows measured at U.S. Geological Survey gauging stations at North Bend and Louisville, Nebraska. [c]Flooding occurred in late June after most nests had hatched. [d]Flooding occurred in late May-early June before birds had initiated many nests. [e]The first flood inundated sandbars along the entire river and occurred during peak nesting. The second flood inundated sandbars below Salt Creek (U.S. Army Corps of Engineers river mile 26) and occurred in late July after most tern but no plover renests had hatched. The second flood killed all chicks on sandbars below river mile 26.			
Akcakaya et al. 2003	California	**Table 2. Hatching to fledging survival rate of California least tern at Venice beach, California, USA.** From Massey et al. (1992): Year, Nests, Fledglings, Fledglings per nest From Minsky (1984): Nests, Total eggs, Failed eggs, Hatchlings per nest S_h[a] 1981: 150, 200, 1.3333, 170, 325, 16, 1.8179, 0.7335 1982: 150, 60, 0.4000, 189, 329, 49, 1.4817, 0.2700 1983: 140, 140, 1.0000, 152, 291, 20, 1.7834, 0.5607 1984: 82, 94, 1.1463, 87, 184, 11, 1.9866, 0.5770 Mean[b]: 1.1599, 1.8626, 0.6237 [a] Fledglings per hatchling, or the hatchling to fledgling survival rate (S_h), is calculated as the ratio fledglings/nest:hatchlings/nest. [b] Mean excludes 1982.	**Table 3. Survival (S; subscripts indicate age class) parameters of the California least tern for normal and El Niño/Southern Oscillation (ENSO) years (95% CI in parentheses).** Parameter: Normal years / ENSO years / Reference or calculation S_1 S_2 S_3 S_4 (hatchling to first breeding): 0.16 (0.13 to 0.18) / 0.03 (0.01 to 0.05) / Massey et al. (1992) S_5 and S_6: 0.81 (0.72 to 0.87) / 0.81[a] / Massey et al. (1992) S_{4+}: 0.92 (0.88 to 0.95) / 0.79 (0.68 to 0.89) / Massey et al. (1992) S_h (hatchling to fledgling): 0.6237 / 0.2700 / Massey et al. (1992) and Minsky (1984) S_0 S_1 (fledgling to age 2): 0.3167 / 0.1372 / $= (S_0 S_0 S_1 S_2)/(S_0 S_0)$ S_0 and S_1: 0.5627 / 0.3704 / $= \sqrt{S_0 S_1}$ [a] Massey et al. (1992) estimated S_5 and S_6 for ENSO years as 0.82 (95% CI = 0.66, 0.96). Because survival is unlikely to increase in an ENSO year, in this model we assumed these survival rates for ENSO years to be equal to the average they estimated for normal years (0.81). **Table 5. Survival rate of adult California least terns at Camp Pendleton, California, USA (from Collins et al. 1998).** Yr: Survival 1987: 0.926 1988: 0.921 1989: 0.920 1990: 0.866 1991: 0.899 1992: 0.781 1993: 0.894 1994[a]: 0.731 Mean: 0.8867 SD: 0.0510 CV: 0.0576 [a] The estimate for 1994 is the product of survival rate and resighting probability, and is not used in calculating the average and the variability. Fledglings per pair in t; Survival from (t-1) to t Fig. 4. Number of fledglings per pair of California least tern in year t, in the North San Diego population (California, USA), as a function of the adult survival rate from year $t-1$ to t in Camp Pendleton (which is part of this population) for years 1988–1994. The coefficient of correlation is −0.12 (SE = 0.44).		We examined a metapopulation model for the California least tern that can be used to predict persistence of populations along the Pacific coast and the effects of various management actions.	

Citation	Location	1st Winter Survival	2nd Winter Survival	Adult Survival	Study Design	Sample Size
Brooks et al. 2013	South coast	The probability (± SE) of a chick surviving from hatching to fledging = 0.449 ± 0.01 (n = 92 chicks). DSR of chicks was negatively related to tide height and rainfall. **Table 3. Daily survival rate (SE) and probability of success for Least Tern (*Sternula antillarum*) nests in Cape Romain National Wildlife Refuge, South Carolina, 2009-2010. Probability of success is the daily survival rate raised to an exponent equal to 21 (incubation days). All estimates were calculated by using coefficients and standard errors of the most-supported model from Table 1.** Variable / Daily Nest Survival / Probability of Success **Colony site** Lighthouse Island: 0.916 (0.006) / 0.157 Cape Island: 0.930 (0.003) / 0.218 Middle White Banks: 0.996 (0.001) / 0.926 **Year** 2009: 0.959 (0.003) / 0.412 2010: 0.977 (0.001) / 0.613 **Predation risk** No: 0.986 (0.001) / 0.749 Yes: 0.916 (0.004) / 0.160  **Figure 3. Daily survival rate (DSR) of Least Tern (*Sternula antillarum*) chicks in Cape Romain National Wildlife Refuge, South Carolina, 2009-2010.**			Daily survival rate (DSR) of nests and chicks was examined at four natural nesting sites in Cape Romain National Wildlife Refuge, South Carolina, 2009-2010. Daily survival rate of chicks was estimated through banding and re-sighting efforts but was restricted to Middle White Banks due to logistical constraints at other colony sites. We banded 1-2 day old chicks at Middle White Banks with a unique color-band combination (2009: n = 35; 2010: n = 57). After the first day of banding, 2-hr observations were conducted every 2-5 days to re-sight banded chicks. In 2009, re-sight surveys were conducted by one or two observers simultaneously, and in 2010 two or three observers conducted re-sight surveys simultaneously.	
Kirsch 1996	Platte River	Mortality of young and productivity did not differ between sandbars and sandpits, but varied tremendously among colonies within both habitats. This variation and the small number of sites studied would make it difficult to detect a small difference in productivity between habitats. Chick mortality also was slightly density dependent on sandpits; reasons for this relationship remain unclear because there were no strong, consistent correlations between number of chicks, disturbance by predators or humans, and chick mortality. **Table 14. Comparisons between sandbars and sandpits of instantaneous rates of laying, egg, and chick mortality for least tern young along the lower Platte River, Nebraska, 1987–90.** Habitat: Sandbar / Sandpit Mortality rate / Sandbar *n* / Sandbar $\bar{x}$ / Sandbar SE / Sandpit *n* / Sandpit $\bar{x}$ / Sandpit SE / Student's *t* / df / *P* Laying: 19 / 0.216 / 0.017 / 21 / 0.207 / 0.024 / 0.320 / 38 / 0.751 Egg: 19 / 0.548 / 0.083 / 21 / 0.471 / 0.082 / 0.661 / 38 / 0.513 Chick: 18 / 0.873 / 0.077 / 19 / 0.722 / 0.076 / 1.394 / 35 / 0.174 Overall: 19 / 1.225 / 0.066 / 21 / 1.082 / 0.048 / 1.785 / 38 / 0.087			Available habitats of both types were characterized and quantified using aerial videography (1989-90), and habitat use was assessed from census data (1987-90). Productivity of adults and causes and correlates of egg and chick mortality were estimated (1987-90).	See table.

Thompson, B.C. 1982. Distribution, colony characteristics, and population status of least terns breeding on the Texas coast. Ph.D. Dissertation. Texas A&M University, College Station. 124 pp.

Site Fidelity, Immigration, and Emigration

Citation	Location	Findings
Boyd and Thompson 1985	North America	The observation of one nest in the Quivira National Wildlife Refuge, Kansas, producing young from a four year old female banded in Port Lavaca, Calhoun Co., Texas, approximately 1250 km S of QNWR and a four year old male banded at QNWR demonstrates that genetic mixing is occurring naturally between members of these adjacent populations. The magnitude of such mixing cannot be estimated from this single event, but the low probability of a band recovery/return for this species (ca. 0.5%) suggests that it could be a reasonably prevalent phenomenon. Site tenacity of the Least Tern is only 50-55% at QNWR (Boyd, unpubl. data), whereas populations on the west coast experience 90-95% site tenacity (Atwood, pets. comm.). Both Atwood, with Least Terns, and Austin (1951) with Common Terns, suggest that the birds tend to disperse more from less stable nesting sites.
Lott et al. 2013	Central US	We compiled data from > 200 published papers and reports into a Geographic Information System (GIS) for additional analyses. We define a breeding "population" as a collection of nest and/or colony locations that are "connected through frequent dispersal, occupying a collection of habitat patches that lack large intervening areas of nonhabitat relative to dispersal distances." We used ILT breeding colony locations from 2002 to 2012 and dispersal information to identify 16 populations and 48 subpopulations. More than 90% of ILT and >83% of river km with suitable nesting habitat occur within the two largest populations.  **Figure 3.** Dispersal distance frequencies, by age class, from published literature (Atwood and Massey 1988, Boyd 1993, Lingle 1993; Renken and Smith 1995a; Akcakaya et al. 2003) and 1261 band recovery records from 1923 through 2011, obtained from the U.S. Geological Survey, Patuxent Wildlife Research Center, Bird Banding Laboratory. Published studies had variably sized study areas, all of which were restricted to <96 km from the original banding site. These studies would only produce long-distance band recoveries in the unlikely event that banded individuals were found dead outside of these small study areas or captured again by researchers in distant locations.  **Figure 4.** Detailed map of ILT populations 5–11 (and their constituent numbered subpopulations) spanning the Southern Great Plains and the Lower Mississippi River valley. See Table 1 for more information. All-caps text above red chevrons gives major dam names.  **Figure 5.** Detailed map of ILT populations 1-4 (and their constituent numbered subpopulations) spanning the Northern Great Plains. See Table 1 for more information. All-caps text above red chevrons gives major dam names.
Renken and Smith 1995b	Mississippi River	During 1987-1991, 194 adult and 1,674 juvenile interior Least Terns were banded during the breeding season within a 210 km stretch of the lower Mississippi River. We attempted to resight Least Terns in subsequent years (1988-1992) to determine site fidelity (the return of a bird to the colony where banded) and distances terns moved from banding sites. Forty percent (78) of all color-marked adults and 1% (19) of all banded juveniles were resighted at least once following banding. Adult Least Terns did not display strong site fidelity. Terns were as likely to use another colony in a subsequent year even when the original colony site was available (above the river surface), or they had nested successfully at the original colony. Our observations support the notion that birds inhabiting unpredictable and variable environments will display little site fidelity toward breeding sites. Even though Least Terns showed little site fidelity to a colony, terns tended to return to the same general region of the river. Most (97%) adult terns moved approximately 9.1 km (median distance, rang = 1.5-80.0 km) from the original banding colony. Least Tern chicks also displayed little site fidelity for their natal colony. Only one of 19 resighted chicks returned to its natal colony site in a subsequent year. Some adults and a chick banded within our study region dispersed 300-1000 km from the Mississippi River valley.

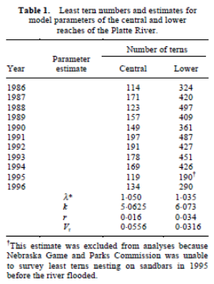
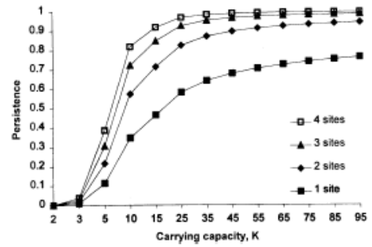
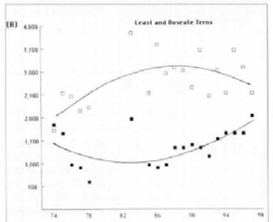
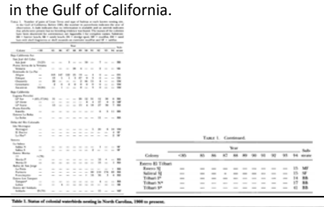
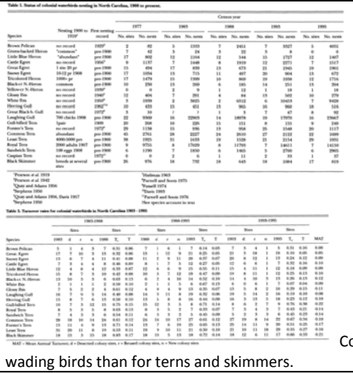
Citation	Location	Findings
Atwood and Massey 1988	California	From 1973 to 1983 a total of 5,425 Least Tern chicks were banded The degree to which Least Terns exhibit year-to-year fidelity to particular colony sites, as well as fidelity toward their natal colony sites, was examined using banding recoveries from 328 known-age birds (banded as chicks) obtained in California. From 1983 to 1986, individuals at Venice Beach had high rates of return (annual rate of return = 78%) to colony sites where they had nested during the preceding year; of those few birds that switched colony sites between successive years, most (92%) moved only short distances from the previous area. Least Terns also showed a significant tendency toward nesting at their natal colony site. TABLE 1. Numbers of Least Tern chicks banded in California, 1973 to 1984. Year Colony[a] 73 74 75 76 77 78 79 80 81 82 83 84 HB 4[b] – – – 35 115 98 105 66 61 69 104 VB – – – – – 112 168 218 213 178 173 47 TI – – 5 44 111 – – – 19 48 40 7 MISC 87 44 21 301 151 159 355 360 401 277 633 596 Total 91 44 26 345 297 386 621 683 699 564 915 754 [a] Colony site abbreviations as follows: HB = Huntington Beach; VB = Venice Beach; TI = Terminal Island; MISC = colonies other than HB, VB, or TI. [b] Number of chicks banded. TABLE 2. Fidelity to natal colony sites in Least Terns. Colony[a] Year Total recoveries Observed recoveries from natal colony site Expected recoveries from natal colony sites[b] χ^2 HB 1983 24 11 4.30 12.72** HB 1984 27 9 4.54 5.36* HB 1985 15 5 2.52 2.93 HB 1986 26 9 3.69 8.90** VB 1983 52 30 16.74 15.45** VB 1984 44 28 13.82 21.21** VB 1985 44 28 13.33 23.16** VB 1986 50 30 14.55 23.13** [a] Colony abbreviations as follows: HB = Huntington Beach, VB = Venice Beach. [b] Assuming random distribution of breeders relative to their natal colony sites. [c] Based on 2 × 1 contingency table comparing observed vs. expected numbers of recoveries at natal and nonnatal colony sites. Chi-square significance levels (df = 1) indicated as follows: ** = $P < 0.01$; * = $P < 0.05$. TABLE 3. Annual and intercolony variation in site fidelity in breeding Least Terns. Colony Year: 1983[a] 1984 1985 1986 $\bar{x}$ Venice Beach 86 (21) 58 (59) 82 (54) 86 (66) 78 Huntington Beach 54 (13) 42 (31) 52 (31) 79 (19) 57 Terminal Island 50 (22) 38 (34) 36 (50) 47 (38) 43 [a] Percent of the previous year's banded population (n indicated in parentheses) that was documented as breeding during the specified year. Thus, of 21 banded individuals that nested at Venice Beach in 1982, 86% returned to breed at this site in 1983.
Akcakaya et al. 2003	California	We examined a metapopulation model for the California least tern that can be used to predict persistence of populations along the Pacific coast and the effects of various management actions.  Fig. 6. Dispersal-distance function based on data from a Massachusetts, USA, population of least tern (data from Atwood 1999). The dispersal model assumes a maximum dispersal distance of 80 km. The curve shows the function $y = a \cdot \exp(-x / b)$, where $a = 0.49$ and $b = 29$ were estimated from a regression of ln(dispersal rate) on distance in km ($R^2 = 0.97$). Table 6. Dispersal rates used in our California least tern metapopulation model: percentage of individuals moving to other populations in 5 distance classes. Dispersal model Distance from source population (km) Low Medium High 0 (remain in the source population) 90 70 50 <30 km 10 15 25 30–60 km 0 10 15 60–90 km 0 5 10 >90 km 0 0 0
Carreker 1985	North America	Nesting sites generally are characterized as unstable areas created and maintained by tidal action or flooding. Due to the sometimes transitory nature of nesting habitat, least terns have been described as having strong group adherence and weak site tenacity, which may aid in discovering recently created habitat (McNicholl 1975). In some cases, however, site tenacity can be a more important determinant of site selection than the physical characteristics of an area (Gochfeld 1983; J. L. Atwood, Department of Biology, University of California, Los Angeles; unpubl.). Least tern colonies can display high site fidelity by continuing to use an area year after year for as long as the site remains suitable (Burger 1984), including marginal sites where successful reproduction has occurred previously (Massey and Atwood 1979). In New Jersey, colony sites were abandoned only when predation, human disturbance, or vegetation encroachment reached intolerable levels (Burger 1984). Terns returned to and nested at sites where colonies were completely wiped out the previous year when such sites had been in use for several years. Year-to-year fidelity has been documented for least terns in California by Atwood (unpubl.), who also found that least terns tended to nest in the general vicinity of their natal colonies. Of 190 banded birds studied, approximately 50% nested within 25 km of where they were hatched, and over 80% nested within 50 km of their natal site.

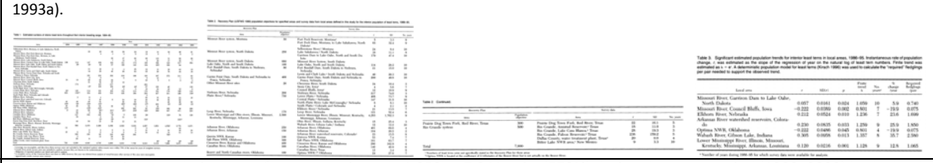
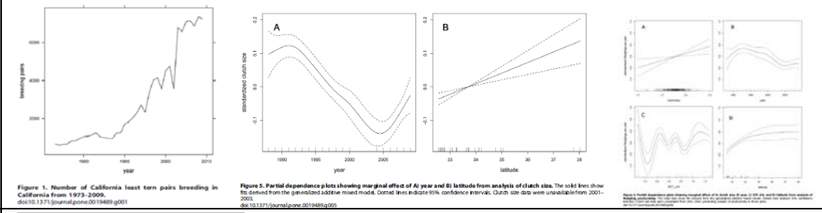
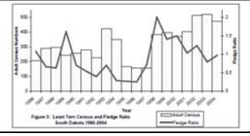
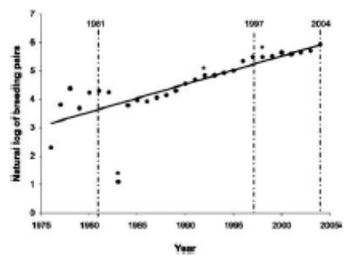
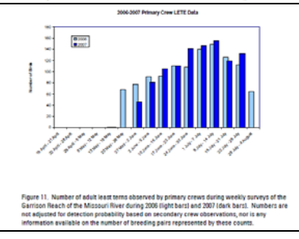
Overwinter Habitat

Category 1	Category 2	Citation	Location	Findings
Reproductive Success	Overwinter Habitat	Sidle and Harrison 1990	Central US	The wintering area of interior least terns is unknown. However, least terns of unknown populations or subspecies are found during the winter along the Central American coast and the northern coast of South America from Venezuela to northeastern Brazil. Roger Boyd (personal communication 1986) reports that about 35 least terns have been recaptured in South America, mostly in Guyana. One interior least tern banded by Boyd, was captured in El Salvador two years later. Also, a banded California least tern was recaptured in Guatemala.

Population Trends through Time

Citation	Location	Observed Trend	Study Design
Boyce et al. 2002	Platte River	The risk of extinction for least terns will be reduced if we can sustain the small central Platte River population in addition to the larger population on the lower Platte.	We review the theoretical rationale of bet-hedging and suggest applications for conservation management of least terns in Nebraska. A stochastic density-dependent model derived using diffusion theory (Foley 1994, 1997) was used to estimate the time to extinction for both the central and lower Platte terns: $$T_e = \frac{2n_0\left[k - \frac{n_0}{2}\right]}{V_r}, \quad (10)$$ where r is the per capita rate of population change; n_0 is the ln of initial population size; k is the ln of carrying capacity and Vr is the variance in r

Citation	Location	Observed Trend	Study Design
		 **Table 1.** Least tern numbers and estimates for model parameters of the central and lower reaches of the Platte River. Year / Parameter estimate / Number of terns: Central / Lower 1986: 114 / 324 1987: 171 / 420 1988: 123 / 497 1989: 157 / 409 1990: 149 / 361 1991: 197 / 487 1992: 191 / 427 1993: 178 / 451 1994: 169 / 426 1995: 119 / 190† 1996: 134 / 290 λ*: 1·050 / 1·035 k: 5·0625 / 6·073 r: 0·016 / 0·034 V_r: 0·0556 / 0·0316 †This estimate was excluded from analyses because Nebraska Game and Parks Commission was unable to survey least terns nesting on sandbars in 1995 before the river flooded.  **Figure 2.** Effects of carrying capacity (K) and the number of populations (sites) on the probability of persistence for interior least terns	
Brown et al. 2001	Northeast	Since 1974, the Least Tern population on Long Island has increased 49 percent from 1,719 to 2,560 pairs in 1997 (linear regression, P = 0.029), but a quadratic regression analysis indicated that this population has been declining, from a peak of 3,800 pairs, since 1983 (P = 0.006, Fig. 4a).  Figure 4. Estimates (nesting pairs, y-axis) of (A) Least (open boxes) and Roseate (solid boxes) Terns and	We identified changes in nesting populations for 18 species of colonial waterbirds in Jamaica Bay Wildlife Refuge (JBWR) and on Long Island (LI), during 1974 - 1998, to provide a basis for future wildlife management decisions. We assume that that count data were obtained using sufficient sampling effort and represent reasonable estimates of actual numbers because (1) waterbird colonies are conspicuous and often found in traditional locations; and (2) considerable effort has been given to improve the accuracy (identify/correct sources of error) and precision (standardized technique) of colonial waterbird census data since the mid- l 970s. Count data were log-transformed to reduce heteroscedasticity (Zar 1996). We used linear and quadratic regression models to determine fit (based on highest r~ value) of transformed count data to year for each species during 1974-98 (Table 1). We reject the null hypothesis (no trend) if P < 0.05 and report the mean± 1 SD population size.
Burger 1989	Northeast	In successive years a trend has indicated increased population levels, and reproductive success, and decreased and then increased number of colonies. During this time, monitoring and managing efforts have increased, suggesting that they are effective in very slowly restoring population levels. 	This paper presents an overview of ten years of monitoring and managing of least terns in New Jersey under the auspices of the Endangered and Non-Game Species Program of New Jersey. The program involves monitoring population levels and reproductive success, protecting colonies from people and predators, manipulating vegetation and habitat, and actively attracting least terns with decoys. In New Jersey we monitored the number of breeding pairs and number of active colony sites for 10 years. The high count in 1976 reflected a count from a helicopter census and may be unusually high because non-breeding birds were counted.
Palacios and Mellink 1996	California	On the basis of the numbers of nesting pairs at each colony in 1992-1994, the breeding population of the Least Tern in the part of the Gulf of California that was surveyed was estimated to be about 400 breeding pairs. Disturbance by off-road vehicles is the main problem for the Least Tern in the Gulf of California. 	Between 1985 and 1994 all known and potential nesting sites of the Least Tern along the Gulf coasts of Baja California,Baja California Sur and Sonora, Mexico, were surveyed with especially intensive field work in 1992. Twenty-nine nesting sites were found; of which some might constitute clusters of alternative sites.
Parnell et al. 1997	South coast	 Table 2. Changes in site types and numbers of occupied sites for colonially nesting birds, 1977 to 1995. Year: 1977 / 1983 / 1988 / 1993 / 1995 Barrier beach: 24 / 20 / 17 / 22 / 26 Natural estuarine: 42 / 22 / 25 / 36 / 34 Dredged undiked: 56 / 21 / 16 / 18 / 19 Dredged diked: 22 / 18 / 8 / 5 / 6 Other: 0 / 0 / 0 / 1 / 3 Total sites: 124 / 81 / 64 / 82 / 88 Colony turnover was highly variable among species (Table 3). Site fidelity was generally higher for wading birds than for terns and skimmers.	This paper summarizes information and assesses changes in colonially nesting bird populations from information gleaned from the 1919 and 1942 editions of Birds of North Carolina (Pearson et al. 1919, 1942), from Audubon Warden reports, from partial surveys by Quay, Parnell, Soots and others between 1942 and 1995, and from data gathered during 5 coastwide surveys conducted since 1977. Least Terns were apparently abundant in North Carolina in earlier times, but numbers were greatly reduced by plume hunters. Pearson et al. (1919) reported them to be rare at the turn of the century, but by 1906 they recorded the presence of 700 young in Pamlico Sound. In 1939, a systematic search of the coastal beaches located 23 colonies and an estimated 25,000 adults (Pearson et al. 1942). In 1955, Funderburg and Quay (1959) reported 13 colonies containing 512 nests along a 95 mile stretch of beach in southeastern North Carolina between New River Inlet and New Inlet. Since 1977, peak nest counts have varied between 1,528 and 2,154. In 1993, the first roof nesting colony was found at New Bern. In 1995, that colony was still active, and colonies were found on roofs at Nags Head and Wilmington. It is very possible that there are additional roof nesting colonies, as our surveys did not cover all coastal shopping centers. Mean annual turnover (16%) was lower than reported from Louisiana (Visser and Peterson 1994) and Massachusetts (Erwin 1978).
Kirsch and Sidle 1999	Central US	Population trends were estimated for 31 local areas surveyed ≥3 years. Five areas had significant positive trend, and 2 areas had significant negative trend; no trends were detected for the remaining 24 local areas. Trend for the entire population was positive (λ = 1.090, 95% CI = 1.056-1.111, n = 31 areas). However, when data from the Lower Mississippi River were excluded, trend was positive but the 95% confidence interval	We estimated population trends from surveys conducted during 1986-95 at 3 spatial scales: (1) "local areas," which included river reaches, reservoirs, and distinct salt flat areas; (2) "drainage area," which included the upper Missouri River, Platte River, and Lower Mississippi River basins; and (3) the entire population. We defined trend as the finite rate of increase (λ), where λ = e^r and r is the slope of the regression of the natural log of least tern

Citation	Location	Observed Trend	Study Design
		included 1 (λ = 1.024, 95% CI = 0.998-1.045, n = 30 areas). Minimum fledging success needed to support observed population trends was 1.12 fledglings/pair for the Lower Mississippi River drainage, 0.52 for the Platte River drainage, and 0.59 for the upper Missouri River drainage. For the interior population as a whole, fledging success of 0.899 was needed to support the trend (λ = 1.090) based on all the, data, whereas fledging success of 0.601 was needed to support the trend (λ = 1.024) when data from the Lower Mississippi River were deleted. The most plausible explanation for the recent increase of the interior population of least tern is surges of immigration from the least tern population along the coast of the Gulf of Mexico. Site fidelity of interior least terns varies among areas and has been estimated as 29% on the central Platte (Lingle 1993a), 80% at sites in Kansas and Oklahoma (Boyd 1993), and 42% on the Lower Mississippi River (Renken and Smith 1995b). Natal philopatry has not been reliably estimated, because of the low resighting rates of least terns banded as chicks (1% [Renken and Smith 1995b], 1.3% [Boyd 1993], 5% Lingle 1993a). 	numbers on year.
Schuetz 2011	California	While the breeding population grew from approximately 1,253 to 7,241 pairs (578%) during the study period (1988–2009) both clutch size and fledgling productivity declined. Clutch size decreased by approximately 0.27 eggs (14%) from 1990–2004 then showed a moderate increase of 0.11 eggs from 2004–2009. Estimates of fledgling productivity showed a similar pattern of decline and moderate increase even after controlling for clutch size. Latitude was a significant predictor of productivity (Figure 6D, p,0.0001) with northern birds fledging young more efficiently than southern birds, even while controlling for larger clutches in the north. 	I assembled data on California least tern population size, clutch size, and fledgling production at 27 breeding sites throughout California using California Department of Fish and Game (CDFG) breeding season surveys (1988-2009).
Aron 2005	Upper Missouri River		USCOE Monitoring Program
Elliott et al. 2007	California	The estimated number of breeding pairs at Alameda Point increased from ten pairs in 1976 to 379 pairs in 2004. The average rate of growth was 10.4% per year (B = 0.0986, SE = 0.014, P < 0.001; Fig. 2).  Figure 2. Estimated population size of the Alameda Point Least Tern colony from 1976-2004. (* = El Niño years. Years of important site changes are noted: 1981 = electric fence was erected; 1997 = Naval Air Station closure; 2004 = chain link fence erected.)	The objectives herein are 1) to describe the breeding phenology and success of Least Terns at Alameda Point and 2) to evaluate the status of this northern population. Discussion on the factors that may be limiting the growth of this population is also presented. Through contracts with the U.S. Navy and USFWS, the colony was managed and protected by the Golden Gate Audubon Society (1979-1999), PRBO Conservation Science (2000-2001) and USFWS (2002-2004). Golden Gate Audubon Society personnel monitored the colony almost exclusively from outside the site and occasionally entered the colony to collect dead nestlings, non-viable eggs, and dropped prey (Collins 2000). However, PRBO and USFWS biologists entered the site on a regular basis to collect the necessary data. Estimates on population size, breeding success, and dropped prey composition were compared for all years (1976-2004). The nesting area is located on the runway complex of the former Naval Air Station, Alameda. The breeding site (hereafter referred to as "the site") was enclosed in 1981 by a 0.91-m high electric fence with hardware cloth near the bottom that acted as a chick barrier. The fence was designed and erected to reduce access by mammalian predators and perching by avian predators. The fence lost its charge in 2001, but still provided an effective barrier to ground-based predators. In April 2004, the U.S. Fish and Wildlife Service (USFWS) replaced the derelict electric fence with a 1.2-m high chain link fence, and the fence was shifted north 20 m to avoid a storm water drain and a highly vegetated portion of the original tern colony. With the new fence in 2004, the site was expanded from 2.4 to 3.9 ha. The substrate within the enclosure consists of asphalt covered with soil, pea gravel, sand, and oyster shells, all of which were added to improve the breeding habitat. Cylindrical clay tiles, wooden A-frame structures, and driftwood were randomly placed in the site to provide shelter and protection for nestlings. Cinder blocks were also placed in a grid system within the site to form approximately 20 m × 20 m grid cells to aid in the mapping of nest locations.
Sherfy et al. 2008	Upper Missouri River		In 2006 and 2007, we located and monitored least tern nests and chicks on 3 created sandbar complexes and 5 natural complexes on the Gavins Point Reach (hereafter, the "focal sandbars"). In 2007, we also began monitoring 2 newly created sandbars and 1 natural sandbar on Lewis and Clark Lake. We monitored nests on 17 4-mile river segments on the Garrison Reach.

Citation	Location	Observed Trend	Study Design
Kirsch 1996	Platte River	Although sandbars and sandpits appeared equally suitable and terns did not prefer either habitat, local productivity will not support this population unless annual postfledging survival is higher than current estimates for the species. Population trend estimated with fledglings per pair = 0.50 was negative for all but the highest (ca 0.90) rates of annual postfledging survival. Furthermore, deterministic models like the one used in this study overestimate trend. Table 22. Estimated population trend (*r*) for model least tern populations with 0.50 fledglings per pair, 6 levels of survival from fledging to 2 years of age, and 6 levels of annual adult survival. Survival from fledging to 2 yrs Adult survival: 0.30, 0.40, 0.50, 0.65, 0.70, 0.80 0.70: −0.260, −0.225, −0.191, −0.166, −0.143, −0.120 0.75: −0.209, −0.175, −0.142, −0.119, −0.096, −0.074 0.80: −0.161, −0.128, −0.096, −0.074, −0.052, −0.031 0.85: −0.116, −0.083, −0.053, −0.031, −0.010, 0.010 0.90: −0.072, −0.041, −0.011, 0.010, 0.030, 0.050 0.95: −0.003, 0.000, 0.029, 0.049, 0.069, 0.088	Available habitats of both types were characterized and quantified using aerial videography (1989-90), and habitat use was assessed from census data (1987-90). Productivity of adults and causes and correlates of egg and chick mortality were estimated (1987-90). Population trend was assessed with a deterministic model using my estimates of productivity and a range of survival estimates for Laridae reported in the literature.

Genetics

Citation	Location	Findings
Massey 1998	General	Here I briefly summarize evidence supporting the separation of Least and Little Terns and make a case for considering all populations of Least Terns on the west coast of the Americas as one subspecies (browni) until and unless field studies indicate otherwise. Geographic isolation, along with measurable morphological and/or vocal differences is considered a necessary, if not a sufficient, condition for the evolution of subspecies of terns. Morphologic differences used to separate *mexicanus* and *staehleri* from *browni* do not stand up to scrutiny; the single-sentence descriptions, small sample sizes, and casual measurements would not be acceptable today.
Thompson et al. 1992	Texas	Analyses of seven bill, leg, wing, and plumage characters measured from 267 museum specimens of adult Least Terns revealed significant morphological differences between sexes but not among three subspecies (*antillarum*, *athalassos*, and *browni*). While individual morphometric characters sometimes did not overlap between a pair of subspecies, discriminant analysis employing criteria developed from the complete morphometric data set correctly classified >90% of *antillarum* specimens but misclassified 39% and 51% of *athalassos* and *browni* specimens, respectively. Cluster analysis did not segregate specimens into groups consistent with current subspecific taxonomy. Electrophoretic variation in proteins encoded by 50 loci revealed no genetic distinctions between S. a. *antillarum* and S. a. *athalassos* for 22 specimens from four breeding sites on the Texas coast, Rio Grande, and Texas panhandle rivers. Morphometric analyses presented here seriously question existence of distinctive differences previously proposed in describing *antillurum*, *athalassos*, and *browni* subspecies. These analyses indicated that *antillarum* largely encompasses the range of measures inherent in North American Least Tern populations for variables examined. This similarity is consistent with Massey's (1976) suggestion that *browni* is indistinguishable behaviorally, vocally, and morphometrically from *antillarum* and Boyd and Thompson's (1985) suggestion that mixing of *antillarum* and *athalassos* m ay be more common than previously thought. Research reported here and by Massey (1976) indicated no practical means for distinction among the North American "subspecies" of Least Terns.
Whittier et al. 2006	North America	DNA sequence variation from two nuclear introns and part of the mitochondrial cytochrome *b* gene were used to evaluate population structure among three subspecies of Least Tern that nest in the United States (California [*Sterna antillarum browni*], Interior [S. a. *athalassos*], Eastern [S. a. *antillarum*]). Genetic indices for mitochondrial DNA did not differ among subspecies, and gene flows (reflecting female dispersal) ranged from 10 to 83 individuals/generation. Reservations are expressed about the validity of the current subspecific divisions and further research is required, including their taxonomic relationship to the Little Tern (*Sterna albifrons*). Only two definitive measurements appear to distinguish all three subspecies of Least Terns in North America: breeding location (albeit vague where Interior and Eastern subspecies likely overlap) and feather colorimetry (Johnson et al. 1998), which may be potentially confounded by environmental conditions during feather development. Because Least Tern subspecies cannot yet be differentiated consistently based on genetics, morphology, or behavior (Massey 1976; Thompson et al. 1992), we are dubious of their current subspecific classification.

Detection Probabilities

Reference	Location	Findings
Shaffer et al. 2013	Upper Missouri River	The practice of recording nests where no nest was found (but a brood was later found) introduces a bias into the data because detection of the nest was dependent on its fate (that is, if the nest had failed, it never would have entered the TPDMS because no brood would have been seen). Guidance documents failed to acknowledged that the many sources of measurement uncertainty for terns and plovers (for example, movements, double-counting, nondetection) make obtaining a true census extremely difficult. Least tern and piping plover chicks exhibit changes in their size and plumage during development that likely lead to differences in age-specific detectability (that is, they become less cryptic as they develop). Additionally, compared to piping plover chicks, least tern chicks are relatively inactive prior to fledging and hide among vegetation or other physical structure at older ages; however, just prior to fledging age, they become more active and are more easily detected because they spend more time on sandbar shorelines. Behavioral changes are less pronounced in piping plover chicks because they are precocial; however, physical features (for example, vegetation, rocky substrate) can make chick detection problematic. In situations where detection bias was known or suspected to be an issue (for example, counts of fledging-age chicks) and we were able to estimate detection probability, we computed the ratio of the expanded total divided by the estimated detection probability to account for the imperfect detection. We recognized that our counts of nests were subject to imperfect detection and we quantified that by estimating nest detection probability. Under the assumption that daily nest detection probability (DNDP) was homogeneous among and within nests, we estimated DNDP, conditional on a nest being active, as the number of nests found divided by the cumulative number of opportunities to find those nests. Number of opportunities to find a particular nest was determined from the number of times the habitat unit (for example, sandbar) was nest-searched subsequent to the nest being initiated and before being found. For example, a nest initiated on day 120 and discovered on day 130 on a sandbar that had been searched on day 123 and 127 would be credited with three opportunities for discovering that nest. We estimated detection probabilities for the six possible combinations of species by study area: Piping plovers at GRR (n=354), piping plovers at GVP (n=713), piping plovers at SAK (n=251), least terns at GRR (n=335), least terns at GVP (n=986), and least terns at Lewis and Clark Lake (n=1314). Although Lewis and Clark Lake was not a focus of the evaluation, we included chick detection data from there (2007–08), from GVP (2006–07) and from SAK (2006, 2009) to bolster sample size for estimating *p*.

		Table 5. Daily detection probabilities for least tern and piping plover nests on three Missouri River study areas, 2006–09, along with probability that a nest would be detected in three and seven visits. [LETE, least tern; PIPL, piping plover; GRR, Garrison River Reach; GVP, Gavins Point River Reach; SAK, Lake Sakakawea] **Table 26.** Average number of attempts to relocate a fledging-age chick (14–18 days for terns, 18–25 days for plovers), and the estimated detection probability from simulation by species and study area. [Detection probability values and confidence intervals interpolated from table 25. LETE, least tern; PIPL, piping plover; GRR, Garrison River Reach; GVP, Gavins Point River Reach; SAK, Lake Sakakawea; NA, not applicable]
Roche et al. 2014	Upper Missouri River	We individually color-marked, recaptured, and re-sighted 1,635 interior least tern chicks and 1,318 piping plover chicks from 2006 to 2009 at 4 study areas along the Missouri River and investigated effects of observer-, subject-, and site-level covariates suspected of influencing detection. Increasing the time spent searching and crew size increased the probability of detecting both species regardless of study area and detection methods were not associated with decreased survival. However, associations between detection probability and the investigated covariates were highly variable by study area and species combinations, indicating that a universal mark-recapture design may not be appropriate. We conducted mark-recapture of piping plovers at SAK in 2006–2009, at 37 sandbars at GRR in 2007, and at 31 sandbars at GVP in 2008–2009. Mark-recapture of least terns occurred at 37 sandbars at GRR in 2006–2007, at 31 sandbars at GVP in 2006–2009, and at 4 sandbars at LCL in 2007–2008. Following chick banding, we revisited each sandbar or shoreline site every 2–3 days and resighted or recaptured chicks. Searches were conducted systematically in a grid formation. The location of vegetation on a sandbar (or lakeshore) along with the shape of the sandbar itself, can have serious consequences on the ease at which resighting or recapture occurs, with large sandbars punctuated by several discrete vegetated areas offering some of the most logistically difficult conditions, and narrow unvegetated relatively small sandbars some of the easiest. Given that 1) increasing search times is probably more cost effective than adding additional team members to a crew, and 2) increasing search times increased the probability of detection for both species at all study areas, the most efficient means by which to increase overall daily detection probability would be to increase the length of site visits.

Species	Study area	Year	Number of opportunities	Number of detections	Daily detection probability	95-percent confidence interval	Three visits	Seven visits
LETE	GRR	2006	241	117	0.49	0.42–0.55	0.86	0.99
LETE	GRR	2007	227	107	0.47	0.41–0.54	0.85	0.99
LETE	GVP	2008	436	194	0.44	0.40–0.49	0.83	0.98
LETE	GVP	2009	290	136	0.47	0.41–0.53	0.85	0.99
PIPL	GRR	2006	366	237	0.65	0.60–0.70	0.96	1.00
PIPL	GRR	2007	411	205	0.50	0.45–0.55	0.87	0.99
PIPL	GVP	2008	572	237	0.41	0.37–0.45	0.80	0.98
PIPL	GVP	2009	314	194	0.62	0.56–0.67	0.94	1.00
PIPL	SAK	2007	202	107	0.53	0.46–0.60	0.90	0.99
PIPL	SAK	2008	167	86	0.51	0.44–0.59	0.89	0.99

Species	Study area	Year	Attempts	Detection probability	95-percent confidence interval
LETE	GRR	2006	2.33	0.54	0.47–0.61
LETE	GRR	2007	1.97	0.50	0.43–0.57
LETE	GVP	2008	2.46	0.49	0.46–0.53
LETE	GVP	2009	1.98	0.42	0.39–0.46
PIPL	GRR	2006	NA	NA	NA
PIPL	GRR	2007	2.54	0.86	0.84–0.89
PIPL	GVP	2008	2.45	0.70	0.67–0.73
PIPL	GVP	2009	4.54	0.80	0.78–0.82
PIPL	SAK	2007	3.38	0.99	0.99–0.99
PIPL	SAK	2008	3.41	0.99	0.99–0.99

Not Used

Category 1	Citation	Location
NA	Hillman et al. 2013	South coast
NA	McGowan 2013	General
NA	Sherfy et al. 2011	Upper Missouri River

References by Category

Reproductive Success

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Appendix C

Model Parameters from Literature Review

Appendix C

Model Parameters from Literature Review

Plover Parameters

Reference	Plovers	Adult Survival	Juvenile Survival (survival of hatchlings through their first winter)	Intercept	Slope (density)	Power Peaking	K	Detectability (adults)	Detectability (fledglings)	Inter-Reach Dispersal (1st yr)	Dispersal Adult	Dispersal 1st Year	Nest success	Predation
Buenau et al. 2014	Nominal Value	0.8	0.6	0.54	-0.31	0	0.64	0.7	0.95	0.4	0			
	Estimation CV	3.2%	9%	17%	24%		22%			25%				
	Temporal CV	1%	1%	5%	1%		1%			1%				
USACE 2013	Nominal Value	0.783	0.6	0.54 and 1.21	-0.31 and -1.11	-0.75	0.64	0.7	0.95	0.4	0			
	Estimation CV	9.6%	9%	17% and 19%	24% and 23%	75%	22%			25%				
	Temporal CV	1%	1%	5%	1%	1%	1%			1%				
Catlin (2009) Thesis	Nominal Value	82%	78%								0.055	0.246	58%	30%
Upper Missouri River	N =	357	685								357		599	599
	Estimation CV	17.13%	383%								59%	521%		
	Number of Years	2	1, but two strata								2		3	3
	Temporal CV	0.17%	3%								139%	22%	5%	9%
Roche et al. 2010c	Nominal Value	0.8												
General	N =	451												
	Estimation CV	66%												
	Number of Years													
	Temporal CV													
Gaines and Ryan 1988	Nominal Value	63%		1.26	0								42%	
North Dakota	N =	64		137									150	
	Estimation CV													

Reference	Plovers	Adult Survival	Juvenile Survival (survival of hatchlings through their first winter)	Intercept	Slope (density)	Power Peaking	K	Detectability (adults)	Detectability (fledglings)	Inter-Reach Dispersal (1st yr)	Dispersal Adult	Dispersal 1st Year	Nest success	Predation
	Number of Years	2		2									2	
	Temporal CV			25%									2%	
Larson et al. 2000	Nominal Value	74%	32%											
North Dakota	N =	204	143											
	Estimation CV	100%	24%											
	Number of Years	8	8											
	Temporal CV	29%												
Anteau et al. 2012a	Nominal Value												20%	16%
Lake Sakakawea	N =												346	346
	Estimation CV												303%	
	Number of Years												4	
	Temporal CV												57%	
Shaffer et al. 2013	Nominal Value								87%				38%	
Upper Missouri River	N =								5					
	Estimation CV								14%				14%	
	Number of Years								3				4	
	Temporal CV								7%				30%	
Catlin et al. 2011b	Nominal Value												45%	
Upper Missouri River	N =												9	
	Estimation CV												41%	
	Number of Years												3	
	Temporal CV												12%	
Felio et al. 2010	Nominal Value												53%	14%
Upper Missouri River	N =												294	

Reference	Plovers	Adult Survival	Juvenile Survival (survival of hatchlings through their first winter)	Intercept	Slope (density)	Power Peaking	K	Detectability (adults)	Detectability (fledglings)	Inter-Reach Dispersal (1st yr)	Dispersal Adult	Dispersal 1st Year	Nest success	Predation
Results are preliminary.	Estimation CV													
	Number of Years												1	
	Temporal CV													
Ivan and Murphy 2005	Nominal Value												32%	
North Dakota	N =												310	
	Estimation CV												20%	
	Number of Years												9	
	Temporal CV													
Sherfy et al. 2008	Nominal Value												52%	
Missouri River	N =												207	
	Estimation CV													
	Number of Years												1	
	Temporal CV													
Sherfy et al. 2009a	Nominal Value												45%	
Missouri River	N =												237	
	Estimation CV													
	Number of Years												1	
	Temporal CV													
Sherfy et al. 2009b	Nominal Value												37%	
Missouri River	N =												194	
	Estimation CV													
	Number of Years												1	
	Temporal CV													
Cohen et al. 2006	Nominal Value	70%	12%											
East Coast	N =	71	43											

Reference	Plovers	Adult Survival	Juvenile Survival (survival of hatchlings through their first winter)	Intercept	Slope (density)	Power Peaking	K	Detectability (adults)	Detectability (fledglings)	Inter-Reach Dispersal (1st yr)	Dispersal Adult	Dispersal 1st Year	Nest success	Predation
	Estimation CV	37%	278%											
	Number of Years	3	2											
	Temporal CV	8%	12%											
Cohen and Gratto-Trevor 2011	Nominal Value	80%	57%											
Great Plains Canada	N =	782	182											
	Estimation CV	70%	118%											
	Number of Years	7	7											
	Temporal CV													
Ledee et al. 2010	Nominal Value	76%												
Great Lakes	N =	150												
	Estimation CV	7%												
	Number of Years	9												
	Temporal CV													
Root et al. 1992	Nominal Value	66%												
North Dakota	N =	352												
	Estimation CV	161%												
	Number of Years	5												
	Temporal CV	23%												
McGowan et al. 2014	Nominal Value	78%	52%											
Great Plains	N =	668	2053											
cites unpublished data from D. Catlin	Estimation CV	4%	23%											
	Number of Years	8												
see Davis et al. 2013 Field Report	Temporal CV	9%												

Not Used for Survival	Reason	Not Used for Nest Success	Reason	Not Used for Detection Probabilities	Reason
Drake et al. 2001	Study conducted on the Gulf coast.	Aron 2005	Concern about data quality (Shaffer et al. 2013)	Roche et al. 2014	Uses data from Shaffer et al. 2013
Plissner and Haig 2000a	Cites other authors	Barber et al. 2010	Study conducted in Northeastern Canada		
Roche et al. 2010b	Study on mortality associated to banding chicks.	Cohen et al. 2009	Time series of nest survival at two locations on the east coast.		
Ryan et al. 1993	Cites other authors	Doherty and Heath 2011	Study conducted in the east coast.		
Aron 2005	Cites other authors	Donlan et al. 2003	Study conducted in the east coast.		
Catlin et al. 2011a	Study estimates daily survival rates.	Espie et al. 1996	Study conducted in the east coast.		
Le Fer et al. 2008a	Study estimates daily survival rates.	Espie et al. 1998	Study conducted in the Great Plains of Canada greater than 20 years ago.		
Murphy et al. 2003b	Study estimates mortality associated with cages.	Goldin and Regosin 1998	Study conducted in the east coast greater than 20 years ago on 17 pairs of plovers.		
Roche et al. 2010a	Decreasing adult survival through time (.7 to 0.85) for birds in the Great Lakes.	Greenwald 2009	Study conducted in the east coast.		
Roche et al. 2008	Study evaluates survival through time for birds in the Great Lakes	Johnson et al. 1997	Cites other authors		
Sidle et al. 1992	Study evaluates nest survival.	Larson et al. 2002	Study evaluates nest exclosures		
Barber et al. 2010	Study estimates mortality associated with cages.	Maslo and Lockwood 2009	Study conducted in the east coast.		
Catlin et al. 2011b	Study evaluates nest survival.	Maxson and Haws 2000	Study conducted in Minnesota.		
Elias et al. 2000	Study estimated survival from hatching to fledging.	McGowan et al. 2007	Study evaluates 4 nests.		
Wemmer et al. 2001	Cites other authors	Murphy et al. 2003a	Study evaluates nest exclosures		
Cohen et al. 2009	Study estimated survival from hatching to fledging.	Patterson et al. 1991	Study conducted on the Atlantic Coast.		
Johnson et al. 1997	Cites other authors	Rimmer and Deblinger 1990	Study conducted in the east coast.		
Knetter et al. 2002	Study estimates the number of fledglings/pair	Roche et al. 2010a	Study evaluated nest abandonment.		
Kruse et al. 2001	Study conducted more than 20 years ago.	Roche et al. 2008	Study conducted in the Great Lakes comparing wild to captive chicks.		
Hunt et al. 2013	Study on mortality associated to handling chicks.	Roche et al. 2012	Study conducted in the Great Plains of Canada		
Brudney et al. 2013	Study evaluates survival by age for birds in the Great Lakes	Saunders et al. 2012	Study conducted in the Great Lakes.		
Dirks 1990	Study conducted more than 20 years ago.	Tremblay et al. 1994	Study conducted in Northeastern Canada		
--		Wemmer et al. 2001	Study conducted in the Great Lakes.		
		Wiens and Cuthbert 1988	Study conducted in Minnesota greater than 20 years ago.		

Tern Parameters

Reference	Terns	Fledgling survival through first winter	Juvenile survival through second winter	Young adult survival	Adult survival	Intercept	Slope (density)	Power peaking	K	Detectability		Dispersal			Nest success = hatched at least one chick	Predation
										Adults	Fledglings	Inter-reach (2nd yr)	Adult	After 1st Year		
Buenau et al. 2014	Nominal Value	0.4	0.4	0.81	0.92	0.15	-0.04	0	0.99	0.7	0.95	0.4				
	Estimation CV	9	9	9	6	57	54		22			25				
	Temporal CV	1	1	1	1	5	1		1			1				
USACE 2013	Nominal Value	0.4	0.4	0.81	0.92	0.15 and 0.26	-0.04 and -0.15	-0.75	0.99	0.7	0.95	0.4				
	Estimation CV	9	9	9	6	57 and 64	54 and 45	75	22			25				
	Temporal CV	1	1	1	1	5	1	1	1			1				
Massey et al. 1992	Nominal Value	0.4	0.4	0.81	0.92											
California	N =	154	154	154	186											
	Estimation CV	47%	47%	57%	24%											
	Number of Years	8	8	8	8											
	Temporal CV															
Alcakaya et al. 2003	Nominal Value	0.5627	0.5627		0.8867	1.599	0					0.395				
California	N =	154	154		186	154										
	Estimation CV											41%				
	Number of Years	3	3		3	3			3							
	Temporal CV	6%	6%		58%	14%			6%							
Atwood and Massey 1988	Nominal Value												0.41			
	N =												3			
	Estimation CV												30%			
	Number of Years												4			
	Temporal CV												39%			
Minsky 1984	Nominal Value								12 %							
California	N =								4							
cited in Alcakaya et al. 2003	Estimation CV															
	Number of Years															
	Temporal CV															

Reference	Terns	Fledgling survival through first winter	Juvenile survival through second winter	Young adult survival	Adult survival	Intercept	Slope (density)	Power peaking	K	Detectability		Dispersal			Nest success = hatched at least one chick	Predation
										Adults	Fledglings	Inter-reach (2nd yr)	Adult	After 1st Year		
Kirsch 1996	Nominal Value														54%	
Platte River	N =														501	
	Estimation CV														167%	
	Number of Years														4	
	Temporal CV														22%	
Shaffer et al. 2013	Nominal Value										0.4875				71%	
Upper Missouri River	N =										1321				4	
	Estimation CV										0.22				23%	
	Number of Years										4					
	Temporal CV										10%					
Renken and Smith 1995a	Nominal Value				85%											
Mississippi River	N =				194											
	Estimation CV				93%											
	Number of Years				5											
	Temporal CV															
Renken and Smith 1995b	Nominal Value											0.98864994	3%			
Mississippi River	N =											1674	194			
	Estimation CV															
	Number of Years															
	Temporal CV															
Carreker 1985	Nominal Value											30%		50%		
California	N =											190		190		
	Estimation CV															
	Number of Years															
	Temporal CV															
Aron 2005	Nominal Value														50%	11%
Upper Missouri River	N =														4	
	Estimation CV														22%	

Reference	Terns	Fledgling survival through first winter	Juvenile survival through second winter	Young adult survival	Adult survival	Intercept	Slope (density)	Power peaking	K	Detectability		Dispersal			Nest success = hatched at least one chick	Predation
										Adults	Fledglings	Inter-reach (2nd yr)	Adult	After 1st Year		
	Number of Years															
	Temporal CV															
Brooks et al. 2013	Nominal Value															47%
South Coast	N =															257
	Estimation CV															
	Number of Years															2
	Temporal CV															
Jenniges and Plettner 2008	Nominal Value														63%	
Platte River	N =														647	
	Estimation CV															
	Number of Years															
	Temporal CV															
Kruse et al. 2001	Nominal Value														49%	23%
Upper Missouri River	N =														354	354
	Estimation CV															
	Number of Years														2	
	Temporal CV															
Sherfy et al. 2008	Nominal Value														79%	
Upper Missouri River	N =														236	
	Estimation CV															
	Number of Years															
	Temporal CV															
Sherfy et al. 2009b	Nominal Value														71%	
Upper Missouri River	N =														136	
	Estimation CV															
	Number of Years															
	Temporal CV															

Reference	Terns	Fledgling survival through first winter	Juvenile survival through second winter	Young adult survival	Adult survival	Intercept	Slope (density)	Power peaking	K	Detectability		Dispersal			Nest success = hatched at least one chick	Predation
										Adults	Fledglings	Inter-reach (2nd yr)	Adult	After 1st Year		
Smith and Renken 1993	Nominal Value														65%	
Mississippi River	N =															
	Estimation CV															
	Number of Years														3	
	Temporal CV															
Szell and Woodrey 2003	Nominal Value														60%	
Mississippi River	N =														3467	
	Estimation CV															
	Number of Years														3	
	Temporal CV														54%	
Stucker et al. 2013	Nominal Value														37% Natural; 70% Constructed	
Upper Missouri River	N =														824	
	Estimation CV															
	Number of Years														3	
	Temporal CV														a function of % Pebble	

Not Used for Survival	Reason	Not Used for Nest Success	Reason	Not Used for Detection Probabilities	Reason
Whittier and Leslie 2009	Estimated chick survival to fledge	Brooks et al. 2013	Estimates ranged from 0% to 93%	Roche et al. 2014	Uses data from Shaffer et al. 2013
Aron 2005	Only cites other studies	Brunton 1997	Evaluated hatching success based on location in colony and nest abandonment on east coast		
Bailey and Servello 2008	Estimated chick survival to fledge	Conway et al. 2003	Study conducted in Texas based on 20 nests		
Dugger et al. 2000	Estimated chick survival to fledge	Elliott et al. 2007	Estimated the fate of eggs not nests		
Lombard et al. 2010	Estimated chick survival to fledge	Koenen et al. 1996	Estimates ranged from 23% to 100% for terns nesting on ridges created from existing substrate in OK		
Sidle et al. 1992	Study conducted more than 20 years ago.	Leslie et al. 2000	Study conducted more than 20 years ago in OK.		
Brooks et al. 2013	Estimated chick survival to fledge	Lombard et al. 2010	Study conducted in the Virgin Islands		
Kirsch 1996	Study conducted more than 20 years ago.	Massey and Atwood 1981	Study conducted more than 20 years ago.		
--		Sherfy et al. 2009a	Used more current study results from Sherfy et al. 2009b		
		Sidle and Harrison 1990	Study conducted more than 20 years ago		
		Zuria and Mellink 2002	Study conducted in Mexico		

Appendix D

Plover Model Review

Appendix D

Plover Model Review

D.1 Ryan et al. 1993

Authors: Mark Ryan, Brian Root, Paul Mayer
Year: 1993
Title: Status of Piping Plovers in the Great Plains of North America: a demographic simulation model
Citation: Conservation Biology 7(3): 581−585
Affiliation: University of Missouri-Columbia
Model type/structure: Stochastic demographic simulation with productivity and adult and chick survival modeled as beta distributions; models females only.
Region/population: Great Plains piping plovers
Objective/focal mechanisms: Population persistence; what changes in vital rates are needed for stability or recovery?
Population structure, spatial resolution and scale: One population, entire Great Plains
Temporal resolution and scale: Annual, 100 years
Habitat: Not included
Management actions: Not included; nest protection and habitat enhancement mentioned as means of increasing survival.
Data sources: Adult survival from Root et al 1992; juvenile survival 70-100% adult survival based on other spp. due to lack of data; mean reproductive rate from other studies in all breeding habitats
Metrics and analyses used: Time to extirpation or recovery; stability and rate of growth/decline
Validation: Compared to regional plover count data from 1986-1990
Results: All simulations indicate that the Great Plains plover population is undergoing a substantial decline (7.6% for "most realistic" projection). Most realistic projections indicate extirpation in about 80 years. Immature survival of 0.6 (90% adult survival) best matched population records. 1.13 chicks fledged per pair, or 8.7% increase in both survival rates, was necessary for stable population. 1% or 2% increase resulted in recovery to 2,550 pairs in 51 or 30 years, respectively.
Conclusions: Populations are expected to decline. The decline might be reversible by improving productivity or survival; delays in management will reduce likelihood of recovery
Caveats mentioned: None
Implications for EA: Basic PVA that several other models are based upon; not directly applicable to EA.

D.2 Plissner and Haig 2000

Authors: Jonathan Plissner, Susan Haig
Year: 2000
Title: Viability of piping plover *Charadrius melodus* metapopulations
Citation: Biological Conservation 92: 163-173
Affiliation: USGS
Model type/structure: VORTEX stochastic simulation package
Region/population: Atlantic Coast and Great Lakes/Great Plains populations, modeled separately
Objective/focal mechanisms: Comparison between three breeding populations; effect of spatial distributions upon population viability (single population vs. metapopulation, not spatially explicit).

Population structure, spatial resolution and scale: Atlantic Coast with four subpopulations (recovery units from 1996 recovery plan); Great Lakes/Great Plains with five subpopulations
Temporal resolution and scale: Annual, 100 years
Habitat: Not included
Management actions: Not included
Data sources: Initial population size from 1996 international census; productivity from annual Atlantic Coast productivity estimates (1.37, CV 0.4) and local studies for Great Lakes/Great Plains (1.25, CV 0.5). Productivity depends upon region. Baseline adult mortality rates from Ryan et al. (1993). First-year mortality from Cape Cod survivorship, lower than Ryan et al (1993); 51.64 for Atlantic Coast, 56.8 for Great Plains (note this is mortality, not survival). Dispersal of 0.01 birds per year between adjacent populations, except 0.02 between rivers and adjacent wetlands. Juvenile dispersal assumed same as adults due to lack of data.
Metrics and analyses used: Sensitivity analysis; population persistence; comparison of metapopulation to panmictic models.
Validation: Validation against Atlantic Coast population trends between 1991 and 1996 (international census). The model underpredicts population trends.
Results: Great Lakes/Great Plains populations would be unlikely to persist given current demographic rates. All Atlantic Coast populations have >95% likelihood of persistence for 100 years, though most subpopulations decrease. The Great Lakes/Great Plains region would need to increase mean reproductive success to 1.7 to remain viable, and even then it would not be stable. It needs a 2.0 fledge ratio to be stable. Lower reproductive variability improves stability. The Great Lakes/Great Plains model is highly sensitive to survivorship. Connectivity in the Atlantic Coast metapopulation increased overall population size and most subpopulations; higher dispersal in Great Lakes/Great Plains populations decreased metapopulation viability. Atlantic Coast panmictic model fared better than the metapopulation model; Great Lakes/Great Plains populations did poorly with both structures. The model underpredicts the Atlantic Coast population.
Conclusions: Metapopulation models are more pessimistic than single population models because of smaller subpopulations and source/sink dynamics, but maintaining small populations is still worthwhile. Population-specific rates of survivorship and dispersal are needed to improve accuracy of predictions.
Caveats mentioned: Does not include density dependence, inbreeding depression, or catastrophic events (though variability in years was included in demographic data collection)
Implications for EA: Great Lakes/Great Plains population unlikely to persist with these vital rates; metapopulation dynamics decreased persistence for these populations. Atlantic Coast population persistence dependent upon increase in reproductive rates due to nest protection. Model does not address habitat or population density so limited application to EA.

D.3 Wemmer et al. 2001

Authors: Lauren C. Wemmer, Uygar Ozesmi, Francesca J. Cuthbert
Year: 2001
Title: A habitat-based population model for the Great Lakes population of the piping plover (*Charadrius melodus*)
Citation: Biological Conservation 99: 169−181
Affiliation: University of Minnesota
Model type/structure: Stochastic discrete-event model, spatially explicit, habitat-based and stage-structured using "Extend" modeling software
Region/population: Great Lakes
Objective/focal mechanisms: Overall population persistence and effects of protection/restoration/land acquisition, as well as role of habitat on private property. Includes pairing process, site selection. Successful pairs return to previous sites, unsuccessful pairs may seek new breeding sites.
Population structure, spatial resolution and scale: Shoreline habitat in Michigan was based on actual availability. Breeding sites were continuous areas of beach with specific number of available territories. Breeding sites were connected to each other by flight distance, but not georeferenced.
Temporal resolution and scale: Annual, 100 years; multiple discrete events modeled per year
Habitat: Habitat quality was implicitly included in site-specific productivity rates measured over 14 years (thought to encompass range of variation.)
Management actions: Funding could be allocated to increase protection efforts, restore or purchase potential breeding habitat, or some degree of both. Actions were implemented in model by increasing minimum fledging success probability; increasing the number of breeding territories at each site; and adding sites with historic records of use but no current nesting. Also modeled only that habitat in state and federal land to see if there was enough to support 100 breeding pairs in Michigan without private lands.
Data sources: Demographic data from banding studies, though the sample size was too small to apply statistical mark-recapture techniques. Michigan Department of Natural Resources data used to determine sites, their favorability and frequency of use, and reproductive success at each site. Annual survival was 0.73 for adults and 0.31 for fledglings.
Metrics and analyses used: Cumulative persistence probability; time to extinction; final population sizes (breeding pairs)
Validation: Deterministic simulations produced model population trend that corresponded closely with observed trend from 1984−1997.
Results: All runs of the fully stochastic model resulted in extinction in 13−36 years (median 22.5). High reproductive success was necessary for a significant probability of persistence in 100 years; improved capacity and high reproductive success guaranteed persistence. Only 28% of best case scenario model runs reached target of 100 pairs in 100 years. Restoring habitat improved outcomes if reproductive success and site capacity were high. Additional sites were not helpful if reproductive success were low or medium, because success at restored sites was assumed to be lower and they acted as sinks. Removal of privately owned land markedly decreased persistence, even in best case scenario. Dispersal reduced % of non-breeders and increased persistence. The model was more sensitive to reproductive success, most sensitive to survival.
Conclusions: Current model considered a "best case" representation of the population. Consistent use of nest/chick protection at all sites could have notable population effects, but may be logistically impossible. Increasing adult survival/protecting wintering sites would also help, but even less feasible. Unknown whether plovers will nest in higher densities, so habitat conservation and

continued availability of private land important. Managers should be prepared to protect opportunistic breeding at new sites.
Caveats mentioned: Fledgling survival hard to estimate because of dispersal; not all plovers breed in their first year; site carrying capacities were hard to estimate accurately; dispersal was not well understood; models did not consider Allee, sex ratio or genetic effects.
Implications for EA: Restoring habitat can improve outcomes; acquiring additional sites that were low quality did not help because they acted as sinks; loss of privately owned land reduced persistence considerably.

D.4 Larson et al. 2002

Authors: Michael A. Larson, Mark R. Ryan, Robert K. Murphy
Year: 2002
Title: Population viability of piping plovers: effects of predator exclusion
Citation: The Journal of Wildlife Management 66(2):361−371.
Affiliation: U Missouri-Columbia, USFWS
Model type/structure: Ryan et al. 1993 model, expanded to differentiate between females nesting at alkaline wetlands and those nesting at river sites; density-independent.
Region/population: Great Plains including plovers nesting at river sites and at alkaline wetlands.
Objective/focal mechanisms: Estimate reproductive success of Great Plains plovers; revise Ryan et al 1993 to use higher survival and differentiate reproductive success between populations breeding on sandbars and on alkaline lakes; evaluate effects of predator-exclusion management.
Population structure, spatial resolution and scale: Two subpopulations: birds nesting on river sites and birds nesting on alkaline wetlands, divided by a constant proportion based on proportions observed in 1991 international breeding census.
Temporal resolution and scale: Annual, 100 years
Habitat: Included only in distinction between breeding success at river vs. alkaline sites. The population was thought to be limited more by predation than resource availability. Habitat quality was assumed constant and habitat was consistently available.
Management actions: Five levels of predation management: no protection, nest cages, permanent electric fencing, cages plus permanent fencing, and cages plus temporary electric fencing.
Data sources: Data from alkaline wetlands in ND and MT; separate mean fledging rates for 5 levels of predator management; tried to estimate true temporal variation in fledgling rates by removing sampling variation; Larson et al. (2000) adult survival 0.737; Melvin and Gibbs Atlantic Coast population for juvenile survival 0.48 (Great Plains juv. survival estimate from Larson et al. (2000) thought to be too low).
Metrics and analyses used: Estimation of instantaneous growth rate *r* for first 50 years of each run from mean or median growth rate, converted to λ (annual growth rate); sensitivity analysis by varying input parameters 10%.
Validation: Comparison with rate of decline estimated by international censuses. Calibration with 2% annual decline estimated in international censuses requires survival 4-11% higher.
Results: Baseline reproductive success was estimated as 0.89 fledglings/pair (alkaline wetlands) and 0.73 (river sites). Predator exclusion increased reproductive success. Nest cages alone and combined with temporary electric fences increased fledgling production to 1.28 and 1.78. Permanent fences resulted in fledging production of 1.15, with nest cages resulting in 2.25. Models were most sensitive to adult survival and least sensitive to fledging rates on river sites and the proportion of pairs at alkaline sites. Baseline model indicated decline of 6.2%/year. The probability of persistence for 50 years was 0.5, and for 100 years was 0.02. Current management would increase annual population growth to -4.3% and extend mean persistence time nearly 100 years. Proposed increase in management would increase the probability of persistence for 50 and 100 years to 0.78 and 0.23,

respectively. Reproductive success of 1.44 (alkaline) and 1.25 (entire Great Plains) needed to stabilize median population size.
Conclusions: The model is likely to be pessimistic because vital rates are likely pessimistic. Permanent fencing and nest cages were the most effective predator management activities; more convenient temporary fences with nest cages also increased reproductive success above population stabilization levels.
Caveats mentioned: Model has more pessimistic than optimistic biases; PVAs best used to compare management actions, not produce absolute numbers.
Implications for EA: Assessment of effectiveness of nest caging/fencing using field data from alkaline wetlands. Fencing and nest cages together are most effective predation management.

D.5 Larson et al. 2003

Authors: Michael A. Larson, Mark R. Ryan, and Robert K. Murphy
Year: 2003
Title: Assessing recovery feasibility for piping plovers using optimization and simulation
Citation: Wildlife Society Bulletin 31(4):1105−1116
Affiliation: U Missouri-Columbia, USFWS
Model type/structure: Larson et al. (2002) model for demographic simulation; linear integer programming model for maximizing fledging rate while minimizing costs.
Region/population: Great Plains including plovers nesting at river sites and at alkaline wetlands
Objective/focal mechanisms: Determine cost-efficient application of predator management strategies at lakes. "2 objectives: to 1) determine which combination of predator-exclusion techniques will maximize fledging rates within realistic budget and geographical constraints or minimize financial costs while reaching the fledging rate required for population recovery, and 2) determine whether recovery of the Great Plains population of plovers is demographically and economically feasible using predator exclusion."
Population structure, spatial resolution and scale: Two subpopulations: birds nesting on river sites and birds nesting on alkaline wetlands, divided by a constant proportion (0.27) based on proportions observed in 1991 international breeding census.
Temporal resolution and scale: Annual, 50 years
Habitat: Included only in distinction between breeding success at river vs. alkaline sites. The population was thought to be limited more by predation than resource availability. Habitat quality assumed constant and habitat consistently available.
Management actions: Five levels of predation management: no protection, nest cages, temporary electric fencing, permanent electric fencing, cages plus permanent fencing, and cages plus temporary electric fencing. Management at lakes only. 70% of lake pairs could receive management based upon distance from field offices.
Data sources: Data on predator exclusion was from Larson et al. (2002). Costs of management implementation were included. Population parameters were the same as Larson et al. (2002).
Metrics and analyses used: Instantaneous growth rate r for each of 10,000 replicates, converted to λ; financial cost for predator exclusion.
Validation: NA
Results: Combinations of cages and fencing were most efficient. When fences could no longer be used, nesting pairs received cages. Fencing alone was never optimal. Using mostly cages maximized fledglings; not managing most pairs minimized costs. Results were sensitive to fledging rates. Needed 92.7% of pairs to receive management to reach a fledging rate of 1.55, which stabilized or increased more than half of simulated populations. That would cost $11.2 million over 50 years.
Conclusions: "recovery of the Great Plains population of piping plovers is demographically and economically feasible using predator-exclusion management to increase reproductive success at

lakes." 9-32% recovery probability. Coordination with management would improve outcomes by increasing amount of fencing feasible.
Caveats mentioned: The study only looked at management on lakes. Optimization solutions applied only to initial state; had to assume decision criteria would remain constant; able to constrain costs only during the first year. Dynamic programming would address this, but more complex.
Implications for EA: Effectiveness of nest caging as a management action: fencing should not be used alone, focus on areas that can be fenced and coordinating efforts improves outcome. Framework for including economic factors into recovery analysis. Analysis was restricted to plovers nesting on lakes, not riverine habitat.

D.6 Calvert et al. 2006

Authors: Anna M. Calvert, Diane L. Amirault, Francois Shaffer, Richard Elliot, Alan Hanson, Julie McKnight and Philip D. Taylor
Year: 2006
Title: Population assessment of an endangered shorebird, the piping plover (*Charadrius melodus melodus*) in Eastern Canada
Citation: Avian Conservation and Ecology 1(3): 4 (online)
Affiliation: Dalhousie U., Canadian Wildlife Service, Acadia U.
Model type/structure: Stochastic matrix projection model with three age classes. Post-breeding birth-pulse matrix model.
Region/Population: Eastern Canada
Objective/focal mechanisms: Relative importance of breeding/nonbreeding seasons to annual population growth; estimate current vital rates for both segments of the population, determine sensitivity to variable population parameters for conservation planning.
Population structure, spatial resolution and scale: Two subpopulations, Gulf of St. Lawrence and southern Nova Scotia.
Temporal resolution and scale: Annual, 50 years.
Habitat: Not modeled; demographic rates specific to population
Management actions: None
Data sources: Monitoring of reproductive success and banding of juveniles and adults; data from 1998/1999 to 2003 for southern Nova Scotia/Gulf of St. Lawrence. Juvenile survival includes probability of fledging and surviving next 12 months. Survival estimates are local survival in case birds emigrated or went to unknown sites. Adult survival = 0.73, juvenile survival (from hatch) = 0.32/0.23, fledge success = 0.61/0.70, juvenile survival post-fledge = 0.53/0.34. Second year recruitment = 0.8/0.85, third-year recruitment = 0.99/0.98, # eggs = 3.8/3.9, hatch success = 0.46/0.51. Could not separate process from observation error.
Metrics and analyses used: Long-term expected growth rates (λ), sensitivities and elasticities, stable age distribution, age-specific reproductive values. Life table response experiment.
Validation: Model estimates of growth rates different from census, but confidence intervals on lambda include census-derived estimates for both segments.
Results: Gulf plovers decreased 3.6%/year, southern Nova Scotia plovers maintained stability (λ = 1.0043). There was a high variance in λ driven by variation in adult apparent survival and hatchling success and a high overlap in confidence intervals for two populations. The stochastic model λ was significantly lower than deterministic, supporting long-term decline. Similar stable age distributions. Differences in juvenile apparent survival could explain differences between the two populations. The current population was not at stable age distribution predicted by model, which would explain difference between census and model projections. Need λ = 1.18/1.08 to reach target in 5 years, 1.08/1.04 to reach target in 10 years. Increase fledglings to 1.08/1.63 females/nest to maintain pop., 1.41/1.86 to reach goal in 10 years.

Conclusions: Cannot say the two populations are statistically different. Adult survival was most important, but juvenile survival may explain observed differences. Would need major increases in reproductive parameters to recover plovers, as elasticities are low. Adult survival important but may not be flexible. Extensive efforts to protect nesting areas still resulted in declines in 2004 and 2005.
Caveats mentioned: Low resighting effort in more remote regions may reduce juvenile survival but mark-recapture models account for that; emigration outside area possible but unlikely. Transient dynamics introduce uncertainty into long-term predictions.
Implications for EA: Limited application because of location/population and did not include management; does confirm general results from other PVAs about importance of survival rates.

D.7 McGowan and Ryan 2009

Authors: Conor P. McGowan, Mark R. Ryan
Year: 2009
Title: A quantitative framework to evaluate incidental take and endangered species population viability
Citation: Biological Conservation 142: 3128−3136
Affiliation: University of Missouri-Columbia
Model type/structure: Stochastic model including sampling variance. Modification of two-stage Ryan et al. (1993) and Larson et al (2002) models. Ceiling-type density dependence.
Region/population: Great Plains population using alkali wetlands, Great Plains rivers, and Missouri River, distinguished by reproductive success.
Objective/focal mechanisms: "Examine the effect of permitted incidental take (egg mortality) and probable take (chick mortality) or population growth, abundance, and probability of quasi-extinction." "Four research hypotheses: (1) stochastic variation in adult survival is the primary factor affecting population viability; (2) stochastic variation in adult survival and incidental take affect population viability; (3) renesting rate can compensate for the negative effects of incidental take; (4) variability of incidental take affects population growth, abundance, and quasi-extinction probability."
Population structure, spatial resolution and scale: Three locations (alkali wetlands, Great Plains rivers, Missouri River). Not spatially explicit. The population redistributes across the locations each year, without interannual correlations, due to lack of data about movements.
Temporal resolution and scale: Annual, 30 years
Habitat: Not modeled; reproductive rates specific to the three locations (alkali wetlands, Great Plains rivers, Missouri River.)
Management actions: Modeled incidental take on eggs and/or chicks, modeled as proportion of eggs/chicks taken, with proportion drawn from log-normal distribution based on prescribed mean and variance from incidental take permit.
Data sources: Adult (0.737) and juvenile (0.48) survival was from Larson et al. (2000) and (2002). The population distribution was based on international censuses. Fecundity parameters were estimated from empirical data: alkali wetlands from known pairs and broods, river from nests/female, clutch size, nest survival and chick survival data in order to separate out take.
Metrics and analyses used: Calculated average growth rates, abundances, and probability of quasi-extinction (100 breeding females). Used simulated data for regression based analysis.
Validation: NA
Results: Simulations estimated ~7.5% population decline, with 0.333 probability of quasi-extinction over 30 years. No clear pattern was observed between different take scenarios and growth/abundance, probably because of adult and juvenile stochasticity. Chick take did not affect final abundance, but did lower growth rates and higher probability of quasi-extinction. Increased renesting neutralizes effects of chick take. Higher proportions nesting in Missouri River led to overall poorer outcomes.

Model with adult survival, # nests, egg take, and chick take as covariates was best predictor of final abundance.
Conclusions: Incidental take worsens population metrics. Take is significant, but the effect is small compared to adult mortality). Chick take should be explicitly addressed and monitored. Renesting helps mitigate for take, but there is not enough data to suggest this level of renesting is likely or possible. Renesting alone will not address downward trend. Accounting for all uncertainty is important. Jeopardy is significant even without take. Need for clear jeopardy decision criteria.
Caveats mentioned: Spatial dynamics between habitat types are not really known.
Implications for EA: Example of breaking down fledgling production process into stages in order to more directly test effects of specific management actions. Demonstrated small, but significant, effect of incidental take on population viability.

D.8 McGowan et al. 2011

Authors: Conor P. McGowan, Michael C. Runge, Michael A. Larson
Year: 2011
Title: Incorporating parametric uncertainty into population viability analysis models
Citation: Biological Conservation 144:1400−1408
Affiliation: USGS Patuxent, Minn. Dept. Natural Resources
Model type/structure: Fully stochastic PVA including demographic, parametric, and temporal uncertainty. Based on Ryan et al. (1993) model and used to compare different methods of incorporating parametric uncertainty. Ceiling-type density dependence, capped at 8,000 females.
Region/Population: Great Plains
Objective/focal mechanisms: Focuses on hierarchical process for incorporating parametric uncertainty in simulation models typically used in PVA, with plovers as an example.
Population structure, spatial resolution and scale: One population
Temporal resolution and scale: Annual, 50 year
Habitat: Not modeled
Management actions: Not modeled
Data sources: Mean adult survival = 0.737, mean juvenile survival = 0.48, mean fecundity = 0.42. Fecundity values and variances appropriate for shorebird this size but not based on empirical data or specific published values.
Metrics and analyses used: Median population, probability of extinction
Validation: NA
Results: Scenarios that separated and excluded parametric uncertainty, or where parametric and temporal uncertainty were included together in annual loop (as if all temporal) predicted near 0% chance of extinction in 50 years. Simulations with parametric uncertainty in replication loop and temporal uncertainty in annual loop predicted high variability and 0.22 probability of extinction within 50 years.
Conclusions: Including all types of uncertainty improves model predictions for making management decisions. Not accounting for parametric uncertainty in population models hinders ability to accurately assess risk. Parametric uncertainty has several sources; methods do not currently exist to separate these or to deal with unknown bias, except through sensitivity analysis.
Caveats mentioned: NA
Implications for EA: Not a management model per se, but our existing habitat/bird population model incorporates the uncertainty described in the paper-; therefore this paper supports our approach for handling uncertainty.

D.9 McGowan et al. 2011

Authors: Conor P. McGowan, Mark R. Ryan, Michael C. Runge, Joshua J. Millspaugh, Jean Fitts Cochrane
Year: 2011
Title: The role of demographic compensation theory in incidental take assessments for endangered species
Citation: Biological Conservation 144: 730−737
Affiliation: University of Missouri, USGS Patuxent
Model type/structure: Stochastic population model from McGowan and Ryan 2009 modified to include density dependence in post-fledging winter survival (negative linear logistic); temporal and sampling variance.
Region/population: Great Plains population using alkali wetlands, Great Plains rivers, and Missouri River, distinguished by reproductive success.
Objective/focal mechanisms: Can density-dependent mortality compensate for the negative effects of take on threatened populations? Do declining populations have less capacity for compensation than stable or growing ones?
Population structure, spatial resolution and scale: Three locations (alkali wetlands, Great Plains rivers, Missouri River), not spatially explicit. The population redistributes across the habitats each year, without interannual correlations, due to lack of data about movements.
Temporal resolution and scale: Annual, 30 years
Habitat: Not modeled; reproductive rates specific to three locations and habitat types (alkali wetlands, Great Plains rivers, Missouri River.)
Management actions: Eight levels of annual egg take: 0, 2%, 4%, 6%, 8%, 10%, 12% and 14%.
Data sources: No empirical estimates for the density-dependent survival, but authors chose values that included the range of published first winter survival estimates. Other data sources same as previous models by authors. They also used a hypothetical stable population to see if stable populations can compensate for take even if declining ones cannot.
Metrics and analyses used: For each year in each replication, ran the 32 different population and take scenarios to reduce variation across scenarios. Regression models to test effects of take on quasi-extinction. Median population abundance over 30 years.
Validation: NA
Results: Median population declined except high survival and no juvenile density dependence. Low survival and juvenile density dependence had greatest declines. Increased take increased probability of quasi-extinction.
Conclusions: Some levels of take may be compensated by density dependence if the population is not rapidly declining. However, there may not be opportunities for such compensation as food-mediated density dependence in wintering grounds is unlikely. Capacity for density dependence most likely in first-year survival or reproduction. Timing and life stage of take affects ability to compensate. Stochasticity undermines compensation.
Caveats mentioned: NA
Implications for EA: Did not find much support for density dependence in overwinter survival as a mechanism; suggested density dependence in reproduction is more likely but did not have the evidence. Example of comparing multiple models to account for structural uncertainty.

D.10 McGowan 2013

Authors: Conor P. McGowan
Year: 2013
Title: A structured approach to incidental take decision making

Citation: Environmental Management 51:241−250
Affiliation: USGS
Model type/structure: Environmentally stochastic simulation model from McGowan and Ryan (2009). Also used McGowan and Ryan (2011) model incorporating density-dependent juvenile survival to show effects of model uncertainty.
Region/population: Great Plains population using alkali wetlands, Great Plains rivers, and Missouri River, distinguished by reproductive success.
Objective/focal mechanisms: Application of PrOACT approach to evaluating incidental take.
Population structure, spatial resolution and scale: Three locations, not spatially explicit. Population redistributes across the habitats each year, without interannual correlations, due to lack of data about movements.
Temporal resolution and scale: Annual, 100 years
Habitat: Not modeled; reproductive rates specific to three locations and habitat types (alkali wetlands, Great Plains rivers, Missouri River.)
Management actions: Maximum take of 10% of eggs, intermediate take of 5%, and a no-take action.
Data sources: First-year survival = 0.48, adult survival = 0.82; others structured as in McGowan and Ryan (2009).
Metrics and analyses used: Change in abundance, change in population growth rate, change in probability of extinction, and charge in probability of reaching recovery criteria.
Validation: NA
Results: Without competing objectives, "no take" is valued highest. When used in conjunction with invented pallid sturgeon model results to illustrate effects of competing objectives, 5% take option had best combined expected value.
Conclusions: The incidental take decision lends itself well to PrOACT approach. Decisions are transparent and understandable and the framework allows uncertainty to be incorporated. Optimal outcomes can be selected even with structural uncertainty. Translating qualitative objectives into quantitative value functions can be difficult.
Caveats mentioned: Results from using this process depend on how objectives are defined and weighted.
Implications for EA: Example of using a PVA model in the PrOACT process.

D.11 Buenau et al. 2014

Authors: Kate E. Buenau, Tim L. Hiller, Andrew J. Tyre
Year: 2014
Title: Modelling the effects of river flow on population dynamics of piping plovers (*Charadrius melodus*) and least terns (*Sternula antillarum*) nesting on the Missouri River
Citation: River Research and Applications, 30(8): 964-975
Affiliation: Pacific Northwest National Laboratory, University of Nebraska-Lincoln, Oregon Dept. Fish and Wildlife
Model type/structure: Dynamic spatially implicit model of sandbar habitat, stochastic population viability model with density-dependent fledgling production. Includes environmental, demographic stochasticity, and observation and estimation error.
Region/population: Missouri River sandbar habitat
Objective/focal mechanisms: Importance of variability in river flow on population dynamics and implications for habitat management. General behavior of model and relative contributions of sources of uncertainty to population dynamics. Effects of rare habitat-creating flows on population dynamics.
Population structure, spatial resolution and scale: Five subpopulations on four river reaches (three with significant plover use) and sandbars in upper Lewis and Clark Lake**.**
Temporal resolution and scale: Annual, 5-20 years

Habitat: Dynamically modeled sandbar habitat created through construction and/or high flows and lost through erosion and vegetation rate. Modeled implicitly as area/river segment.
Management actions: Habitat construction
Data sources: Adult survival 0.8 from Roche et al. (2010), juvenile survival 0.6 (expert opinion), density-dependent fledgling production estimated from USACE monitoring data and USGS/USACE habitat quantification from satellite imagery, 2006-2010.
Metrics and analyses used: Median and 95% CI of habitat area and population size for 5 years, CV of observed adult abundance.
Validation: Compared with monitoring data from 1998 to 2010. Both observed populations well within confidence interval of projections, but plover median population size overestimates observed dynamics.
Results: Highest population sensitivity was to adult survival, followed by juvenile survival. Estimation error was the largest source of variability, followed closely by flow variability. Flow has strong effects on the outcome of management actions; can be larger driver of dynamics than intensity of habitat creation. Large increases in habitat area caused by high flows led to response in plover population size with effects lasting about 10 years before decline if no habitat creation occurs.
Conclusions: River flow, including flow variability and extreme events, is a major driver of habitat and population dynamics and should be accounted for when adaptively managing habitat and populations.
Caveats mentioned: The model was parameterized from years where habitat was degraded, may not accurately represent productivity and population dynamics when large amounts of high quality habitat is available. Habitat quality was not specifically modeled. The model did not include reservoirs that may act as a buffer for plovers when sandbar habitat is degraded. It also did not include within-season variability in habitat availability.
Implications for EA: Effects of habitat creation as a management action; effects of flow on population dynamics and management outcomes; example of dynamics after habitat-creating high flows. Model accounts for dynamic nature of sandbar habitat.

D.12 McGowan et al. 2014

Authors: Conor P. McGowan, Daniel H. Catlin, Terry L. Shaffer, Cheri L. Gratto-Trevor, Carol Aron
Year: 2014
Title: Establishing endangered species recovery criteria using predictive simulation modeling
Citation: Biological Conservation 177: 220−229
Affiliation: USGS, Virginia Tech, Environment Canada, USFWS
Model type/structure: Stochastic metapopulation projection model related to previous models published by McGowan et al. Includes ceiling-type density dependence and includes parametric uncertainty, temporal variability, demographic stochasticity, and observation error.
Region/population: Great Plains population, four subpopulations
Objective/focal mechanisms: Use population projection model and regression analysis of simulated data to link recovery criteria to extinction risk; determine population size, distribution, and growth rate needed to reach prescribed extinction risk and effects of uncertainty on recovery criteria.
Population structure, spatial resolution and scale: Four populations: Northern Rivers (upper Missouri and reservoirs), Southern Rivers (middle Missouri, Niobrara, Loup, Platte), US Alkali Lakes, and Prairie Canada.
Temporal resolution and scale: Annual, 50 years
Habitat: Not included. Fecundity varies by region based on difference in reproductive success that may be caused by differences in habitat quality**.**
Management actions: Not included. Authors note that fecundity of plovers nesting in the Southern Rivers region is affected by habitat management and may decline if those actions are not continued.

Data sources: Adult survival = 0.78 (Catlin unpub., Larson et al 2000, Cohen and Gratto-Trevor 2011, Roche et al. 2010). Juvenile survival = 0.52, (Catlin unpub.). Productivity = 0.77 female fledglings/pair (Catlin; Southern Rivers), 0.32 (Shaffer et al. 2013; Northern Rivers), 0.6 (McGowan and Ryan; Alkali Lakes), 0.52 (Gratto-Trevor; Canada). Long distance dispersal rate = 0.02 (Catlin; Gratto-Trevor; Roche et al. 2012).
Metrics and analyses used: Mean expected population growth rate, probability of extinction in year 50.
Validation: NA
Results: The mean population growth rate was >1 and extinction probability = 0.033 for entire metapopulation. The southern rivers region had lowest individual extinction probability due to management. Increasing the population ceiling decreased extinction risk somewhat and increased median abundance over time. Doubling the dispersal rate decrease overall extinction risk but not the risk for individual regions; small change in total population size and some increase in median abundance within regions. Observation error increases extinction probability for a given population size. Results include tables of extinction risk for combinations of initial size, growth rate, and fecundity.
Conclusions: This model was more optimistic than its precursors. Dispersal rates were too low to cause large reductions in extinction probability, even when doubled. Provides method for establishing recovery criteria despite uncertainty.
Caveats mentioned: Uncertainties in demographic rates, density dependence, dispersal, etc. acknowledged
Implications for EA: Study relates more to target-setting than effects of management actions. The productivity rate for birds nesting in the Southern Rivers region was likely increased by habitat and predation management actions there; growth rates and persistence will likely decline if actions are discontinued.

Appendix E

Tern Model Review

Appendix E

Tern Model Review

E.1 Akcakaya et al. 2003

Authors: H. Resit Akcakaya, Jonathan L. Atwood, David Breininger, Charles T. Collins, Brean Duncan
Year: 2003
Title: Metapopulation dynamics of the California Least Tern
Citation: The Journal of Wildlife Management 67(4): 829−842
Affiliation: Applied Biomathematics, Antioch, Dynamac, California State University-Long Beach
Model type/structure: Metapopulation model with stochastic age-structured matrix (5 age classes) for each population, using RAMAS Metapop. Modeled zero-fecundity years as local catastrophes separate from normal variability in fecundity.
Region/population: California
Objective/focal mechanisms: Simulate future changes in population under alternative assumptions about threats and management. Local catastrophes (no fledglings) assumed due to predation.
Population structure, spatial resolution and scale: 17 populations of active breeding sites or clusters of sites <5 km apart**.**
Temporal resolution and scale: Annual, 50 years
Habitat: Not included, though each site has a static carrying capacity.
Management actions: Did not evaluate specific management actions in detail; modeled predator management as a reduction in probability of local catastrophe to zero.
Data sources: Published sources (Massey et al. 1992) and annual surveys from California Fish and Game. Average age of first breeding is 3. Survival to age 2 is 0.16 during normal years and 0.03 for El Niño/Southern Oscillation years. Population-specific initial abundance, carrying capacity, relative fecundity and local catastrophe probability. 2-4% probability of El Niño causing regional catastrophe. Colony fidelity of 50% from Massachusetts population; California site fidelity appears to be higher (50-90% for adults).
Metrics and analyses used: Risk-based sensitivity analysis. Risk of extinction, risk of decline by given percentage in 50 years, or risk of decline within time horizon to threshold.
Validation: The matrix model was underestimating observed growth, perhaps because of change in age structure after 1982-83 El Niño, or because vital rates have improved since Massey et al. 1992. Under-observation of fledgling production may also contribute. Used both low and high stage matrix to compensate.
Results: Did not detect density dependence in fledgling data, but assumed ceiling population sizes of 2x the individuals observed at that population. Medium parameters predicted 0 risk of extinction or substantial decline. Low parameters had 98% chance of 15% decline in 25 years. Model results were most sensitive to vital rates, moderately sensitive to carrying capacity and had only low sensitivity for other parameters. Predator control had an effect on the risk of decline only for low parameter values, 1-8% decrease in risk depending upon number and locations of populations with predator control. Modeling of predator control was limited however.
Conclusions: Generally predicted continuing population increases and low risk of substantial decline in next 50 years. The amount and focus of predator control influenced the effectiveness, so management should be focused on populations that make the greatest contributions to viability.
Caveats mentioned: Two types of data most improve model: additional information about survival and estimates of effects of management actions on fecundity.

Implications for EA: The habitat and to some extent tern behavior in California are sufficiently different that this study has limits as a line of evidence. It is the only published PVA-style model for least terns, however.

E.2 Lott et al. 2012

Authors: Casey A. Lott, Steven F. Railsback, Colin J.R. Sheppard, Michael C. Koohafkan
Year: 2012
Title: TernCOLONY 1.0 model description; The development and testing of TernCOLONY 1.0: an individual- based model of Least Tern reproduction
Citation: ERDC-TR-DOER; ERDC-TR-DOTS
Affiliation: American Bird Conservatory, Lang, Railsback & Associates
Model type/structure: Spatially explicit, agent-based model of colony persistence. Includes tern life history, tern-habitat interactions, predator interactions, georeferenced habitats, habitat restoration and management.
Region/population: Arkansas river; designed to be applied to any large regulated river
Objective/focal mechanisms: Determine what factors affect population persistence of least terns on large rivers below multi-purpose dams.
Population structure, spatial resolution and scale: Regional breeding population (tested with population below Keystone Dam on Arkansas River).
Temporal resolution and scale: Daily during breeding season, tested with hydrographs from 1977−2008.
Habitat: Sandbars within channel
Management actions: Dam releases, ecosystem restoration
Data sources: Published sources (Lott & Wiley 2012), USACE and American Bird Conservatory monitoring data, nest counts from 2005 & 2008 USACE surveys (Tulsa district).
Metrics and analyses used: Reproductive success (regional and site-specific) used as metric. Sensitivity analysis on 37 different parameters (9 values of each parameter tested).
Validation: Pattern-oriented analysis used for validation/evaluation of the model: 8 patterns for colony site selection, 6 patterns for nest-site selection, 12 patterns for mortality and reproduction (Lott et al 2012); sensitivity analysis on 37 parameters.
Results: High variability in annual abundance, but median is similar; mortality sources vary from year to year; reproductive success not density dependent; distribution of annual reproductive success is bimodal; reproductive success varies among sites within given year; flooding mortality; regional reproductive success lower with poor habitat conditions; higher flooding mortality with degraded habitat; higher predation with poor habitat; reproductive success is sensitive to changes in parameters that control mortality and number of breeding attempts; flooding loss is sensitive to parameters that affect mortality.
Conclusions: Flooding and habitat quality strongly influence mortality; nesting stage has higher mortality than chick stage.
Caveats mentioned: Field data not collected across wide range of values (limited to environmental conditions that terns were exposed to at the time of data gathering); limited habitat data available during testing, however, most missing data currently available.
Implications for EA: Flooding impacts mortality; nesting stage is critical life stage

E.3 Kanapaux and Kiker 2013

Authors: William Kanapaux, Gregory A. Kiker
Year: 2013

Title: Development and testing of an object-oriented model for adaptively managing human disturbance of least tern (*Sternula antillarum*) nesting habitat
Citation: Ecological Modelling 268: 64−77
Affiliation: University of Florida Gainesville
Model type/structure: Question and Decision system framework (links GIS with modeled abiotic and biotic interactions). Object-oriented, agent-based modeling approach. Pattern-oriented modeling.
Region/population: South Carolina
Objective/focal mechanisms: Adaptively managing human disturbance. Effects of human users on tern nesting and how can they be managed.
Population structure, spatial resolution and scale: 4.8 km beach divided into seven sections, four sections with nesting habitat.
Temporal resolution and scale: Nesting season, daily time step
Habitat: Sites described, but habitat itself not explicitly addressed in model
Management actions: Management of access: 1) full access, 2) beach closed one day a week, 3) no access, 4) section closures.
Data sources: Field observations during 2009 and 2010 nesting seasons to collect behavioral data on tern nesting and information on human use of beach.
Metrics and analyses used: Local sensitivity analysis; management scenarios. Metric is fledgling production.
Validation: "Reasonable simulation of conditions found onsite in 2009-2011 nesting seasons". Human disturbance has been found to limit nesting and cause population declines.
Results: No significant difference in productivity for closures of 0−3 days. Complete closure of beaches did improve productivity significantly. More information about responses to human disturbance is the most important for understanding the effects of disturbance on productivity.
Conclusions: Only full closure resulted in gains sufficient enough to maintain population. Interactions between high human use of beaches and propensity for overwash (inundation of nests.)
Caveats mentioned: Assumes some conspecific attraction, constant clutch size, constant effects of human disturbance, all spatial units with nesting habitat equally likely to be selected, food is abundant, predation is not significant.
Implications for EA: Assesses effects of human disturbance and potential management actions, but situation is considerably different from Missouri River.

E.4 Buenau et al. 2014

Authors: Kate E. Buenau, Tim L. Hiller, Andrew J. Tyre
Year: 2014
Title: Modelling the effects of river flow on population dynamics of piping plovers (*Charadrius melodus*) and least terns (*Sternula antillarum*) nesting on the Missouri River
Citation: River Research and Applications, 30(8): 964-975
Affiliation: Pacific Northwest National Laboratory, University of Nebraska-Lincoln, Oregon Dept. of Fish and Wildlife
Model type/structure: Dynamic spatially implicit model of sandbar habitat, stochastic population viability model with density-dependent fledgling production. Includes environmental, and demographic stochasticity and observation and estimation error.
Region/population: Missouri River sandbar habitat
Objective/focal mechanisms: Importance of variability in river flow and implications for habitat management. General behavior of model and relative contributions of sources of uncertainty to population dynamics. Effects of rare habitat-creating flows on population dynamics.
Population structure, spatial resolution and scale: Five subpopulations on four river reaches and sandbars in upper Lewis and Clark Lake.

Temporal resolution and scale: Annual, 5-20 years
Habitat: Dynamically modeled sandbar habitat created through construction and/or high flows and lost through erosion and vegetation rate. Modeled implicitly as area/river segment.
Management actions: Habitat construction
Data sources: Survival rates estimated from Massey et al. 1992 (adults >3 years 0.92, 2−3 years 0.81, Juvenile 0.4), density-dependent fledgling production estimated from USACE monitoring data and USGS/USACE habitat quantification from satellite imagery, 2006−2010 (density dependence found to be extremely mild).
Metrics and analyses used: Median and 95% CI of habitat area and population size for 5 years, CV of observed adult abundance.
Validation: Compared with monitoring data from 1998 to 2010. Both observed populations well within confidence interval of projections, but tern median population size underestimates observed dynamics.
Results: Highest sensitivity to adult survival, followed by juvenile survival. Estimation error is by far the largest source of variability in model predictions for least terns, flow variability contributes very little. Flow does not show strong effects on management outcomes. Slightly better long-term prospects with continued construction compared to single high flow event.
Conclusions: Because terns show very little density dependence, flow variability is not a major driver of dynamics, though habitat availability does eventually become a factor.
Caveats mentioned: Underpredictions in tern dynamics may result because model was parameterized during years where habitat was degraded and predation was significant in some areas. Habitat quality and other factors of habitat preference not modeled. Inundation risk during a season (intra-season flow dynamics) which may affect terns more than overall habitat availability was not modeled specifically.
Implications for EA: Habitat area (above a minimum amount needed) does not have strong effect on terns, so they do not respond to habitat construction very noticeably. May need to consider other factors to better understand habitat needs and potential management.

Appendix F

Images Enlarged from Plover Literature Review Table

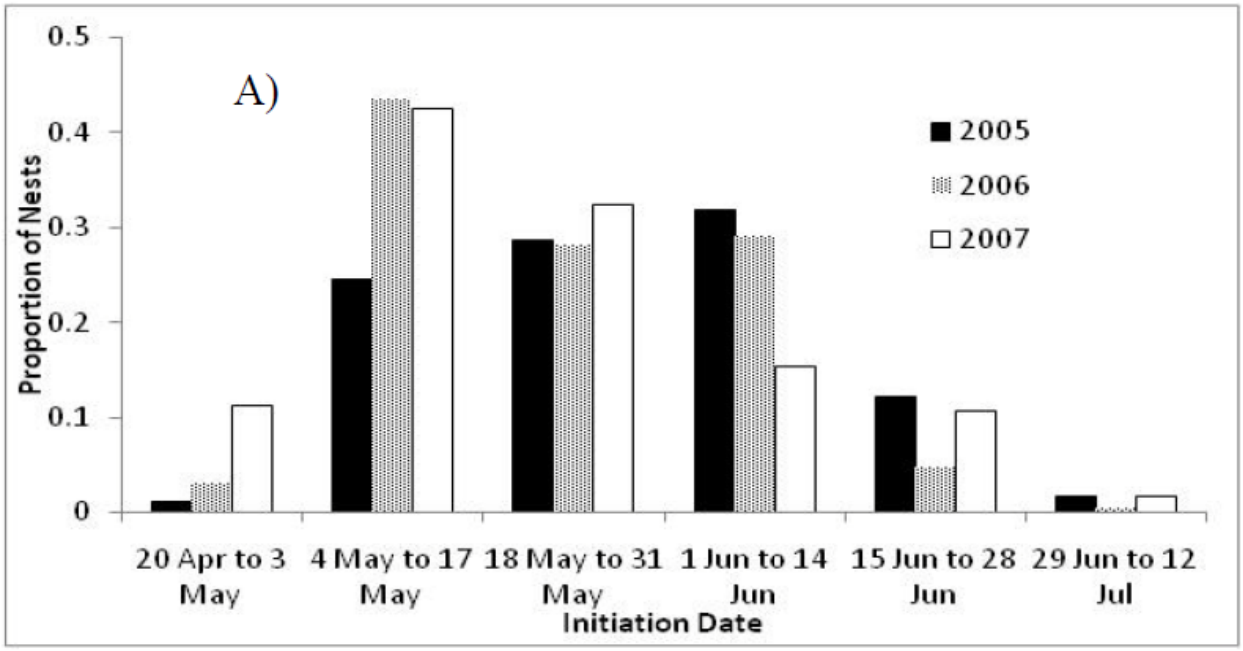
Appendix F

Images Enlarged from Plover Literature Review Table

F.1 Reproductive Success

F.1.1 Catlin 2009

Nest Initiation:



Nest Success:

Year	Total Nests	Known Successful	Possibly Successful
2005	183	97 (53.0%)	12 (6.6%)
2006	205	105 (51.2%)	6 (2.9%)
2007 [b]	211	107 (50.7%)	18 (8.5%)
Total	599	309 (51.6%)	36 (6.0%)

Nest Fate:

Table 2: Raw nest success (% total) for piping plover nests on the Missouri River during the 2005 – 2007 nesting seasons. Known successful nests had one or more chicks associated with the nest or the parents of the nest. Possibly successful nests did not have chicks associated with the nest or adults but eggs disappeared within 2 days of the projected hatch date. Failed nests did not have chicks associated with the nest or adults and disappeared outside the window of reasonable hatch dates (excluding abandoned nests).

Year	Total Nests	Known Successful	Possibly Successful	Failed Predation	Failed Abandoned	Failed Flooding	Failed Other [a]	Failed Unknown
2005	183	97 (53.0%)	12 (6.6%)	18 (9.8%)	7 (3.8%)	1 (0.5%)	5 (2.7%)	43 (23.5%)
2006	205	105 (51.2%)	6 (2.9%)	17 (8.3%)	21 (10.2%)	10 (4.9%)	4 (2.0%)	42 (20.5%)
2007 [b]	211	107 (50.7%)	18 (8.5%)	48 (22.7%)	15 (7.1%)	9 (4.3%)	2 (0.9%)	12 (5.7%)
Total	599	309 (51.6%)	36 (6.0%)	83 (13.9%)	43 (7.2%)	20 (3.3%)	11 (1.8%)	97 (16.2%)

[a] Includes bank erosion, weather events, and nests stepped on by researchers.

[b] Change in protocols led to more failures classified as predation. If eggs were missing before the hatch date and none of the other causes of failure were implicated, predation was thought the likely cause. The other frequent causes of nest loss, abandonment and flooding were obvious to observers. Many of the unknown nest losses from 2005 and 2006 were probably as a result of predation.

F.1.2 Catlin et al. 2011b

Nest Success:

Table 2. Raw nest success (% total) for piping plover nests on the Missouri River during the 2005–2007 nesting seasons. Known successful nests had ≥1 chick associated with the nest or the parents of the nest. Possibly successful nests did not have chicks associated with the nest or adults but eggs disappeared within 2 days of the projected hatch date. Failed nests did not have chicks associated with the nest or adults and disappeared outside the window of reasonable hatch dates (excluding abandoned nests).

			Known successful		Possibly successful		Failed predation		Failed abandoned		Failed flooding		Failed other[a]		Failed unknown	
Yr	Habitat type	Total nests	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%
2005	Natural	76	33	43.4	8	10.5	6	7.9	3	3.9	0	0.0	0	0.0	26	34.2
	Managed	36	12	33.3	2	5.6	12	33.3	2	5.6	0	0.0	0	0.0	8	22.2
	Engineered	71	52	73.2	2	2.8	0	0.0	2	2.8	1	1.4	5	7.0	9	12.7
2006	Natural	47	14	29.8	1	2.1	7	14.9	5	10.6	9	19.1	0	0.0	11	23.4
	Managed	31	7	22.6	1	3.2	8	25.8	3	9.7	1	3.2	0	0.0	11	35.5
	Engineered	127	84	66.1	4	3.1	2	1.6	13	10.2	0	0.0	4	3.1	20	15.7
2007	Natural	54	21	38.9	7	13.0	14[b]	25.9	2	3.7	6	11.1	1	1.9	3	5.6
	Managed	16	6	37.5	4	25.0	4[b]	25.0	1	6.3	0	0.0	0	0.0	1	6.3
	Engineered	141	80	56.7	8	5.7	29[b]	20.6	12	8.5	3	2.1	1	0.7	8	5.7
Total		599	309	51.6	37	6.2	82	13.7	43	7.2	20	3.3	11	1.8	97	16.2

[a] Includes bank erosion, weather events, and nests stepped on by researchers.
[b] Change in protocols led to more failures classified as predation. If eggs were missing before the hatch date and none of the other causes of failure were implicated, we concluded that predation was the cause of failure. The other frequent causes of nest loss, abandonment and flooding, were obvious to observers. Many of the unknown nest losses from 2005 and 2006 were probably as a result of predation.

F.1.3 Cohen et al. 2009

Nest Initiation:

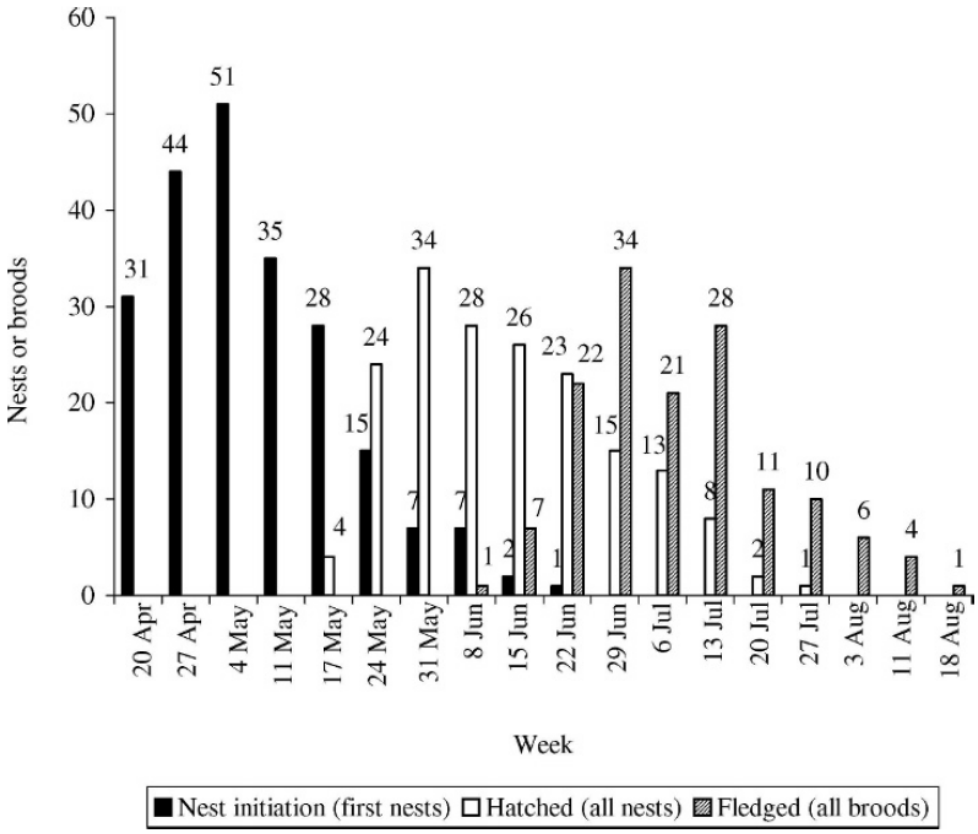


Figure 11. Piping plover nest initiation, hatch, and fledge dates at West Hampton Dunes and the reference area, New York, USA, 1993–2004. The number above each bar is the number of nests or broods represented by the bar.

Nest Success:

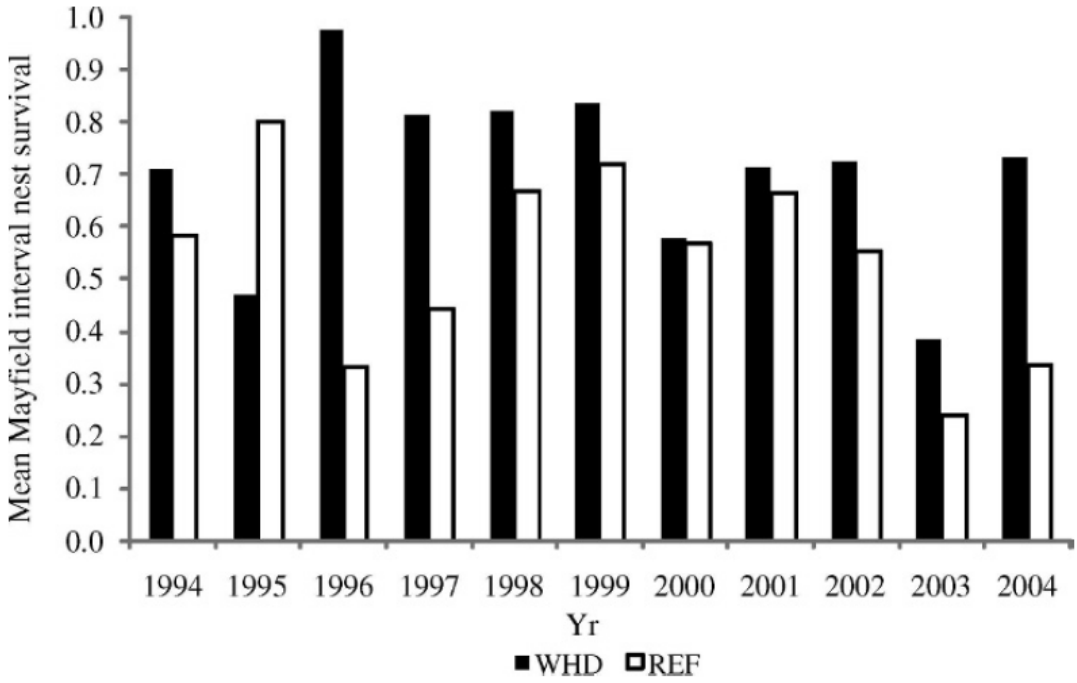


Figure 14. Mean nest survival of piping plovers (Mayfield method) at West Hampton Dunes (WHD) and the reference area (REF), New York, USA, by year.

Nest Fates:

Table 6. Piping plover nests lost (no. and %) to different causes, West Hampton Dunes (WHD) and the reference area (REF), New York, USA, 1993–2004.

			Nests lost to											
			Cat		Crow[b]		Fox		Unidentified predator		Abandonment		Other[c]	
Site	Nest fenced?[a]	*n*	no.	%	no.	%	no.	%	no.	%	no.	%	no.	%
WHD	No	358	12	3.4	14	3.9	26	7.3	61	17.0	20	5.6	22	6.1
	Yes	35	0	0.0	0	0.0	8	22.9	0	0.0	3	8.6	1	2.9
REF	No	151	11	7.3	19	12.6	2	1.3	45	29.8	14	9.3	10	6.6
	Yes	47	0	0.0	0	0.0	0	0.0	0	0.0	14	29.8	2	4.3

[a] Fenced by predator exclosures.
[b] *Corvus brachyrhynchos.*
[c] Blackbird (Icteridae), gull (*Larus* spp.), dog (*Canis lupus familiaris*), raccoon (*Procyon lotor*), flooding, pedestrian; contributed to <5% of nest losses in each row.

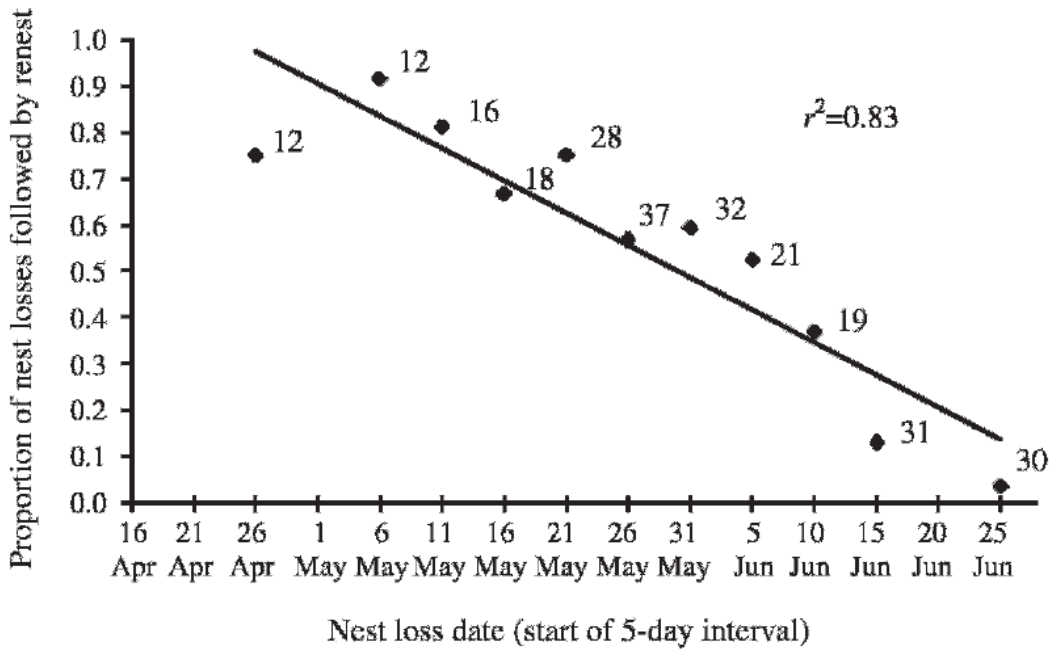


Figure 17. Mean proportion of piping plover nest losses followed by a renest versus date of nest loss at West Hampton Dunes and the reference area, New York, USA, 1994–2004. We calculated mean probabilities within 5-day intervals, except for the first and penultimate interval, which were 10 days, and the last interval, which was 45 days. Numbers to the right of points are sample sizes for the interval. Fitted line and coefficient of determination (r^2) are shown to illustrate the relationship, but our inferences were based only on model-averaged estimates (all possible logistic regressions of 5 variables) and relative importance of the effect of nest-loss date.

Fledglings per Nest or Minimum Adult Pairs:

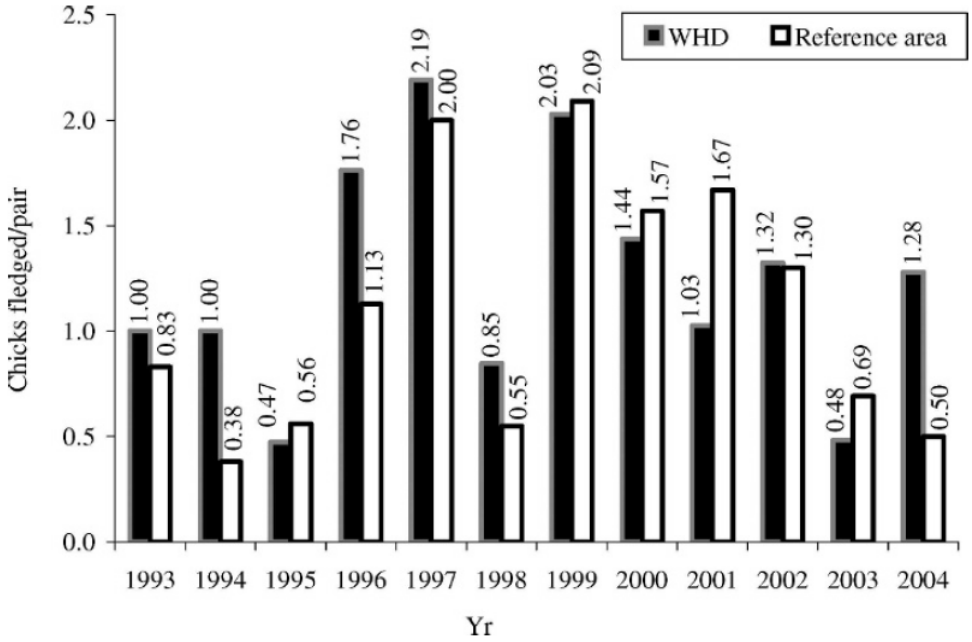


Figure 21. Piping plover reproductive output (chicks fledged/pair) at West Hampton Dunes (WHD) and the reference area, New York, USA, 1993–2004. Numbers over the bars are the values represented by the bars.

F.1.4 Doherty and Heath 2011

Nest Success – Nest Fate:

Table 1. Number of failed eggs and nests in each nest fate category for breeding piping plovers on Suffolk County Park beaches, New York, beaches in 2006 and 2007. Three nests failed due to unknown reasons.

		Fate category									
		Successful		Mixed-fate		Abandoned		Depredated		Flooded	
Yr	Hatching success	Failed eggs	Nests	Failed eggs	Nests	Failed eggs	Nests	Failed eggs	Nests	Failed eggs	Nests
2006	0.65	0	22	16	14	29	9	15	4	5	2
2007	0.56	0	24	8	7	23	8	40	11	8	3
Total	0.60	0	46	24	21	52	17	55	15	13	5

Table 2. Fates of exclosed and unexclosed piping plover nests on Suffolk County Park beaches, New York, in 2006 and 2007. Predator exclosure use increased piping plover hatching success.

		Fate category									
		Successful		Mixed-fate		Abandoned		Depredated		Flooded	
Management	Hatching success	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%
Exclosed	0.68	20	49	11	27	7	17	1	2	2	5
Not exclosed	0.56	26	39	10	15	10	15	14	21	3	5
Total	0.60	46	46	21	17	17	16	15	14	5	5

F.1.5 Donlan et al. 2003

Nest Success – Nest Fate – Fledglings per Nest or Minimum Adult Pairs:

Table 1. *Charadrius melodus*. Results of piping plover reproductive success at reference and impact area beaches. Source of data: McGourty (1996), Table 2 and Casey (1996, Appendix E)

Year	Sites	Pairs	Nests	Eggs	Hatched	Hatched (%)	Fledged (%)	Fledged per pair	Fledglings
Reference area beaches									
1996	6	20	22	87	75	86	42	56	2.1
1995	4	13	18	63	46	73	24	52	1.9
1994	4	9	10	39	31	80	22	71	2.4
1993	3	9	11	45	32	71	23	72	2.6
1992	2	5	6	23	15	65	12	80	2.4
Impact area beach									
1996	1	9	16	53	22	42	9	41	1.0
1995	1	9	9	35	31	89	14	45	1.6
1994	1	8	9	33	17	52	8	47	1.0
1993	1	8	10	38	33	87	7	21	0.9
1992	1	6	6	19	7	37	4	57	0.7

F.1.6 Espie et al. 1996

Nest Success – Nest Fate – Fledglings per Nest or Minimum Adult Pairs:

Table 5. Sample means and standard deviations of habitat variables for successful, flooded, and depredated nests.

Variable	Successful nests	Flooded nests	Depredated nests	*F* (ANOVA)[a] 1	2
Near water	68.23±40.10 (30)	43.92±26.45 (24)	53.00±20.53 (13)	6.06*	0.53
Main water	131.70±93.64 (30)	69.29±51.44 (24)	81.62±69.13 (13)	8.66**	3.30
Vegetation	120.83±142.12 (30)	92.87±82.32 (24)	23.83±15.28 (13)	0.06	5.05*
Pebbles	1025.67±354.38 (21)	1051.92±390.11 (20)	946.89±21.74 (11)	2.53*	0.25
Stones	37.05±26.27 (21)	47.03±18.89 (20)	31.83±21.74 (11)	1.94	0.32

Note: Numbers in parentheses are sample sizes.

[a]1, successful vs. flooded nests; 2, successful vs. depredated nests; *, $p < 0.05$; **, $p < 0.01$; others $p > 0.05$.

F.1.7 Espie et al. 1998

Nest Success – Nest Fate – Fledglings per Nest or Minimum Adult Pairs –Sample Size:

Table 1
Nest chronology, reproductive success, nest loss and nest elevations for piping plovers at Lake Diefenbaker, Saskatchewan, 1992–1993

	1992			1993				
	Mean	1 SE	n	Mean	1 SE	n	t	p
Nest initiation site	12 May	2.14	23	9 May	1.05	9	0.81	0.42
Hatch date	15 June	2.14	23	10 June	1.05	9	1.38	0.18
Fledge date	7 July	1.29	19	3 July	1.99	2	1.09	0.29
Eggs laid/pair	3.9	0.05	30	4.0	0.00	39	−1.64	0.10
Chicks hatched/pair	2.8	0.31	30	0.8	0.26	39	4.82	<0.001
Fledglings/pair	2.0	0.27	30	0.1	0.08	39	7.54	<0.001
% of nests lost to predators	20.0		30	17.9		39		
% of nests lost to flooding	3.3		30	59.0		39		
Nest elevation (m)	554.67	0.21	28	555.10	0.07	39		

F.1.8 Felio et al. 2010

Nest Fate:

Table 3. Piping plover nest success for the 2008 breeding season.

	Hatched		Failed
Known Hatch	149	Abandonment	13
Probable Hatch	9	Known Predation	41
		Flood	33
		Weather (hail or rain)	12
		Unknown	43
Total	**158**		**142**

F.1.9 Gaines and Ryan 1988

Nest Success:

Table 2. Piping plover breeding population and reproductive success at Chain-of-Lakes, North Dakota, 1984–85.

	1984	1985
No. breeding pairs	46	91
No. nests	53	97
% egg success*	34	33
% nest success*	41	42
% chick success*	77	65
No. fledged	68	95
Fledged/breeding pair	1.48	1.04

* Calculated using the Mayfield method (Mayfield 1961, Johnson 1979).

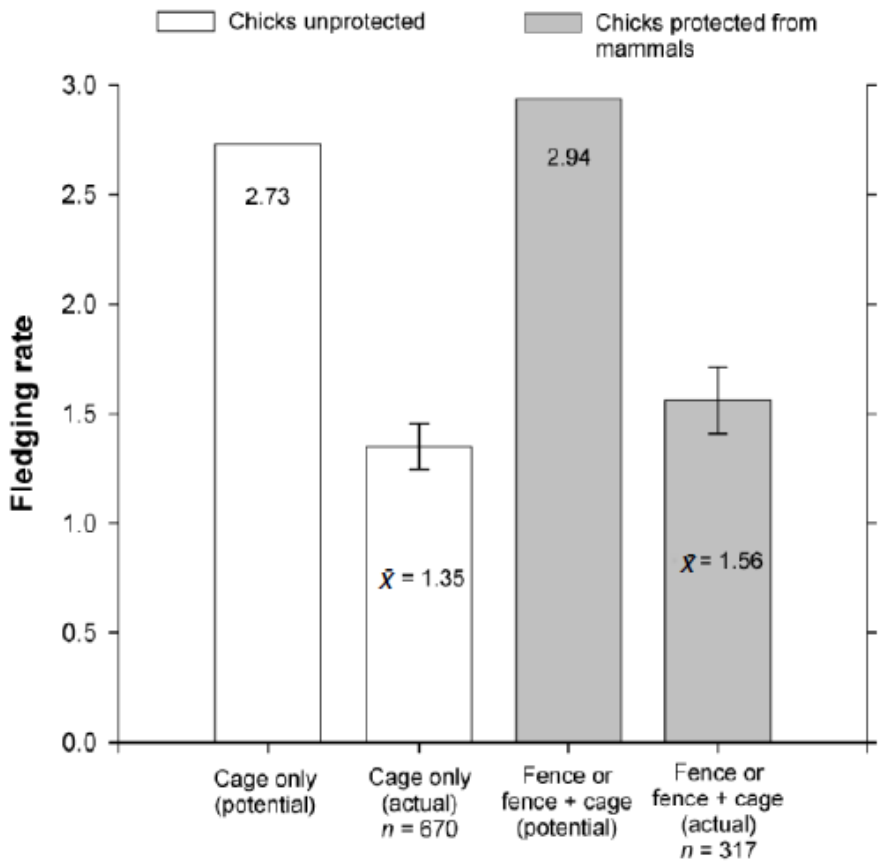
Fledglings per Nest or Minimum Adult Pairs:

Table 4. Piping plover reproductive success in the northern Great Plains, 1980–85.

Location	Year	No. pairs	Chicks fledged	Chicks fledged/pair	Source
Minn.	1982	15	26	1.7	Wiens and Cuthbert (1984), Wiens (1986)
	1983	21	44	2.1	
	1984	22	13	0.6	
Manit. I	1982	9	2	0.2	Haig (1987)
	1983	7	3	0.4	
Manit. II	1984	17	24	1.4	Haig (1987)
	1985	16	10	1.6	
Sask.	1980	11	12	1.1	Whyte (1985)
	1981	9	7	0.7	
N.D.	1984	46	68	1.5	This study
	1985	91	95	1.0	
Total		264	304		
$\bar{x} \pm$ SE				1.12 ± 0.33	

F.1.10 Ivan and Murphy 2005

Fledglings per Nest – Minimum Adult Pairs:



F.1.11 Knetter et al. 2002

Fledglings per Nest – Minimum Adult Pairs:

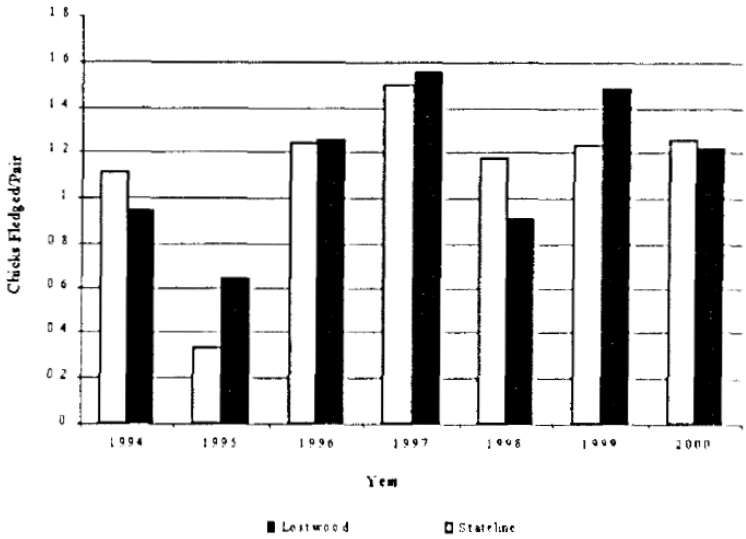


Figure 2. Number of piping plover chicks fledged/pair in the Lostwood and Stateline regions in northwest North Dakota and northeast Montana during 1994–2000 (all records).

F.1.12 Kruse et al. 2001

Nest Success – Nest Fate – Fledglings per Nest or Minimum Adult Pairs – Sample Size:

TABLE 1. Productivity summary for Piping Plovers and Least Terns nesting on the Gavins Point and Fort Randall river reaches of the Missouri River, South Dakota, 1991 and 1992.

	Piping Plover	Least Tern
Total Nests	238	354
Total Destroyed	128 (53.8%)	179 (50.6%)
Predation	61 (25.6%)	80 (22.6%)
Chicks Hatched	368	330
Chicks Fledged	57 (15.5%)	81 (24.5%)

F.1.13 Larson et al. 2002

Nest Success – Nest Fate – Fledglings per Nest or Minimum Adult Pairs – Sample Size:

Table 1. Baseline input parameters for the stochastic demographic simulation model of the female segment of the Great Plains population of piping plovers.

Parameter	Estimate	SD	CV	Source
Beginning population size, N_0	1,500			Censuses[a]
Proportion on alkaline sites, P_a	0.73			Censuses[a]
Alkaline fledging rate, F_a	0.445[b]	0.215	0.483	Table 4
River fledging rate, F_r	0.365[b]	0.176	0.483	Table 5
Annual adult survival, S_a	0.737	0.205	0.279	Larson et al. (2000)
Annual immature survival, S_i	0.48	0.13	0.279	Melvin and Gibbs (1996)

[a] Haig and Plissner (1993) and Plissner and Haig (2000*a*).
[b] Female fledglings produced per breeding female per year (i.e., [fledglings per pair]/2).

Table 2. Fledging rates of piping plovers at alkaline wetlands and freshwater lakes in the Great Plains without predator-exclusion management.

Location	$\bar{x}$[a]	SE[b]	Years	Source
Williams Preserve[c], North Dakota	0.86	0.12	13	M. R. Ryan et al.[d]
Mountrail County, North Dakota	0.45	0.12	4	Murphy et al. (2000)[e]
North Dakota–Montana state line	0.89[f]	0.24	4	Murphy et al. (2000)[e]
Sheridan County, Montana	1.04		5	Rabenberg et al. (1993)[g]
Lake of the Woods, Minnesota	1.30		4	Wiens (1986)[h], Haig and Oring (1987)
Manitoba (large lakes)	0.9	0.1	4	Haig (1987)
Big Quill Lake, Saskatoon	0.76		2	Whyte (1985)
Weighted mean/sum	0.89[i]	0.10	36	

[a] Fledglings produced per breeding pair per year.
[b] Data were not available to calculate an SE for some means.
[c] John E. Williams Memorial Nature Preserve, McLean County, North Dakota, USA.
[d] Some data were published by Prindiville Gaines and Ryan (1988) and Mayer and Ryan (1991); the rest were unpublished data, collected in 1989–1990, 1992–1996, and 1999, from M. R. Ryan and The Nature Conservancy staff.
[e] Mountrail and State Line data were kept separate for this analysis but were combined by Murphy et al. (2000).
[f] Years do not overlap with data from Rabenberg et al. (1993).
[g] Most of the data were presented earlier by Smith (1991).
[h] Some of the data were presented earlier by Wiens and Cuthbert (1984).
[i] Calculated using location means weighted by *n* years.

Table 3. Fledging rates of piping plovers at river sites and reservoirs in the Great Plains without predator-exclusion management. Data were not available to calculate a standard error for any location mean.

Location	$\bar{x}$[a]	Years	Source
Lake Diefenbaker, Saskatoon	0.49	10	Espie et al. (1998)
Gavins Point, South Dakota	0.45	7	Kruse (1993)[b]
Missouri River, North Dakota	0.86	4	Mayer (1993)
Northeastern Montana	1.61	4	Rabenberg et al. (1993)
Platte River, Nebraska	1.00	1	J. Dinan, Nebraska Game and Parks Commission, personal communication
Weighted mean/sum	0.73[c]	26	

[a] Fledglings produced per breeding pair per year.

[b] Some of the data were presented earlier by Schwalbach (1988) and Dirks (1990).

[c] Calculated using location means weighted by *n* years; SE = 0.21.

Table 4. Fledging rates of piping plovers protected with nest cages and temporary electric fencing on alkaline wetland beaches in 3 areas of northwestern North Dakota and northeastern Montana, USA.

Location	$\bar{x}$[a]	SE	*n*	Years	Breeding pairs
Nest cages					
Williams Preserve[b]	1.25	0.11	5	1995–1999	55
Mountrail County	1.24	0.15	6	1994–1999	114
North Dakota–Montana state line	1.46	0.28	4	1996–1999	138
Weighted mean	1.28[c]	0.07			
Nest cages and temporary fencing					
Williams Preserve[b]	1.39	0.52	2	1996–1997	27
Mountrail County	1.59	0.22	6	1994–1999	105
North Dakota–Montana state line	2.09	0.19	3	1996–1998	21
Weighted mean	1.78[c]	0.21			

[a] Fledglings produced per breeding pair per year.
[b] John E. Williams Memorial Nature Preserve, McLean County, North Dakota, USA.
[c] Calculated using location means weighted by SE^{-1}.

F.1.14 Maslo and Lockwood 2009

Nest Success:

Table 2
Effects of conservation treatments on hatching success (1998–2007).

Condition	N	DSR[a]	Hatching Success (%)	SE[b]	95% Confidence Interval	
					Lower	Upper
Unprotected						
NJ	522	0.95	19	0.002	0.949	0.958
Sandy Hook	77	0.92	6	0.009	0.903	0.941
Brigantine	25	0.92	6	0.017	0.881	0.950
Exclosed						
NJ	464	0.99	62	0.001	0.984	0.988
Sandy Hook	122	0.99	59	0.002	0.980	0.988
Brigantine	12	0.99	81	0.004	0.976	0.998
Electrified						
NJ	157	0.97	43	0.002	0.972	0.980
Sandy Hook	96	0.97	34	0.003	0.962	0.976
Brigantine	55	0.99	78	0.002	0.987	0.996

Sample size for Corson's Inlet is too small to obtain reliable statistical results. Sandy Hook and Brigantine nests are included in the NJ results.
[a] Daily survival rate.
[b] Standard error.

Nest Fate:

Table 3
Percentage of Abandoned Nests Under Varying Conservation Treatments.

Site	% Abandoned		
	Unprotected	Exclosed	Electrified
All NJ Breeding Beaches[b]	7 (359)[a]	19 (373)	30 (187)[d]
Sandy Hook	6 (78)	19 (123)	39 (123)
Brigantine	0 (25)	0 (12)	0 (55)
Corson's Inlet	20[c] (5)	0 (2)	0 (8)

[a] Total number of nests under each conservation treatment is indicated in parentheses.
[b] $N = 29$.
[c] Only 1 nest was abandoned at this site. Percentage is artificially inflated due to small sample size.
[d] Total number of nests includes the three study sites and one application of electric in Ocean City, NJ.

F.1.15 Maxson and Haws 2000

Nest Success – Nest Fate – Fledglings per Nest or Minimum Adult Pairs:

Table 2. Fate of Piping Plover nests at Lake of the Woods, Minnesota, 1988-2000.

Year	Total nests	Nest fate			
		Hatched	Depredated	Washed away	Abandoned
1988	13	9	4	0	0
1989	10	2	2	5	1
1990	7	3	3	1	0
1991	6	5	0	1	0
1992	5	3	1	0	1
1993	6	4	0	0	2
1994	7	5	0	0	2
1995	8	5	3	0	0
1996	9	3	2	3	1
1997	2	1	1	0	0
1998	4	3	0	0	1
1999	5	2	0	1	2
2000	6	3	0	1	2
Total	88	48	16	12	12

Table 3. Piping Plover hatching and fledging success at Lake of the Woods, Minnesota, 1982-2000[1].

Year	Eggs		Chicks		
	No. laid	% hatched	No. fledged	% fledged	No. fledged per breeding pair
1982	86	44.2	26	68.4	1.7
1983	85	75.3	44	68.7	2.1
1984	99	19.2	13	68.4	0.6
1985			7-10		0.4-0.5
1986			9		0.8
1987			2-21		0.3-3.0
1988	52	69.2	12-15	33.3-41.7	1.0-1.2
1989	32	9.4	1	33.3	0.1
1990	26	42.3	4	36.4	0.7
1991[2]	24	83.3	2-4	10.0-20.0	0.3-0.7
1992	20	60.0	4	33.3	0.8
1993	23	60.9	9	64.3	1.8
1994	26	73.1	4-7	21.1-36.8	0.7-1.2
1995	29	65.5	7-8	36.8-42.1	1.0-1.1
1996	20	65.0	4-6	30.8-46.2	0.8-1.2
1997	5	80.0	0	0.0	0.0
1998	13	65.5	7-8	87.5-100.0	2.3-2.7
1999	16	37.5	5	83.3	1.2
2000	22	45.5	7	70.0	1.4

[1]1982-1984 data from Wiens (1986), 1985-1987 data from Haig and Oring (1987).
[2]Predator exclosures placed around most nests 1991-2000.

F.1.16 McGowan et al. 2007

Nest Initiation –Nest Success – Nest Fate – Fledglings per Nest or Minimum Adult Pairs – Sample Size:

Table 1. Habitat characteristics within one square meter of four Piping Plover (*Charadrius melodus*) nests found on an island in the Missouri River between South Dakota and Nebraska, spring 2006.

Nest	No. Cottonwood saplings	No. Willow saplings	Vegetation height (m)	% Vegetative ground cover	Distance to un-vegetated habitat (m)	Nest fate
1	5	1	0.3-0.9	1-5	32.6	Failed Unknown
2	5	1	0.2-1.0	1-5	19.1	Hatched
3	9	2	0.2-1.5	1-5	68.6	Depredated Raccoon
4	1	3	0.3-1.8	1-5	13.3	Abandoned

F.1.17 Patterson et al. 1991

Nest Success – Nest Fate – Fledglings per Nest or Minimum Adult Pairs:

Table 1. Piping plover productivity on Assateague Island, 1986–87.

Area	Nests (n)	% nest success[a]	% chick survival[b]	Chicks fledged/ pair
Maryland				
Northern 11 km	53	39.4	53.8	1.11[c]
Off-road vehicle section	3	58.5	60.0	1.50[c]
Virginia				
Wild Beach	23	18.2	8.3	0.20[d]
Wash Flats	22	83.5		
Tom's Cove Hook	24	9.4	50.0	0.19[c]

[a] Calculated using Mayfield's (1975) method and a 34-day egg laying/incubating period.
[b] Number of chicks hatched/number of chicks fledged.
[c] Based on number of nesting pairs.
[d] Based on number of successful nests. Some pairs that lost nests in this area renested on Wash Flats.

Table 4. Probability of nest survival for piping plover nests on different substrates, Assateague Island, 1986–87.

Area	Nests (n)	Probability of nest survival	P[a]
Maryland nests			
Pure sand	23	0.6036	0.021
Sand, shell, cobble	33	0.2779	
Wild Beach nests (Va.)			
Pure sand	20	0.1512	0.388
Sand, shell, cobble	3	0.3837	
Wash Flat nests (Va.)			
Pure sand	7	0.7372	0.569
Sand, shell, cobble	15	0.8802	
Tom's Cove Hook nests (Va.)			
Pure sand	11	0.1679	0.326
Sand, shell, cobble	11	0.0573	

[a] Hensler's (1985) test results.

Table 5. Mean percentage of chicks fledging in piping plover broods, Assateague Island, 1986–87.[a]

	n	$\bar{x}$	SE
Type of recreational access			
ORV use allowed	5	56.7A[b]	19.44
ORV use not allowed	33	48.2A	6.92
Type of foraging habitat			
Bay	20	69.2A[c]	7.95
Tidal pool	3	66.7AB	16.67
Miscellaneous	3	22.2AB	11.11
High energy beach[d]	12	18.8B	8.59

[a] Means within a category with the same letters are not different ($P \geq 0.05$).

[b] Mann-Whitney U-test results (corrected for ties).

[c] Kruskal-Wallis (corrected for ties) and subsequent multiple comparison test results (Zar 1984).

[d] The high energy beach is the portion of the beach subject to wave action throughout the tidal cycle.

F.1.18 Rimmer and Deblinger 1990

Nest Success:

TABLE 1. Hatching success (nests hatching one or more eggs) of Piping Plover nests at Crane Beach, Ipswich, Massachusetts between 1986–1989.

Nests	n	Number hatched	Number failed
Treated	26	24 (92%)[a]	2 (8%)
Untreated	24	6 (25%)	18 (75%)
Total	50	30 (60%)	20 (40%)

[a] = Hatching success significantly different ($\chi^2 = 20.84$, $P < 0.001$) and higher than untreated nests.

F.1.19 Roche et al. 2010a

Sample Size – Years:

TABLE 1. Annual numbers of Great Lakes Piping Plover nests lost because the attendant adults disappeared or deserted the nest or because of egg predation or flooding, versus the total number of nests initiated. In addition, we list the number of uniquely marked male (M) and female (F) birds affected by each form of nest loss. For all categories of nest loss, blank spaces or values < n indicate males or females that were not uniquely banded.

		Abandonment											
		Disappearance[a]			Desertion			Egg predation			Flooding		
Year	Nests	n	M	F	n	M	F	n	M	F	n	M	F
1993	19	2			1	1		1	1	1	0		
1994	21	1		1	0			2	2	2	0		
1995	23	0			0			2	2	2	0		
1996	23	2		1	1	1	1	2	2	1	0		
1997	25	1		1	1	1	1	1		1	0		
1998	26	0			0			1	1	1	0		
1999	39	0			1	1	1	2	2	1	4	4	3
2000	34	2	1	1	0			4	1	1	0		
2001	36	1		1	0			4	2	2	0		
2002	56	5	1	5	2	1	2	4			2	1	2
2003	51	1	1		2	1	2	1	1	1	0		
2004	59	3	3	1	2	1		1	1		4	2	2
2005	58	7	3	1	1	1	1	1	1	1	1	1	
2006	60	4	1	3	1	1	1	3	2	2	3	3	2
2007	69	5	2	4	3	3	1	1		1	4	4	1
Total	599	34	12	19	15	12	10	30	18	17	18	15	10

[a]There were three instances (one each in 2002, 2004, and 2007) in which both pair members disappeared; the remaining pair members were categorized as "widowed."

F.1.20 Roche et al. 2008

Nest Success:

Table 4 – Comparative reproductive success of nesting captive-reared and wild-reared male piping plovers; p-values are from permutation tests (see text)

		Eggs laid			Chicks hatched (all nests)			Chicks hatched (successful nests)			Chicks-fledged		
	n	Mean	SD	*p*-value	Mean	SD	*p*-value	Mean	SD	*p*-value	Mean	SD	*p*-value
Captive-reared	10	3.82	0.38	0.15	2.28	1.31	0.03	2.96	1.03	0.04	1.15	1.39	0.08
Wild-reared male	57	3.91	0.25		3.10	1.17		3.56	0.68		1.80	1.28	

A successful nest was defined as a pair that fledged at least one chick.

Fledglings per Nest or Minimum Adult Pairs:

	C				W			
	1	2	3	4+	1	2	3	4+
Breeding Propensity	0.78	0.58	0.83	0.96	0.63	0.58	0.83	0.96
Fledging Rate	0.58	0.58	0.58	0.58	0.91	0.91	0.91	0.91
Stage 1 Survival	0.08	0.08	0.08	0.08	0.29	0.29	0.29	0.29
Stage 2+ Survival	0.68	0.68	0.68	0.68	0.78	0.78	0.78	0.78
Fecundity	0.04	0.23	0.33	0.38	0.17	0.41	0.59	0.68
Lambda				0.71				1.00

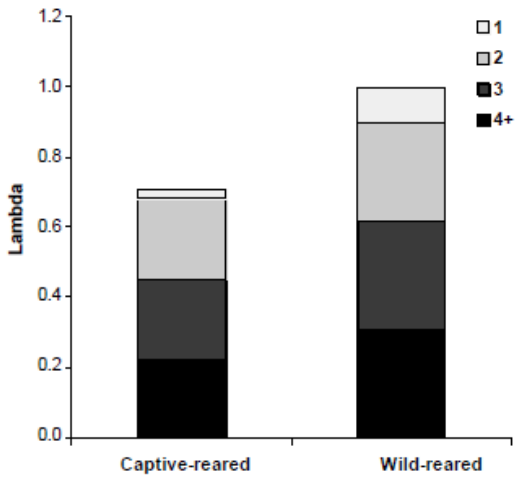


Fig. 2 – Age-specific reproductive contributions of captive-reared and wild-reared piping plovers in the first four years following hatch. Values are depicted as age-specific proportions of the total contribution to the finite rate of population change (lambda) for each group.

Sample Size – Years:

Table 1 – Summary of captive-rearing activities for Great Lakes piping plovers, 1992–2008

	Viable eggs salvaged	Eggs hatched	Chicks released	Resighted NB	(1+) B
1992	2	2	2	0	0
1993	4	3	3	0	0
1996	3	3	2	0	0
1997	4	4	4	1	2
1998	8	8	8	0	0
1999	4	2	2	0	0
2000	10	10	9	2	1
2001	4	3	3	0	0
2002	19	19	19	1	1
2003	12	12	8	1	1
2004	9	9	10	0	0
2005	18	18	15	3	0
2006	27	27	17	4	3
2007	27	16	12	0	2
2008	41	28	28	na	na
Totals	192	164	142	12	10

Non-breeding (NB) and breeding (B) captive-reared plovers resighted in subsequent years (1+) are listed according to the year they were released.

F.1.21 Roche et al. 2012

Fledglings per Nest or Minimum Adult Pairs:

TABLE 3. Flood years, fledging rates, and flood indices used to designate flood years for Piping Plover breeding areas in Saskatchewan, 2002–2008. At Big Quill Lake, the flood index was the Environment Canada water level (see Acknowledgments) at the beginning of the breeding season (1 May); at Lake Diefenbaker, the flood index was change in lake water level between the beginning (1 May) and end (1 July) of the breeding season (C. White pers. comm.); and in the Missouri Coteau (because hydrological measurements of small wetlands are not systematically collected), raw spring pond counts from the Waterfowl Breeding Survey (SK Missouri Coteau state) were used as an index of flooding events (see Acknowledgments). For each population, we standardized flooding index values (mean = 0, SD ± 1) and used these values (>0) to designate flood years. Fledging rates are the mean number of chicks fledged per individual used in our mark–recapture analysis by state and year combination; we report the sample size in parentheses.

	Big Quill Lake			Lake Diefenbaker			Missouri Coteau		
Year		Fledge rate	Meters above sea level		Fledge rate	Change (m)		Fledge rate	Number of ponds
2002		1.30 (n = 54)	514.1	Flood	0.17 (n = 104)	4.6		0.83 (n = 63)	193,792
2003		1.71 (n = 82)	514.3		1.00 (n = 122)	1.0	Flood	0.51 (n = 108)	851,096
2004		0.47 (n = 106)	514.0		0.89 (n = 149)	1.2		0.57 (n = 135)	372,215
2005		1.10 (n = 115)	513.9	Flood	0.03 (n = 172)	4.0		0.55 (n = 132)	550,857
2006	Flood	0.25 (n = 128)	515.1	Flood	0.37 (n = 131)	2.8		0.58 (n =110)	636,964
2007	Flood	0.30 (n = 20)	516.3	Flood	0.17 (n = 77)	3.1		0.00 (n = 65)	600,079
2008	Flood	NA	516.8	Flood	0.07 (n = 69)	4.3		0.00 (n = 46)	251,415

Sample Size:

TABLE 1. Total number of Piping Plover nests initiated, flooded, and moved in three Saskatchewan breeding areas. Piping Plover nests at risk of flooding were moved (translocated) out of reach of high water levels at Lake Diefenbaker. Nest searching was not conducted at Big Quill Lake and the Missouri Coteau after 2006.

Location	Year	Total nests	Number flooded	Percentage flooded	Number moved
Lake Diefenbaker	2002	139	49	35	12
	2003	95	8	8	22
	2004	100	2	2	0
	2005	150	90	60	0
	2006	104	4	4	30
	2007	97	8	8	55
	2008	95	17	18	50
Big Quill Lake	2002	30	0	0	—
	2003	33	0	0	—
	2004	43	0	0	—
	2005	40	0	0	—
	2006	39	2	5	—
Missouri Coteau	2002	55	0	0	—
	2003	66	0	0	—
	2004	57	2	4	—
	2005	54	0	0	—
	2006	56	2	4	—

F.1.22 Saunders et al. 2012

Nest Success:

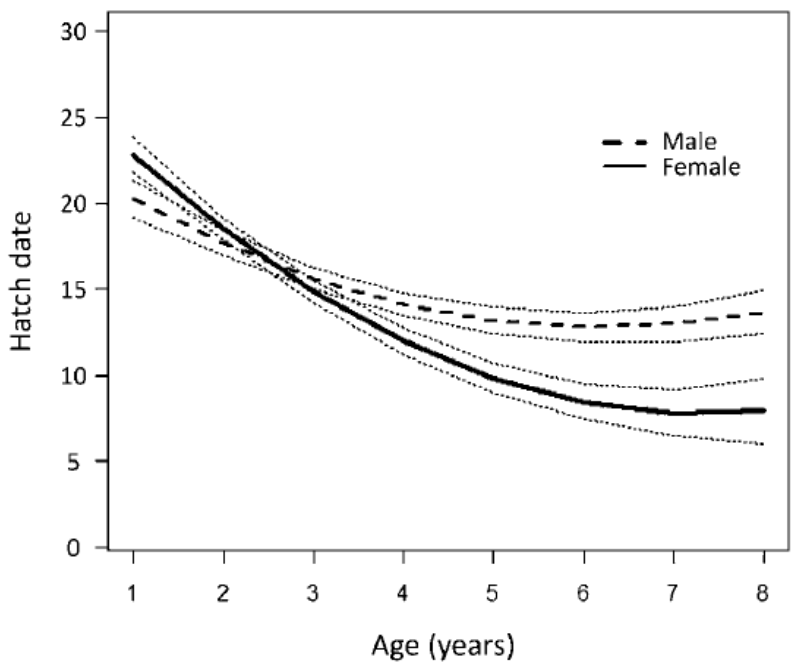


FIG. 2. Effects of female (n_f = 181) and male (n_m = 183) ages on hatch date (1 = 1 June) during 1993–2009 in the Great Lakes region, excluding renesting attempts. Solid regression line represents the linear model where hatch date was modeled as a function of male and female age, with male age held constant. The dashed regression line represents the model where female age was held constant. HDATE = $\beta_0 + \beta_1$ * agef + β_2 * agef^2 + β_3 * agem + β_4 * agem^2, where β_0 = 36.5 ± 1.83 [SE], β_1 = –5.12 ± 0.83, β_2 = 0.37 ± 0.09, β_3 = –2.75 ± 0.71, and β_4 = 0.21 ± 0.07. Dotted lines indicate ± SE.

Nest Fate:

TABLE 2. Number of brood-rearing attempts by Great Lakes Piping Plover pairs from 1993–2009, according to male and female age. Individuals appear multiple times if they nested successfully in >1 year.

	Female age										
Male age	1	2	3	4	5	6	7	8	9	10	∑
1	47	20	6	9	1	1	—	1	1	—	86
2	30	31	15	7	4	2	3	—	—	—	92
3	13	27	19	8	6	3	2	2	—	—	80
4	9	14	13	11	8	—	—	—	—	—	55
5	4	4	8	6	7	4	—	1	—	—	34
6	—	5	5	6	5	3	2	—	—	—	26
7	1	1	3	2	5	3	2	1	1	—	19
8	1	—	1	3	—	2	1	1	—	1	10
9	1	1	—	—	—	—	—	1	1	—	4
10	—	—	1	—	—	1	1	—	1	—	4
11	—	1	—	2	—	1	—	—	—	—	4
12	—	—	1	—	—	—	—	—	—	—	1
∑	106	104	72	54	36	20	11	7	4	1	

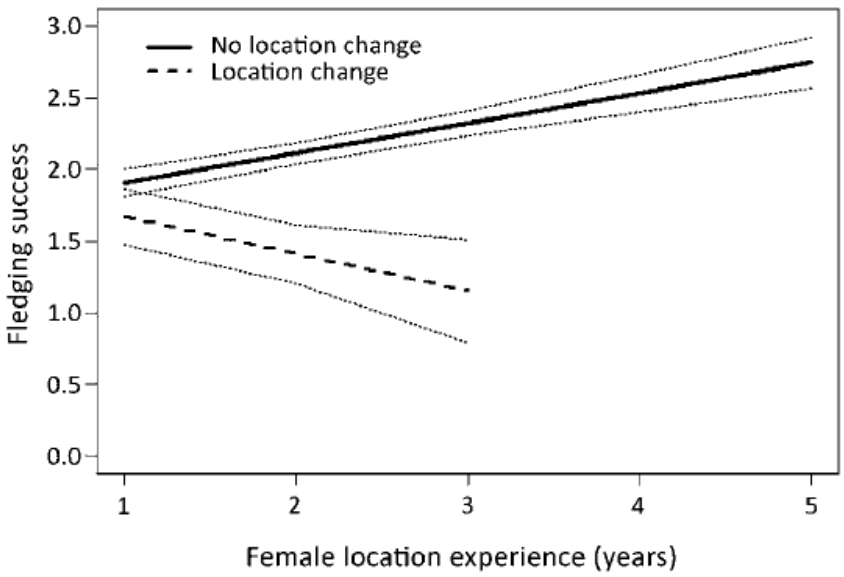
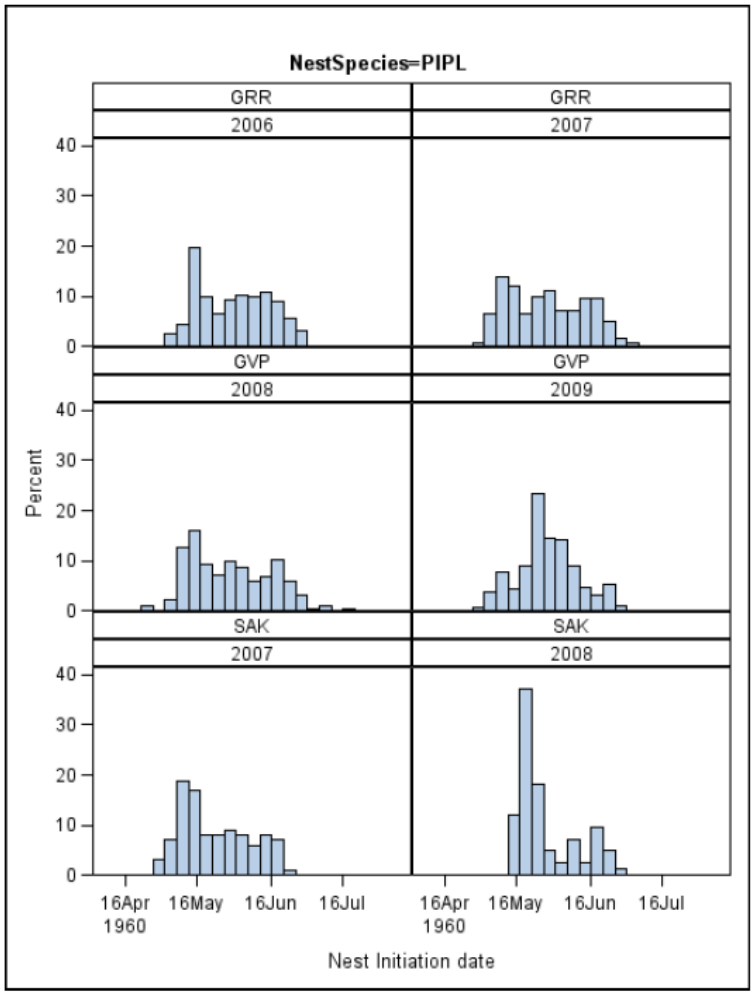


FIG. 3. Effects of female location experience and female location change on fledging success of Piping Plovers in the Great Lakes population. Solid regression line represents fledging success as a function of female location experience and the interaction term, for females that did not change locations (i.e., female location change held constant at FLOCH = 0). The dashed regression line represents fledging success for females that did change locations (i.e., female location change held constant at FLOCH = 1). Dotted lines indicate ± SE. Graphs include 95% of the observed range of variation in female location experience. FLEDGE = 0.56 ± [SE = 0.08] + 0.07 [SE = 0.03] * LEXPF + 0.11 [SE = 0.22] * FLOCH – 0.24 [SE = 0.14] * LEXPF * FLOCH.

F.1.23 Shaffer et al. 2013

Nest Initiation:



Nest Success:

Species	Study Area	Year	% Nest Success			
			USGS Estimate	USGS 95% Confidence Interval	COE Reported Apparent	COE Reported Mayfield
LETE	GRR	2006	65	54 – 73	77	76
LETE	GRR	2007	78	68 – 85	89	87
LETE	GVP	2008	66	59 – 73	75	69
LETE	GVP	2009	73	64 – 80	79	76
PIPL	GRR	2006	21	16 – 27	54	46
PIPL	GRR	2007	47	39 – 54	73	65
PIPL	GVP	2008	42	35 – 48	67	53
PIPL	GVP	2009	41	34 – 49	54	46
PIPL	SAK	2007	25	17 – 33	41	12
PIPL	SAK	2008	28	19 – 37	44	16

Fledglings per Nest or Minimum Adult Pairs:

Species	Study Area	Year	Reported by COE	Low Renesting			High Renesting		
				USGS Expanded Estimate	USGS 95% Confidence Interval	Relative Bias (%)	USGS Expanded Estimate	USGS 95% Confidence Interval	Relative Bias (%)
LETE	GRR	2006	0.81	0.78	0.51–1.17	4	0.90	0.59 – 1.35	-10
LETE	GRR	2007	1.06	0.80	0.46–1.32	33	0.89	0.52 – 1.48	18
LETE	GVP	2008	1.14	1.23	0.98–1.50	-7	1.37	1.09 – 1.68	-17
LETE	GVP	2009	1	1.43	1.16 – 1.75	-30	1.61	1.30 – 1.97	-38
PIPL	GRR	2006		NA			NA		
PIPL	GRR	2007	0.97	0.96	0.46 – 1.77	1	1.12	0.54 – 2.06	-14
PIPL	GVP	2008	1.37	0.89	0.76 – 1.05	54	1.03	0.88 – 1.21	33
PIPL	GVP	2009	1.09	0.60	0.57 – 0.63	83	0.68	0.65 – 0.72	60
PIPL	SAK	2007	0.7	0.34	0.04 – 1.11	107	0.42	0.06 – 1.37	68
PIPL	SAK	2008	0.68	0.27	0.05 – 0.83	155	0.32	0.06 – 0.98	116

F.1.24 Sherfy et al. 2008

Nest Initiation:

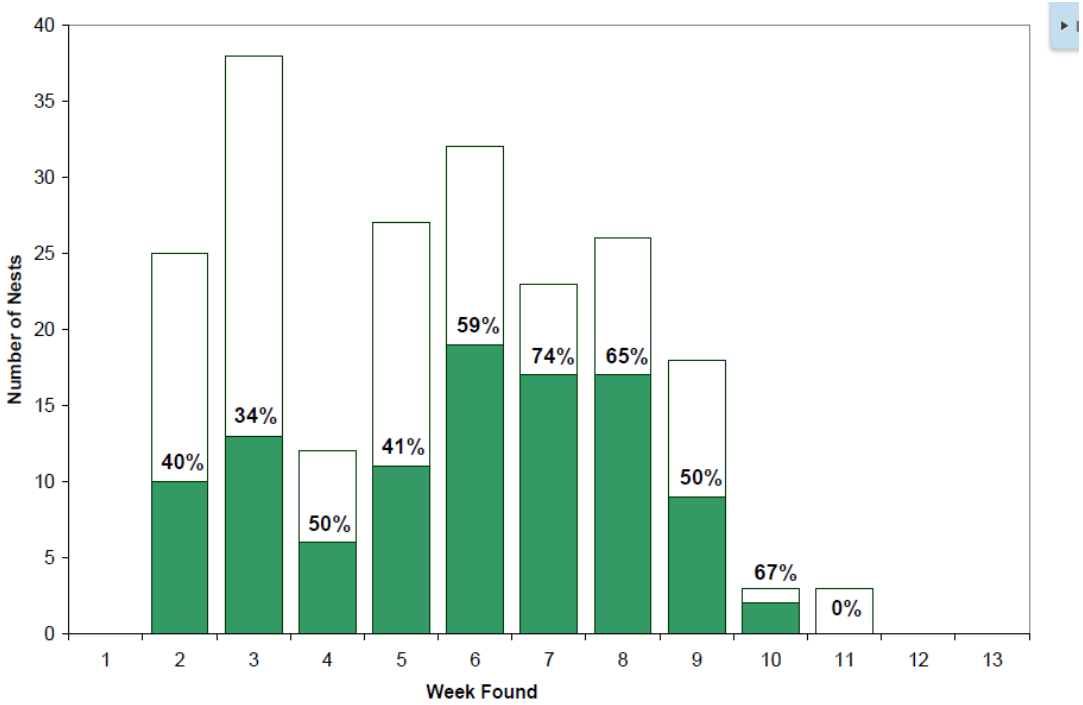


Figure 9. Number of piping plover nests found on the Garrison Reach by week and the ultimate fate of those nests. Dark bars indicate number of successful (known or probable) nests and open (white) bars indicate number of unsuccessful (known or probable) nests. Percentages indicate percent of nests found during that week that ultimately produced at least 1 hatchling. Week 1 = 29 April - 5 May; Week 13 = 22 July - 28 July, 2007.

Next Success:

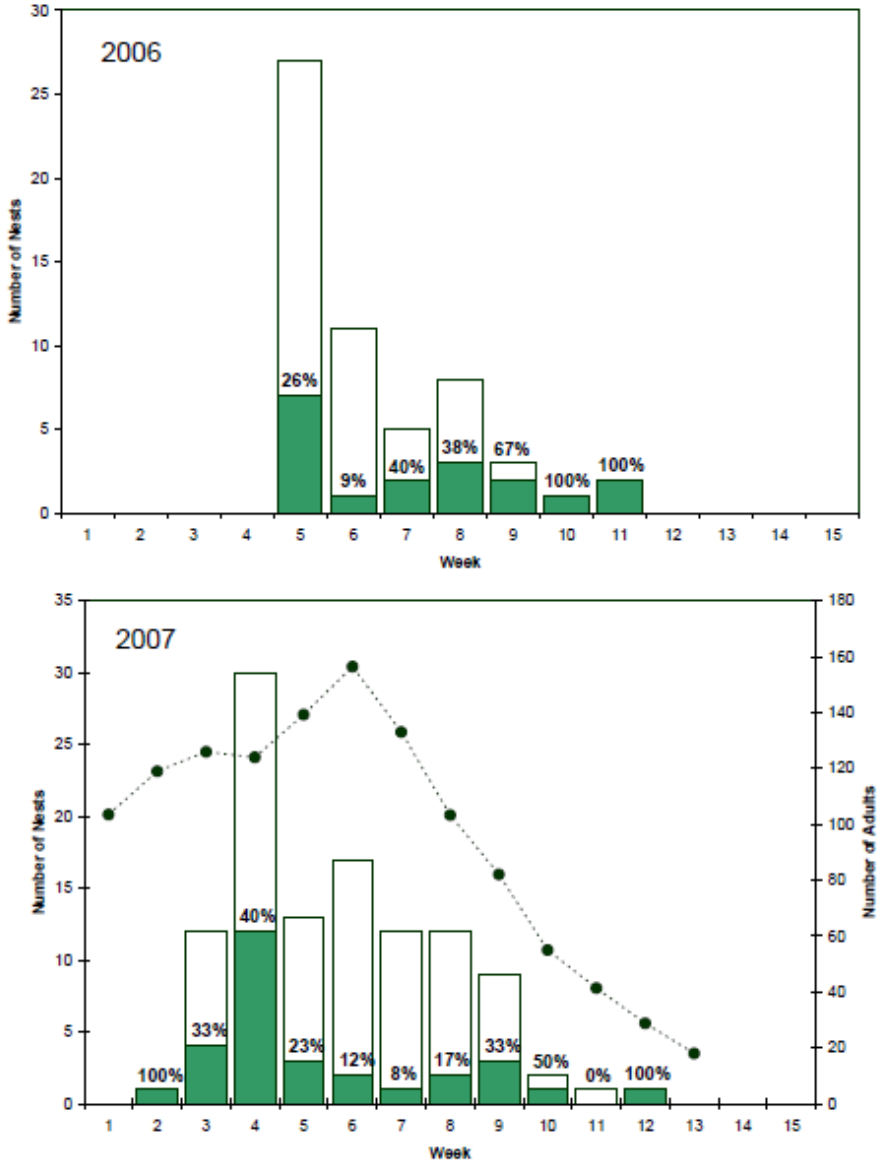


Figure 13. Number of piping plover nests found on Lake Sakakawea by week and the ultimate fate of those nests for 2006 and 2007. Dark bars indicate number of successful (known or probable) nests and open (white) bars indicate number of unsuccessful (known or probable) nests. Percentages indicate percent of nests found during that week that ultimately produced at least 1 hatchling. The black circles represent adult counts from weekly surveys during 2007 (see Table 28). Week 1 = 30 April - 6 May, 2006 and 27 April - 3 May, 2007.

Nest Fate:

Table 24. Numbers and fates of least tern and piping plover nests from intensive surveys of 17 4-mile river segments on the Garrison Reach during 2007.

	Nest fate					
	Nests	Successful	Probable successful	Probable Failure	Failure	Unknown
Least tern	112	45	28	0	23	16
Piping plover	207	44	64	0	86	13
Total	319	89	92	0	109	29

F.1.25 Sherfy et al. 2009a

Nest Initiation:

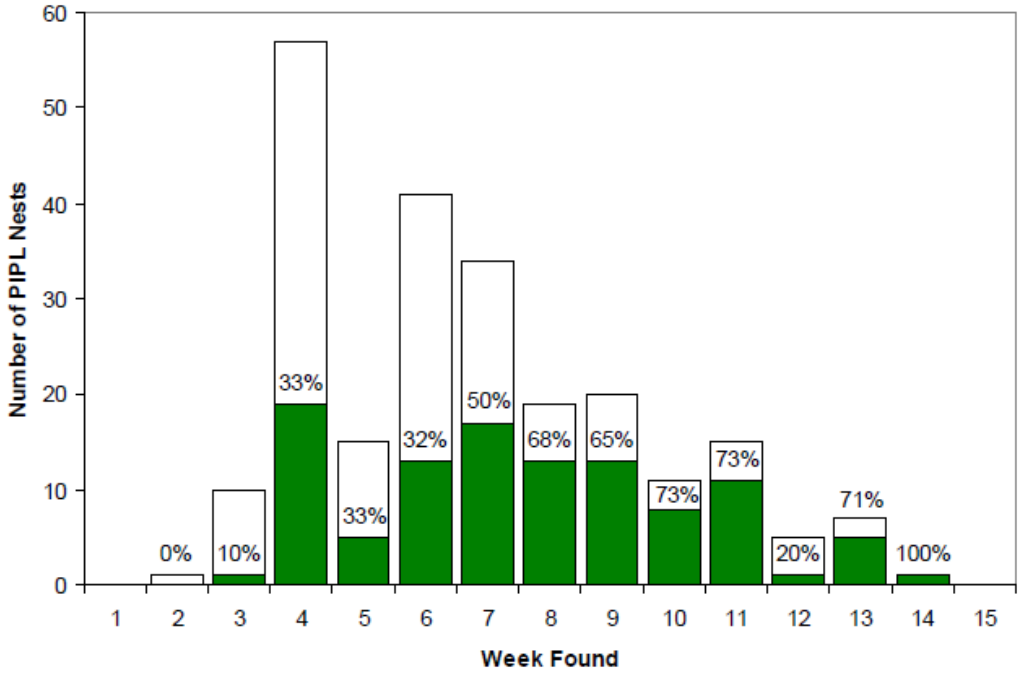


Figure 9. Number of piping plover nests found on the Gavins Point Reach by week and the ultimate fate of those nests. Dark bars indicate number of successful (known or probable) nests and open (white) bars indicate number of unsuccessful (known or probable) nests. Percentages indicate percent of nests found during that week that ultimately produced at least one hatchling. Week 1 = 20 April to 26 April; Week 15 = 27 July to 2 August, 2008.

Nest Success – Nest Fate:

Table 12. Numbers and fates of least tern and piping plover nests on created and natural sandbars on the Gavins Point Reach (GVP) and Lewis and Clark Lake (LCL) in 2008.

Species	Study Area	Sandbar Type	Total Nests	Nest Fate				
				Successful	Probable Successful	Probable Failure	Failure	Unknown
Least Tern	GVP	Mechanically Created	155	80	25	0	38	12
		Natural	41	15	5	0	19	2
	LCL	Mechanically Created	140	58	38	0	31	13
Piping Plover	GVP	Both	237	94	13	0	108	22
Total			537	247	81	0	196	49

F.1.26 Sherfy et al. 2009b

Nest Initiation:

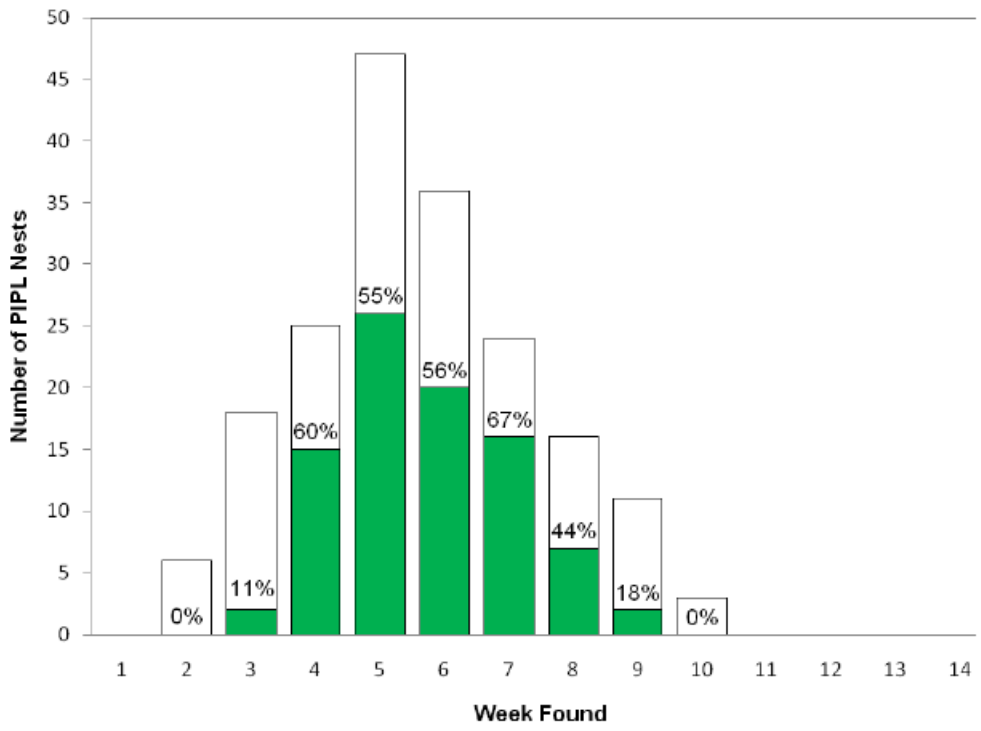


Figure 5. Number of piping plover nests found on the Gavins Point Reach by week and the ultimate fate of those nests. Dark bars indicate number of successful (known or probable) nests and open (white) bars indicate number unsuccessful (known or probable) nests. Percentages indicate percent of nests found during the week that ultimately produced at least one hatchling. Week 1 = 26 April to 2 May; Week 14 = 26 July to 1 August, 2009.

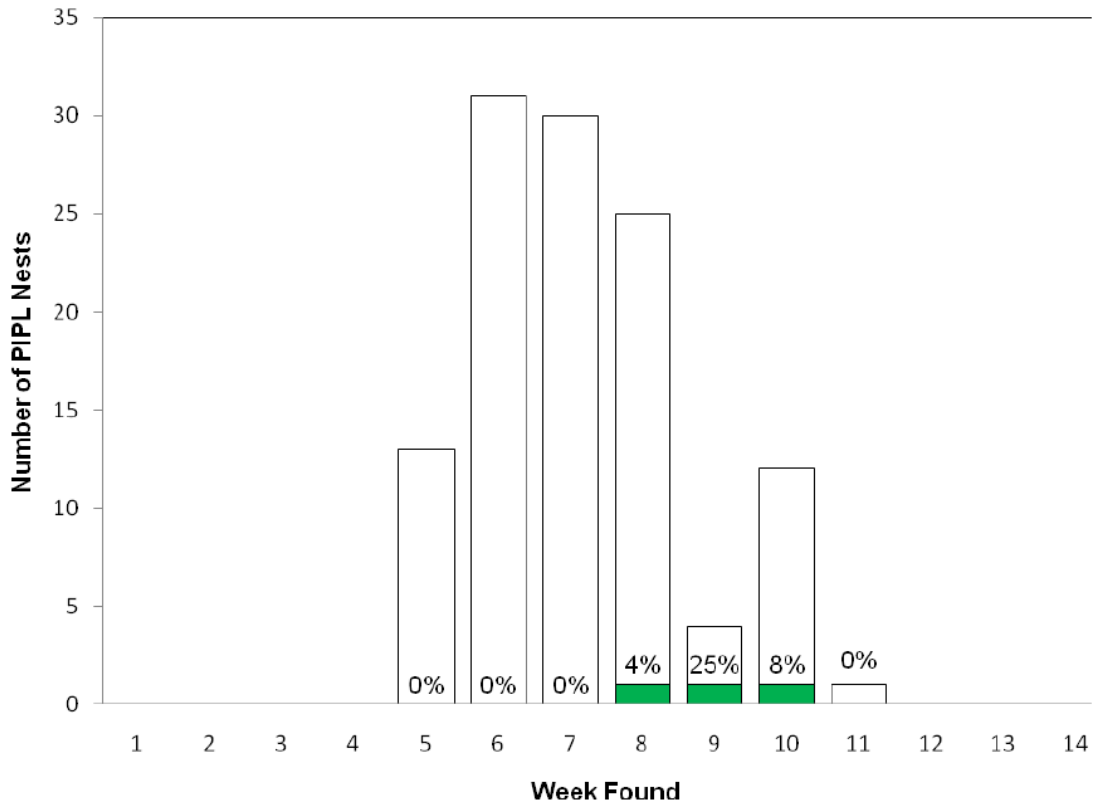


Figure 9. Number of piping plover nests found on Lake Sakakawea by week and the ultimate fate of those nests. Dark bars indicate number of successful (known or probable) nests and open (white) bars indicate number unsuccessful (known or probable) nests. Percentages indicate percent of nests found during the week that ultimately produced at least one hatchling. Week 1 = 26 April to 2 May; Week 14 = 26 July to 1 August, 2009.

Nest Success – Nest Fate:

Table 7. Numbers and fates of least tern and piping plover nests on the Gavins Point Reach in 2009.

		Nest Fate				
	Total Nests	Successful	Probable Successful	Probable Failure	Failure	Unknown
Least Tern	136	91	6	1	33	5
Piping Plover	194	72	16	0	98	8
Total	330	163	22	1	131	13

F.1.27 Tremblay et al. 1994

Nest Success – Nest Fate – Fledglings per Nest or Minimum Adults Pairs – Sample Size:

Kouchibouguac National Park, New Brunswick.

Year	Territorial pairs	Number of nests	Hatching rate[a]	Fledging rate[a]
1982	16	16	2.7[b]	–
1983	21	21	2.8[c]	2.1[d]
1984	11	4	–	–
1985	12	3	–	–
1986	15	12	–	–
1987	15	15	1.2	0.93
1988	13	9	2.6	2.2
1989	16	16	2.6	2.1
1990	15	15	3.4	1.9
1991	17	16	1.7	1.2
1992	15	13	3.3	2.3
1988–1992	15.2	13.8	2.5	1.94

F.1.28 Wiens and Cuthbert 1988

Nest Success – Nest Fate – Fledglings per Nest or Minimum Adult Pairs:

TABLE 3

RELATIONSHIP BETWEEN MATE RETENTION AND PREVIOUS REPRODUCTIVE SUCCESS IN LAKE OF THE WOODS PIPING PLOVERS

Reproductive success in original season	Number of pairs	Fate of pairs in subsequent season			
		Reunite	Separate	Discontinue	Status unknown
Succeeded	27	5	3	12	7
Failed	9	0	3	4	2

Sample Size:

TABLE 1

PIPING PLOVER POPULATION SIZE AT FOUR SITES AT LAKE OF THE WOODS

	Year		
	1982	1983	1984
Breeding adults			
Pine and Curry Island	24	32	36
Morris Point	4	6	8
Zippel Bay	0	2	0
Rocky Point	2	2	0
Non-breeding adults	14	7	3
Total	44	49	47

F.1.29 Dirks 1990

Nest Success – Nest Fate:

Table 4. Piping plover nest fates along the Missouri River in South Dakota during 1988 and 1989.

	Gavins Pt.		Ft. Randall		Total 1988		Total 1989	
	1988	1989	1988	1989	N	(%)	N	(%)
Nests with Known Fate	139	103	7	9	146	(100)	112	(100)
No. Hatched	42	36	1	5	43	(26)	41	(34)
No. Destroyed or Abandoned	97	67	6	4	103	(61)	71	(59)
Cause of Destruction/Abandonment								
Predator	25	27	0	2	25	(24)	29	(41)
Flooding	4	1	3	1	7	(7)	2	(3)
Human Disturbance	4	1	0	0	4	(4)	1	(1)
Weather	5	1	0	0	5	(5)	1	(1)
Sandbar Erosion	4	1	0	0	4	(4)	1	(1)
Unknown Agent	55	36	3	1	58	(56)	37	(52)
Fate Unknown	19	9	3	0	22	(13)	9	(7)
Total No. Nests Found	158	112	10	9	168	(100)	121	(100)

F.2 Invertebrate Prey

F.2.1 Anteau and Sherfy 2010

Findings:

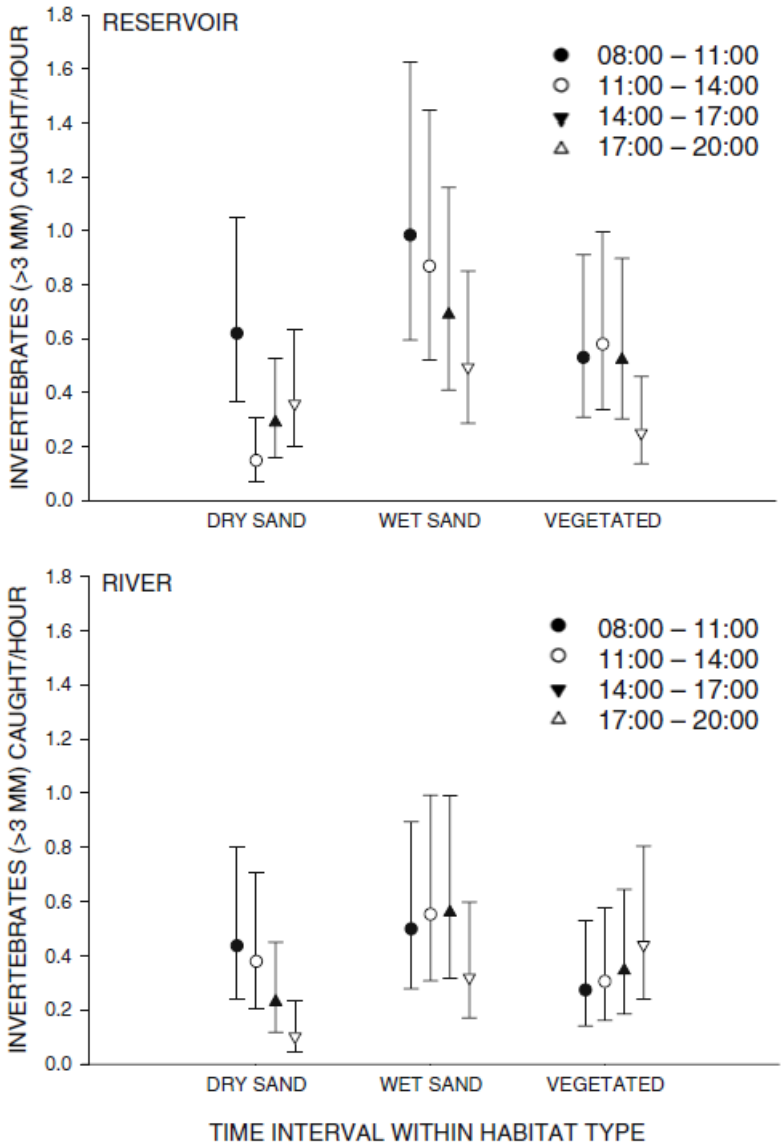


Fig. 1 Least-squares geometric mean (± 95% CL) counts per hour of large invertebrates (>3 mm) captured on sticky traps during four time intervals within three habitat types at reservoir (Lake Sakakawea) and riverine (Garrison Reach) study areas of the Missouri River in summers 2006 and 2008. Significant differences among estimates generally are indicated by a 95% CL not overlapping the point estimate of another level

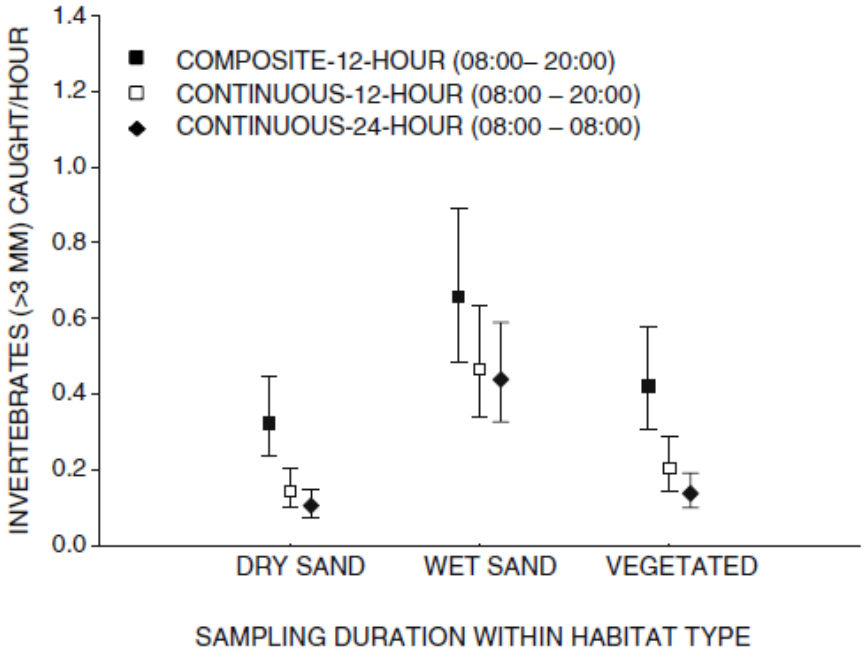


Fig. 2 Least-squares geometric mean (±95% CL) counts per hour of large invertebrates (>3 mm) captured on sticky traps deployed for three durations by three habitat types for reservoir (Lake Sakakawea) and riverine (Garrison Reach) study areas of the Missouri River combined in summers 2006 and 2008. Significant differences among estimates generally are indicated by a 95% CL not overlapping the point estimate of another level

F.2.2 Catlin et al. 2012

Findings:

Table 1. Mean (± 1 SE) piping plover foraging habitat[a] on natural and artificial sandbars of various ages on the Missouri River, 2005–2009.

Sandbar type	Sandbar age (yrs)	*N*[b]	Mean foraging area (ha)
Natural	ca. 8–12	47	3.89 ± 0.77
Artificial	0	10	11.62 ± 2.16
	1	8	4.62 ± 1.85
	2	5	4.81 ± 3.10
	3	4	3.62 ± 2.30
	4	3	2.16 ± 1.09
	5	1	0.09

[a] Foraging area is open or sparsely vegetated wet sand; [b] The sampling unit for this study was sandbar within year.

Table 2. Mean (± 1 SE) number of arthropods collected during the chick-rearing period (June–August) in piping plover foraging habitats on natural sandbars and artificial sandbars of various ages on the Missouri River, 2005–2009. We present data separately for arthropods collected on sticks and arthropods collected in core samples.

	Natural	Artificial (age in yrs)					
Sample type	ca. 8–12 yrs.	0	1	2	3	4	5
Sticky traps	5.75 ± 0.65	2.93 ± 0.58	3.13 ± 0.45	3.35 ± 0.47	3.58 ± 0.69	3.83 ± 1.03	4.09 ± 1.45
Core samples	20.40 ± 4.00	13.04 ± 4.21	11.86 ± 2.84	10.79 ± 2.55	9.81 ± 3.08	8.93 ± 3.84	8.12 ± 4.57

Table 4. Number (%)[a] of arthropods collected on sticky traps in piping plover foraging habitat (natural and artificial sandbars) on the Missouri River, 2005– 2009[b].

	Natural	Artificial (age in yrs)					
	ca 8–12	0	1	2	3	4	5
Araneae	158 (0.9)	11 (0.4)	15 (0.4)	11 (0.6)	14 (1.4)	11 (1.2)	0 (0.0)
Coleoptera	207 (1.2)	47 (1.8)	51 (1.4)	64 (3.5)	16 (1.6)	8 (0.9)	3 (3.1)
Diptera	10,272 (58.3)	2,353 (89.8)	3,242 (86.7)	1,340 (73.9)	788 (79.7)	815 (89.9)	87 (89.7)
Hemiptera	603 (3.4)	72 (2.7)	59 (1.6)	65 (3.6)	20 (2.0)	37 (4.1)	0 (0.0)
Homoptera	589 (3.3)	44 (1.7)	52 (1.4)	23 (1.3)	28 (2.8)	2 (0.2)	0 (0.0)
Hymenoptera	273 (1.6)	30 (1.1)	29 (0.8)	19 (1.0)	12 (1.2)	5 (0.6)	4 (4.1)
Orthoptera	1,677 (9.5)	13 (0.5)	18 (0.5)	24 (1.3)	52 (5.3)	6 (0.7)	0 (0.0)
Collembola	3,007 (17.1)	10 (0.4)	238 (6.4)	252 (13.9)	44 (4.4)	5 (0.6)	0 (0.0)
Unknown	806 (4.6)	22 (0.8)	32 (0.9)	13 (0.7)	15 (1.5)	18 (2.0)	2 (2.1)
Other	16 (0.1)	18 (0.7)	4 (0.1)	3 (0.2)	0 (0.0)	0 (0.0)	1 (1.0)
Total	17,608	2,620	3,740	1,814	989	907	97

[a] Percent of total individuals collected per sandbar type and age; [b] Chi-square test of equal proportions: $\chi^2_{54} = 2901.6$, $P < 0.001$.

Table 5. Number (%)[a] of arthropods collected in sediment cores in piping plover foraging habitat (natural and artificial) on the Missouri River, 2005 – 2009[b].

	Natural	Artificial (age in yrs)					
	ca 8–12	0	1	2	3	4	5
Annelids	6,001 (23.0)	1,008 (24.8)	692 (12.5)	445 (13.4)	102 (9.9)	107 (6.8)	13 (4.3)
Coleoptera	1,565 (6.0)	155 (3.8)	474 (8.6)	264 (7.9)	111 (10.8)	60 (3.8)	14 (3.8)
Diptera	14,916 (57.2)	2369 (58.2)	3993 (72.0)	2181 (65.6)	686 (66.9)	1,323 (84.4)	235 (78.3)
Eggs	902 (3.4)	180 (4.4)	118 (2.1)	68 (2.0)	41 (4.0)	13 (0.8)	0 (0.0)
Mollusca	259 (1.0)	87 (2.1)	20 (0.4)	3 (0.0)	2 (0.2)	8 (0.5)	1 (0.3)
Copepods	90 (0.3)	46 (1.1)	68 (1.2)	30 (0.9)	0 (0.0)	9 (0.6)	13 (4.3)
Other	2,323 (8.9)	224 (5.5)	177 (3.2)	330 (9.9)	84 (8.2)	47 (3.0)	24 (8.0)
Total	26,057	4,068	5,542	3,321	1,026	1,567	300

[a] Percent of total individuals collected per sandbar type and age; [b] Chi-square test of equal proportions: $\chi^2_{36} = 1790.8$; $P < 0.001$.

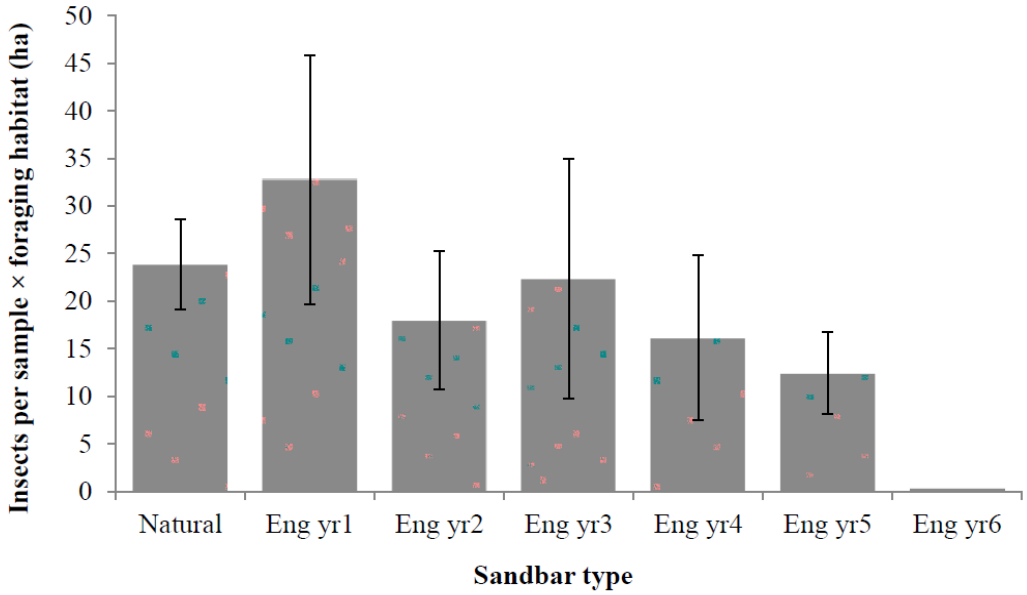


Figure 2. Predicted number of arthropods sampled on Missouri River sandbars times the average amount of foraging habitat available on artificial and natural sandbars, 2005–2009. Error bars represent 1 SE, calculated using the Delta Method (Powell 2007).

F.2.3 Catlin et al. 2013

Findings:

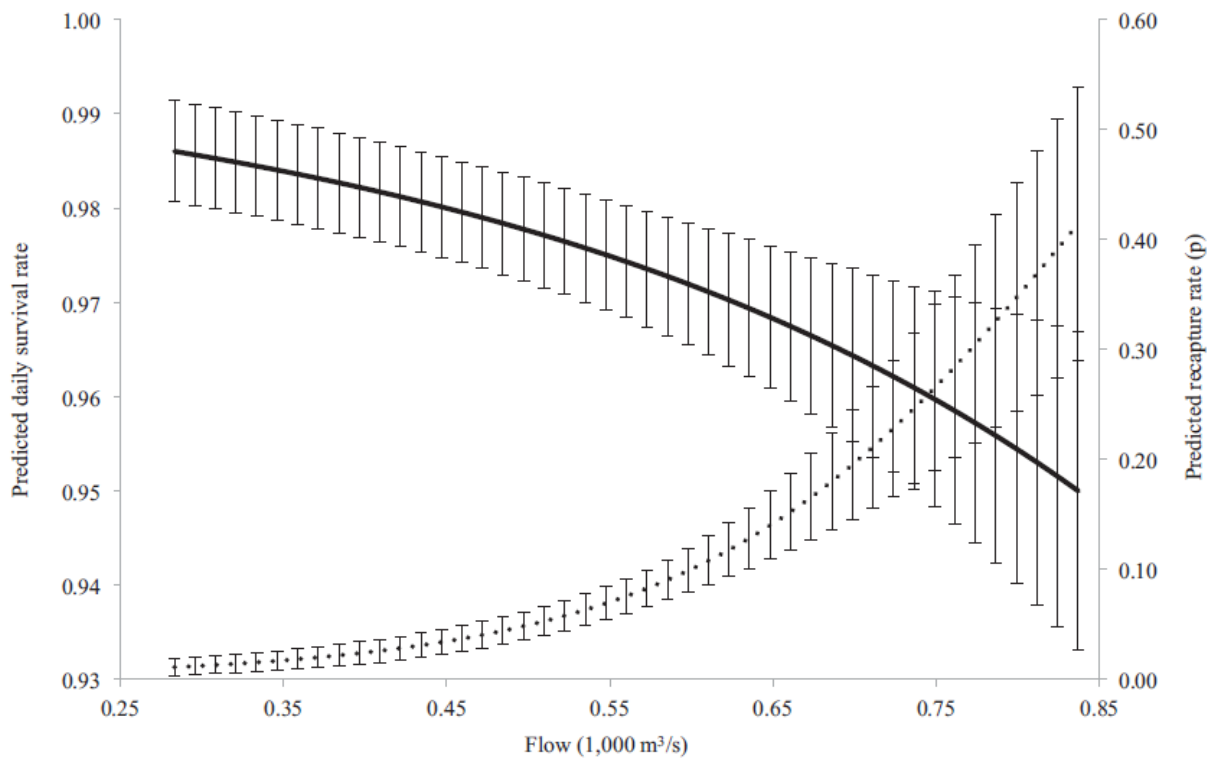


Figure 4. Average predicted daily survival rates (solid line) and recapture rates (dotted line) for piping plover chicks (26 days post-hatch; $n = 1{,}099$) on the Missouri River relative to river flow from the Gavins Point Dam, 2006–2009. Predictions are from the top-ranked (weight = 0.884) model of survival and recapture rate. We created 10,000 Monte Carlo estimates using the β estimates and variance–covariance matrices from Program MARK. The estimates are the means of those 10,000 estimates and the error bars represent 1 SD.

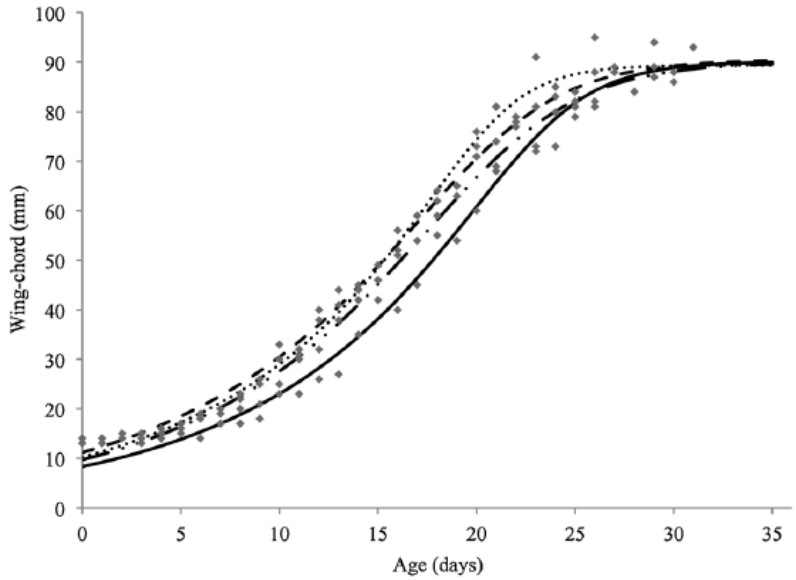


Figure 7. Predicted growth of piping plover chick wing-chords (mm) on the Missouri River (n = 1,099) in 2006 (solid line), 2007 (dotted line), 2008 (dashed line), and 2009 (dotted and dashed line). The raw data are also pictured (gray diamonds).

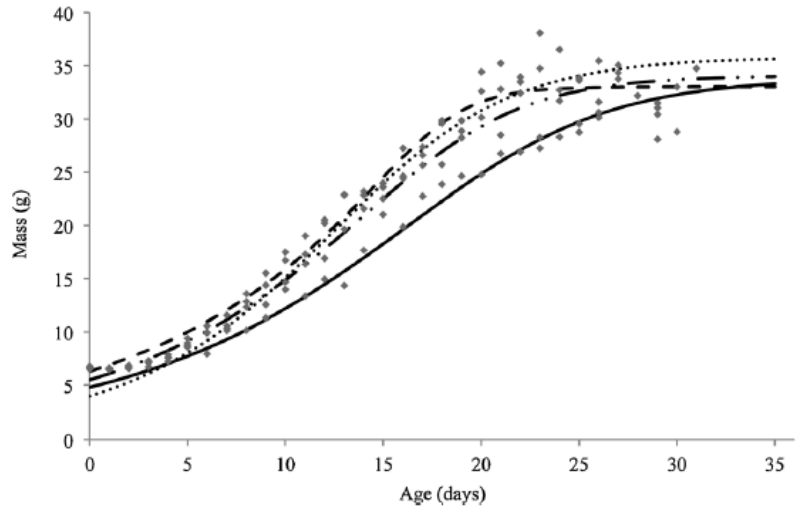


Figure 8. Predicted growth of piping plover chick mass (g) on the Missouri River in 2006 (solid line), 2007 (dotted line), 2008 (dashed line), and 2009 (dotted and dashed line). The raw data are also pictured (gray diamonds).

F.2.4 Le Fer et al. 2008b

Sample Size:

Table 3. Invertebrate biomass (Mean mg dry weight per set), and number in saturated and moist habitats, along protected and exposed shoreline at the three Missouri River reaches in North and South Dakota, during the chick-rearing period, 2001-2003. Results of ANOVA on ranks comparing exposed and protected shoreline.

		Biomass				Number			
Reach	N	Mean ± SE	Mean Rank	F_1	P	Mean ± SE	Mean Rank	F_1	P
Epilimnetic									
Protected	24	19.5 ± 3.7	23.6	4.2	0.05	13.1 ± 2.1	24.9	11.9	0.002
Exposed	12	11.2 ± 3.7	15.8						
		6.1 ± 1.7	12.8						
Hypolimnetic									
Protected	16	4.3 ± 1	17.3	0.03	0.87	6 ± 0.8	17.0	0	1.0
Exposed	17	3.7 ± 0.7	16.7			6.1 ± 0.9	17.0		
Reservoir									
Protected	12	4.9 ± 1	13.5	0.3	0.6	7.5 ± 1.3	14.6	0	0.99
Exposed	14	8.8 ± 3.2	15.2			9.2 ± 2.3	14.6		

Table 2. Invertebrate abundance and biomass (mean mg dry weight per set) ± SE in habitats during chick-rearing period in North Dakota and South Dakota, 2001-2003. Results of ANOVA on ranks followed by Fisher's LSD for significant effects ($P \leq 0.05$). Habitats with the same letters are not significantly different. N = Number of brood foraging sites.

		Biomass[a]			Numbers[b]		
Habitat	N	Mean ± SE	F_5	P	Mean ± SE	F_5	P
Hypolimnetic							
Saturated	24	4.2 ± 1.2 A	4.6	0.005	4.1 ± 0.6 BC	5.3	0.003
Moist	24	4.5 ± 1.0 A			7.6 ± 1.1 A		
Dry	20	2.1 ± 0.7 B			3.1 ± 0.6 C		
Vegetation	17	4.5 ± 1.0 A			6.5 ± 1.0 AB		
Epilimnetic							
Saturated	28	24.8 ± 3.8 A	11.4	< 0.001	14.9 ± 3.5 A	11.4	< 0.001
Moist	28	7.9 ± 1.6 B			7.4 ± 0.9 AB		
Dry	16	10.6 ± 5.1 B			3.0 ± 1.0 C		
Vegetation	24	6.5 ± 1.4 B			5.2 ± 0.7 B		
Reservoir							
Saturated	21	10.9 ± 2.3 A	14.8	< 0.001	11.5 ± 1.7 A	21.1	< 0.001
Moist	20	2.9 ± 0.7 B			5.1 ± 0.7 B		
Dry	16	1.7 ± 0.5 B			2.6 ± 0.6 C		
Vegetation	21	2.2 ± 0.8 B			2.3 ± 0.3 C		

[a]Two Way ANOVA on ranks: habitat: $F_{5,247} = 22.8$, $P < 0.001$; reach: $F_{2,247} = 3.1$, $P = 0.05$; habitat x reach: $F_{6,247} = 7.5$, $P < 0.001$.

[b]Two Way ANOVA on ranks: habitat: $F_{5,247} = 23.1$, $P < 0.001$; reach: $F_{2,247} = 12.1$, $P < 0.001$; habitat x reach: $F_{6,247} = 5.1$, $P < 0.001$.

F.3 Nest Site Selection

F.3.1 Burger 1994

Finding:

TABLE 2. Comparison of foraging piping plovers on the three study sites (1988 and 1989) in New Jersey. Each sample was 2-min observation of a Foraging Plover. Given are x ± SD.

	Brigantine	Holgate	Corson's Inlet	Wilcoxon χ^2 (p)
Number of samples	801	821	579	
Seconds spent:				
Feeding	73.2 + 28.3	106.1 ± 21.7	85.0 ± 37.2	168.9 (0.0001)
Alert	24.4 ± 20.6	7.3 ± 40.7	13.4 ± 18.2	253.8 (0.0001)
Conspecific aggression	0.2 ± 2.2	0.4 ± 2.4	1.3 ± 6.9	38.0 (0.0001)
People within 100 m	1.9 ± 2.4	0.1 ± 0.5	2.8 + 2.1	520.4 (0.0001)
Number of people				
Fishing	0.05 + 0.6	0	0.04 ± 0.3	151.5 (0.0001)
Sunbathing	0.26 ± 0.9	0	0.33 ± 2.1	151.3 (0.0001)
Walking	2.12 ± 1.1	0	2.26 ± 2.8	49.8 (0.0001)
Number of feeding birds	2.3 ± 2.8	0.9 ± 2.5	3.4 + 8.5	589.2 (0.0001)

F.3.2 Cohen et al. 2008

Finding:

TABLE 1. Characteristics at Piping Plover nests and random sites, Westhampton Island, New York, 2004 (n = 32, 1-m^2 nest and paired random plots within 50 m).

	Vegetation				Coarse grains				
	Cover (%)				Cover (%)				
Plot type	$\bar{x}$	SE	Max	Plots in[a] >5% cover	$\bar{x}$	SE	Max	Plots in[a] >5% cover	Plots near[b] ≥ 1 large object
Nest	7.6	1.7	45.7	0.41	9.1	2.6	55.6	0.41	0.28
Random	4.4	1.9	48.6	0.19	5.5	2.2	47.2	0.22	0.16

[a] Mean of the index used in our nest site selection model (0 = < 5% cover, 1 = 5–~100% cover).

[b] Mean of the large objects (rock, shell, wood >64 mm in at least one dimension) index used in our nest site selection model (0 = no large objects, 1 = ≥ 1 large object). Where large objects were present, counts ranged from 1 to 5.

F.3.3 Flemming et al. 1992

Finding:

Table 3. Variety of nest types used by piping plovers in New Brunswick and Nova Scotia, 1983, 1987, 1988, 1989.

Nest type	No. of nests		
	Eastern New Brunswick	Northern Nova Scotia	Southern Nova Scotia
Pebble	12	9	14
Grass	4	1	34
Mixed	3	1	16
Sand	0	0	9

F.3.4 Marcus et al. 2007

Finding:

Table 3. Substrate composition and vegetation cover at Least Tern (LT) or Piping Plover (PP) nests (representing what the birds selected) and at random points within used plots (representing what was available to them in the area around their nest) at gravel mines along the Platte and Elkhorn rivers, Nebraska, 2000 and 2001. A used plot is defined as a plot with at least one nest initiated in it. Values with the same letter within a row are not significantly different (P > 0.05).

		Attractant plots		Control plots	
		Nests	Random	Nests	Random
Fine Sand	LT	16.8 A	13.4 AB	16.1 A	8.5 B
	PP	18.7 A	17.2 A	19.8 A	12.5 B
Coarse Sand	LT	56.5 A	53.5 A	64.9 A	85.4 B
	PP	43.2 A	54.7 A	43.3 A	79.2 B
Small Gravel	LT	18.9 A	23.3 A	9.0 AB	3.0 B
	PP	28.4 A	20.4 A	29.6 A	17.6 A
Large Gravel	LT	7.9 A	10.0 A	9.5 A	3.0 B
	PP	10.0 A	6.2 A	11.6 A	1.6 A
Vegetation	LT	2.2 A	3.1 A	1.5 A	2.0 A
	PP	4.6 A	1.7 A	3.2 A	3.0 A

F.3.5 Maslo et al. 2011

Finding:

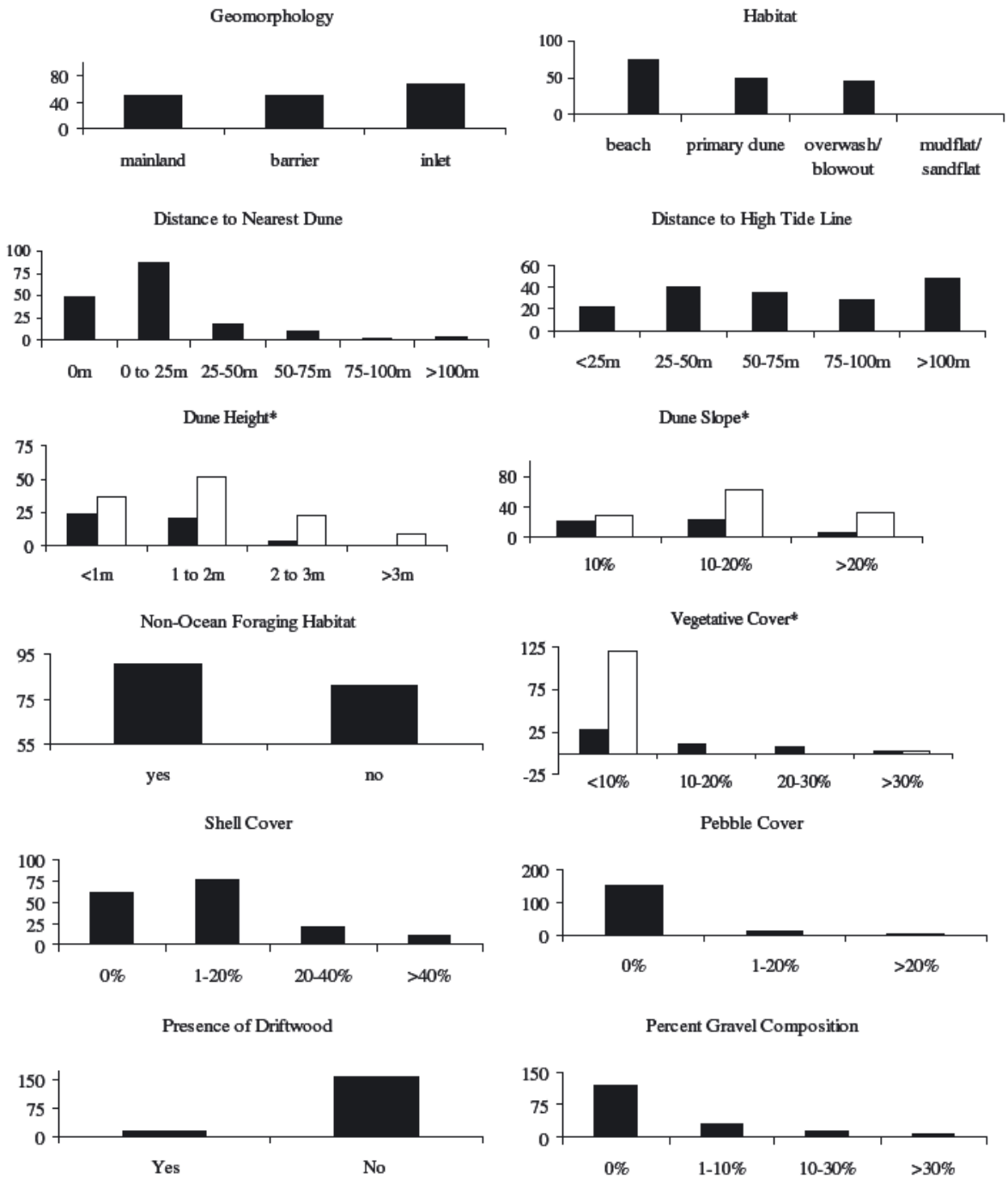


Figure 1. Number of nests found arranged by habitat characteristic. The total number of nests observed is $N = 171$. Open bars indicate nests located near dunes. *Closed bars indicate nests located on primary dunes.

F.3.6 Sidle and Kirsch 1993

Finding:

Table 2. Numbers of adult Least Terns (LT) and Piping Plovers (PP), percentages of all individual birds using sand pits, and the number of sandbar and sand pit nesting sites on the central and lower reaches of the Platte River, 1988-1991. Data were derived from Lingle (1993a), and Nebraska Game and Parks Commission (1988, 1989, and unpublished data), and Sidle *et al.* 1991.

	Central Platte						Lower Platte					
	Number of birds		% birds on pits		No. nesting sites		Number of birds		% birds on pits		No. nesting sites	
Year	LT	PP	LT	PP	Bar	Pit	LT	PP	LT	PP	Bar	Pit
1988	124	96	61	62	6	16	497	161	26	19	24	11
1989	194	94	94	64	6	16	409	188	14	13	38	10
1990	176	74	91	78	3	17	361	149	40	35	23	13
1991	158	58	87	76	3	14	295	73	30	37	35	11
Mean	163.0	80.5	83.2	70.0	4	15.8	390.5	142.8	27.5	26.0	30.0	11.2
SD	29.9	18.0	15.1	8.2	1.7	1.3	85.0	42.3	10.8	11.8	7.6	1.3

F.3.7 Catlin 2009

Finding:

Table 1: Use vs. availability of sandbar habitat for piping plover nesting on the Missouri River during the 2005 – 2007 field seasons with respect to habitat modifications. Use was defined as the number of nests that were initiated on each type of habitat within each year. Subscripts on the proportion of available habitat represent selection (+ selection for, - selection against, and = no selection).

	Natural Sandbars			Modified Natural Sandbars			Engineered Sandbars		
Year[a]	n[b]	Pu_i (95% CI)[c]	Pa_i[d]	n	Pu_i (95% CI)	Pa_i	n	Pu_i (95% CI)	Pa_i
2005	87	0.431 (0.348, 0.514)	$0.510_{=}$	42	0.208 (0.140, 0.276)	$0.193_{=}$	73	0.361 (0.280, 0.442)	$0.297_{=}$
2006*	48	0.232 (0.162, 0.302)	0.500_{-}	31	0.150 (0.091, 0.209)	0.275_{-}	128	0.618 (0.537, 0.699)	0.224_{+}
2007*	54	0.252 (0.181, 0.323)	0.384_{-}	18	0.084 (0.039, 0.129)	0.161_{-}	142	0.664 (0.587, 0.741)	0.455_{+}

[a] Years that showed statistically significant overall selection for or against certain sandbar types from the χ^2 goodness of fit test ($P < 0.05$) are marked with an asterisk (2005: $\chi_2^2 = 2.15$, $P = 0.341$; 2006: $\chi_2^2 = 50.32$, $P < 0.001$; 2007: $\chi_2^2 = 14.28$, $P < 0.001$).

[b] Number of nests initiated on each habitat type within each year.

[c] Proportion of use (Pu) for each year in each of the sandbar classes for each year, as determined from aerial photography and land cover classification.

[d] Proportion of nesting habitat available (Pa) in each of the sandbar classes for each year, as determined from aerial photography and land cover classification.

F.3.8 Cohen et al. 2009

Finding:

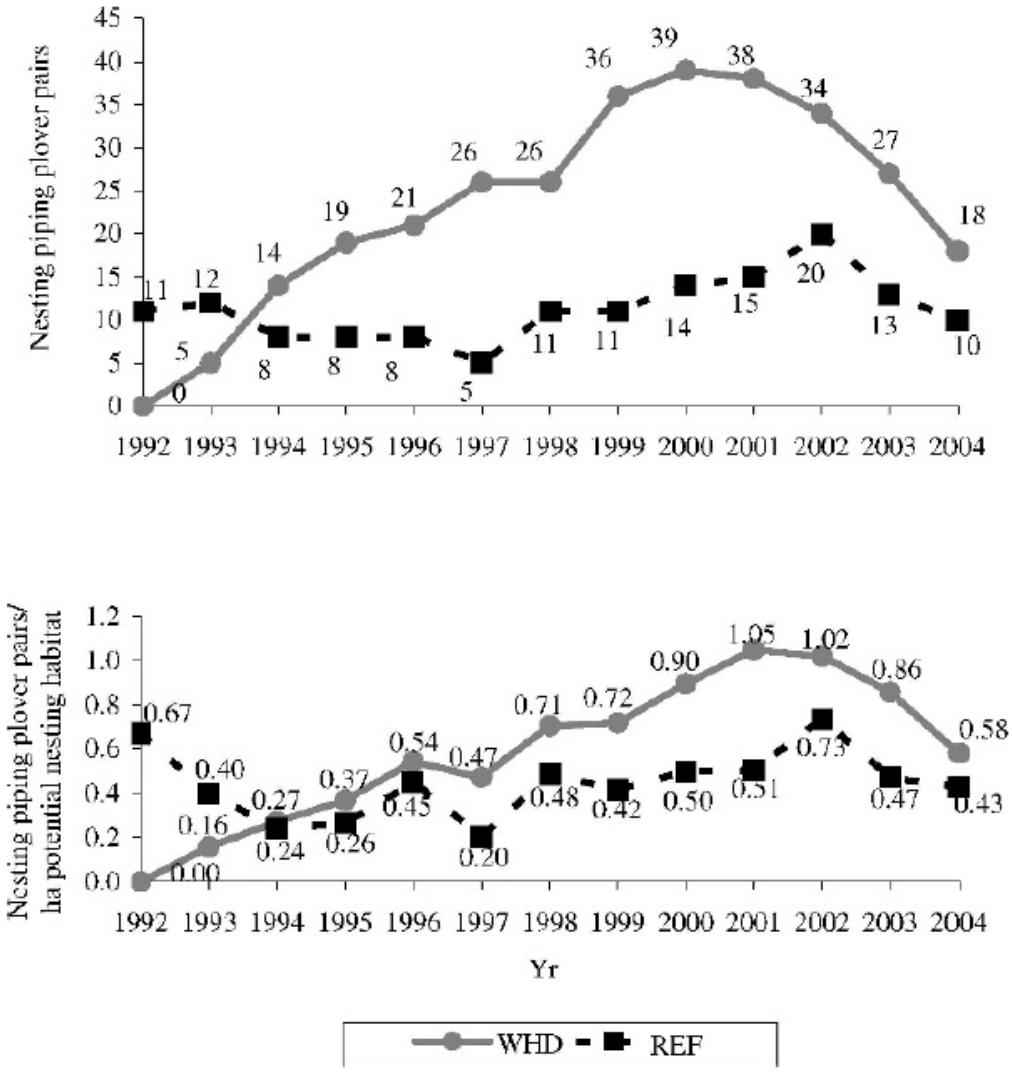


Figure 9. Piping plover nesting-pair numbers and density at West Hampton Dunes (WHD) and the reference area (REF), New York, USA, 1992–2004. Numbers adjacent to the points are the values represented by the points.

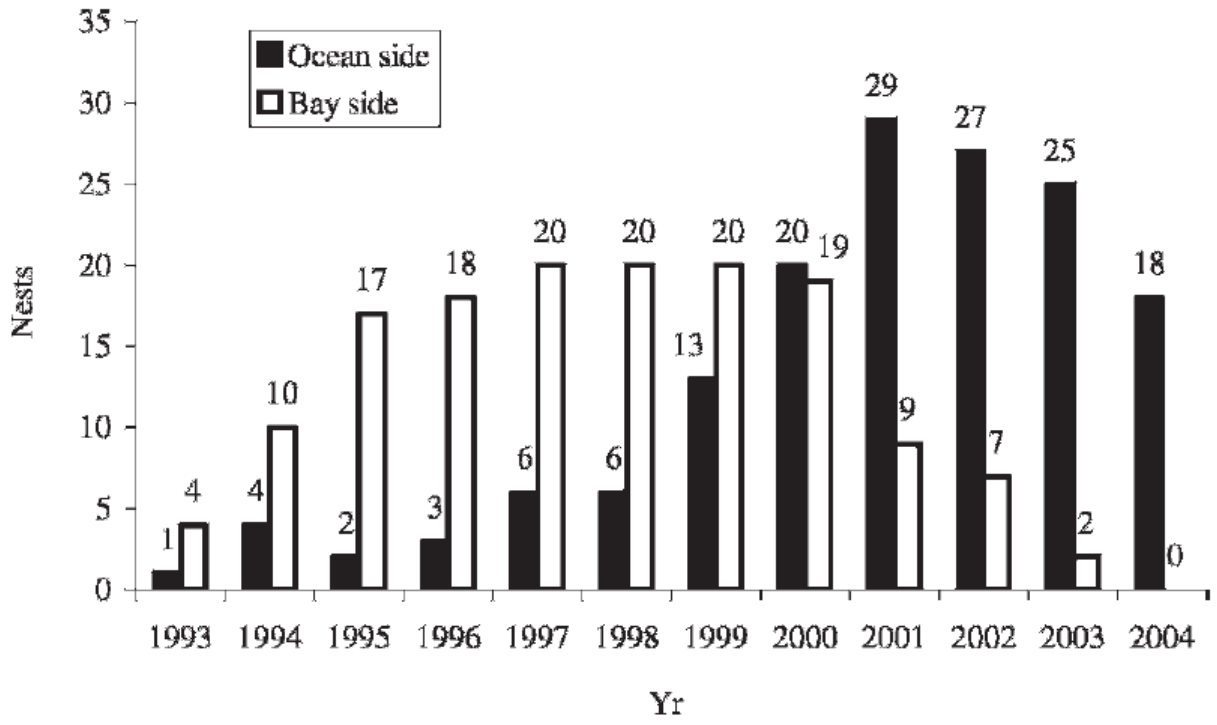
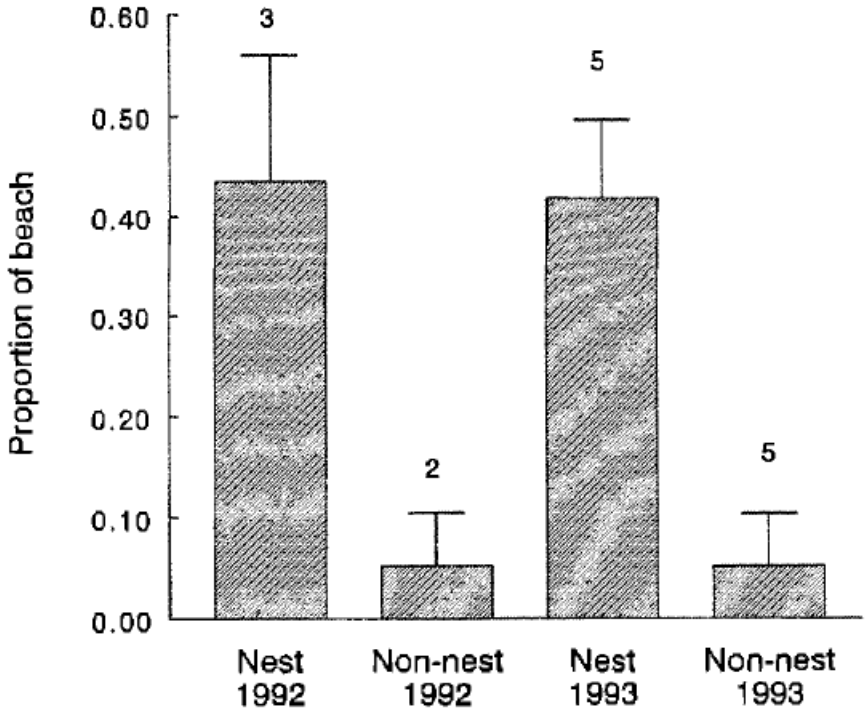


Figure 12. Piping plover first nest attempts at West Hampton Dunes, New York, USA, 1993–2004. Numbers over the bars are the number of nests represented by the bar.

F.3.9 Espie et al. 1996

Finding:

Fig. 1. Proportions of gravel substrate (mean ± SE) found on nest and non-nest beaches in 1992 and 1993. Nest beaches had significantly more gravel substrate than non-nest beaches. There is no significant difference in the amounts of gravel between years and no significant year × beach interaction. Numbers above the bars are sample sizes.



F.3.10 Espie et al. 1998

Elevation:

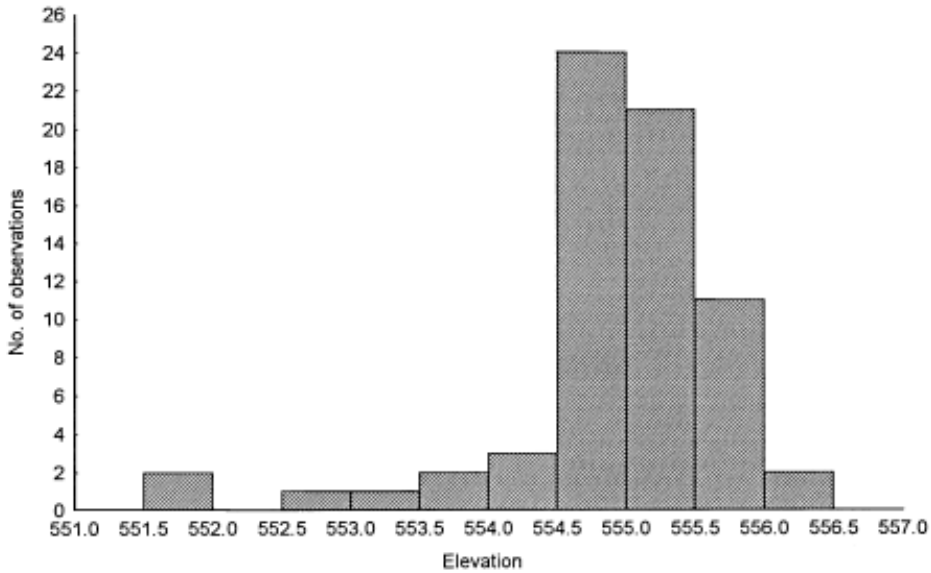


Fig. 1. Frequency histogram of piping plover nest elevations (m above sea level) at Lake Diefenbaker for 1992 and 1993 combined ($n = 67$).

F.3.11 Felio et al. 2010

Finding:

Table 1. Nest distribution across sandbar types.

		2005	2006	2007	2008	2009
Engineered (Year 0)	n	57	na	16	64	89
Engineered (Year 1)	n	16	119	na	43	76
Engineered (Year 2)	n	na	9	115	na	37
Engineered (Year 3)	n	na	na	11	95	na
Engineered (Year 4)	n	na	na	na	10	32
Engineered (Year 5)	n	na	na	na	na	4
Engineered Total	n	73	128	142	212	238
	%	36%	62%	66.5%	72%	79.3%
Natural	n	87	48	54	62	61
	%	43.5%	23%	25%	21%	20.3%
Natural/Modified	n	42	31	18	20	1
	%	20.5%	15%	8.5%	7%	0.3%
Total		202	207	214	294	300

F.3.12 Gaines and Ryan 1988

Finding:

Table 1. Mean beach width and substrate characteristics of piping plover nesting territories and unoccupied areas at Chain-of-Lakes, North Dakota, 1984–85.

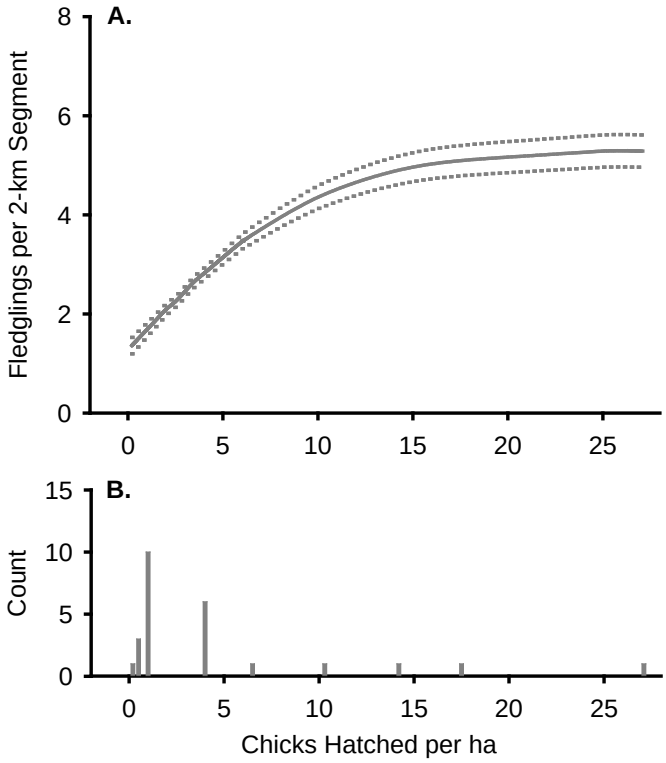
Habitat variables	Year	Territories			Unoccupied areas			
		$\bar{x}$	SE	n[a]	$\bar{x}$	SE	n	P
Beach width (m)	1984	26.8	1.1	218	11.6	1.1	91	<0.0001
	1985	39.1	1.1	390	15.6	1.3	106	<0.0001
Vegetation								
% cover	1984	4.0	0.2	218	4.2	0.4	90	0.8
	1985	3.1	0.2	390	3.5	0.3	106	0.2
Spatial heterogeneity[b]	1984	120.3	3.2	189	99.7	6.8	41	0.007
	1985	135.3	2.7	382	116.4	6.7	93	0.004
Gravel								
Cover (no./0.1 m^2)	1984	48.1	2.0	220	30.1	3.2	92	<0.0001
	1985	53.8	1.2	391	34.4	2.8	105	<0.0001
Spatial heterogeneity	1984	73.4	3.8	204	87.6	8.1	57	0.094
	1985	70.9	2.2	387	86.4	6.5	80	0.007

[a] No. transects.
[b] $\bar{x}$ CV of distance (Roth 1976).

F.4 Nest Density

F.4.1 Anteau et al. 2014b

Findings:



F.4.2 Haffner et al. 2009

Findings:

Table 1. Space use by breeding male and female Great Lakes Piping Plovers expressed as minimum convex polygon (95% MCP) and linear beach distance (95% linear) for the 2003–2004 breeding seasons. Data are shown for the entire breeding season and separately for incubation and chick-rearing periods, and are summarized separately for males and females when analysis suggested differences in range use between the sexes (see Table 2).

	N	Mean	SE	Range
95% MCP (ha)				
Season	35	2.9	0.5	0.4–11.2
Male	17	3.5	0.7	0.6–10.9
Female	18	2.3	0.6	0.4–11.2
Incubation	35	1.8	0.4	0.1–10.7
Chick rearing	35	2.0	0.3	0.1–6.0
Male	17	2.2	0.4	0.2–5.5
Female	18	1.4	0.3	0.1–4.6
95% Linear (m)				
Season	35	475	53	130–1435
Incubation	35	340	40	50–1210
Chick rearing	35	370	40	100–980
Male	17	431	66	130–980
Female	18	317	53	100–930

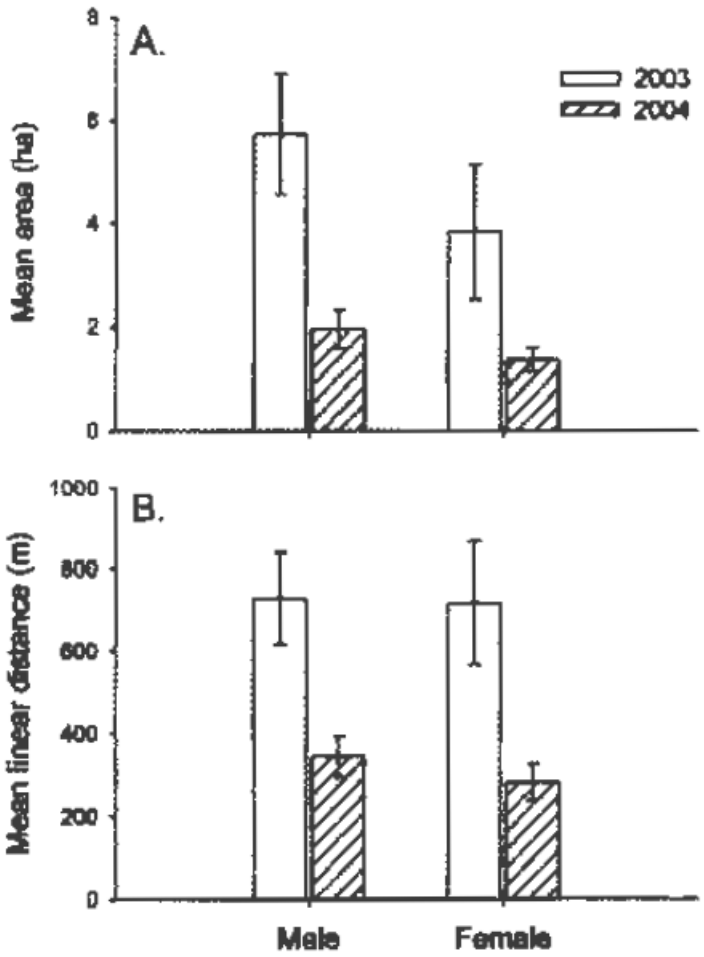
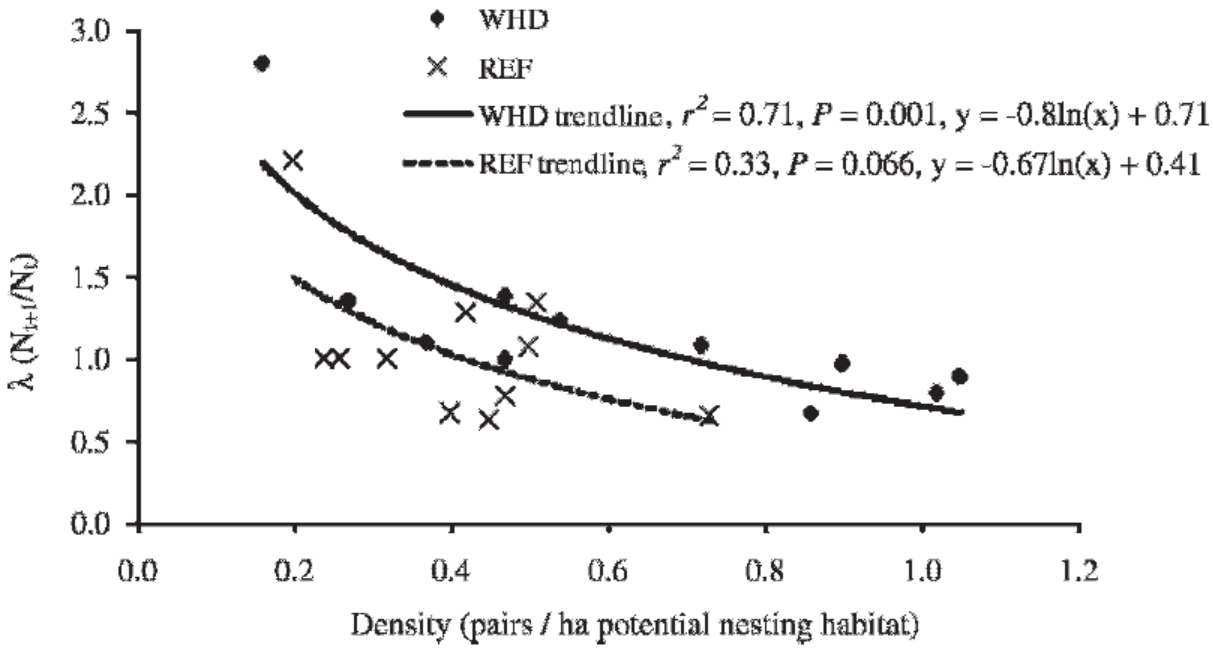


Fig. 2. Mean (± SE) home range sizes (A) and mean linear distances (B) for male and female Great Lakes Piping Plovers in 2003 and 2004.

F.4.3 Cohen et al. 2009

Findings:



F.5 Habitat Quality and Quantity

F.5.1 Anteau et al. 2014a

Findings:

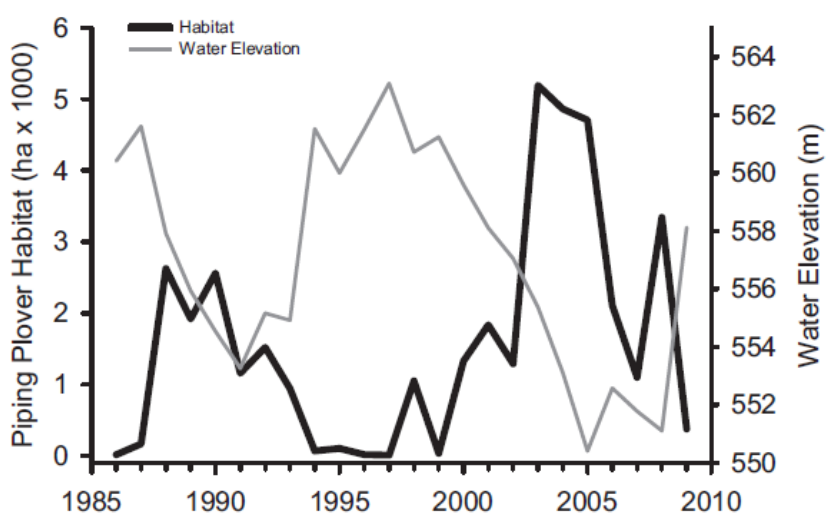


Fig. 8. Annual summaries of piping plover habitat (ha × 1000) for Lake Sakakawea 1986–2009, including water-surface elevations (m MSL).
Adapted from U.S. Army Corps of Engineers (2010).

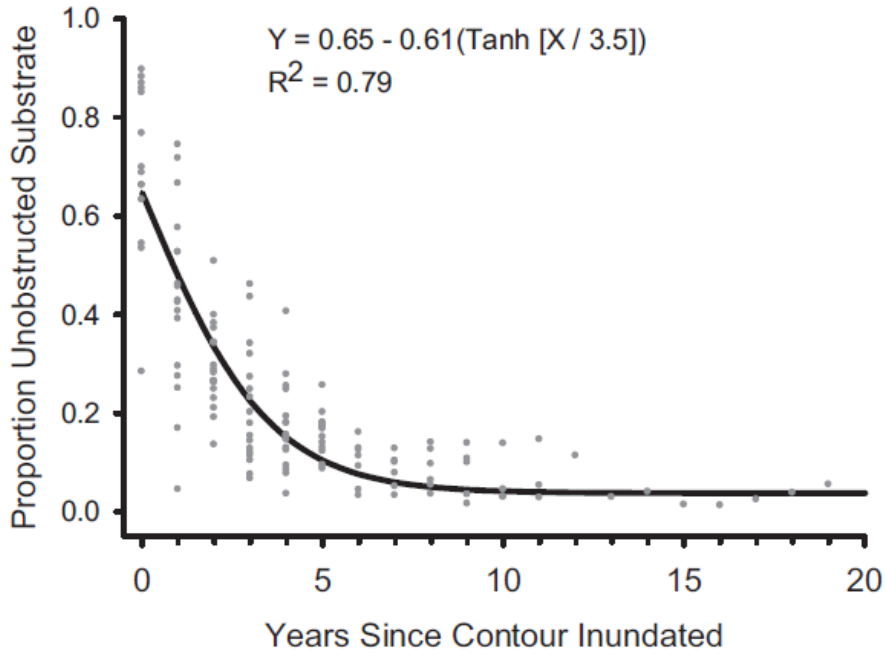


Fig. 9. Relationship of the years since a 1-m-elevation contour was last inundated and the proportion of unobstructed substrate (≤30% cover of vegetation or debris) for Lake Sakakawea.

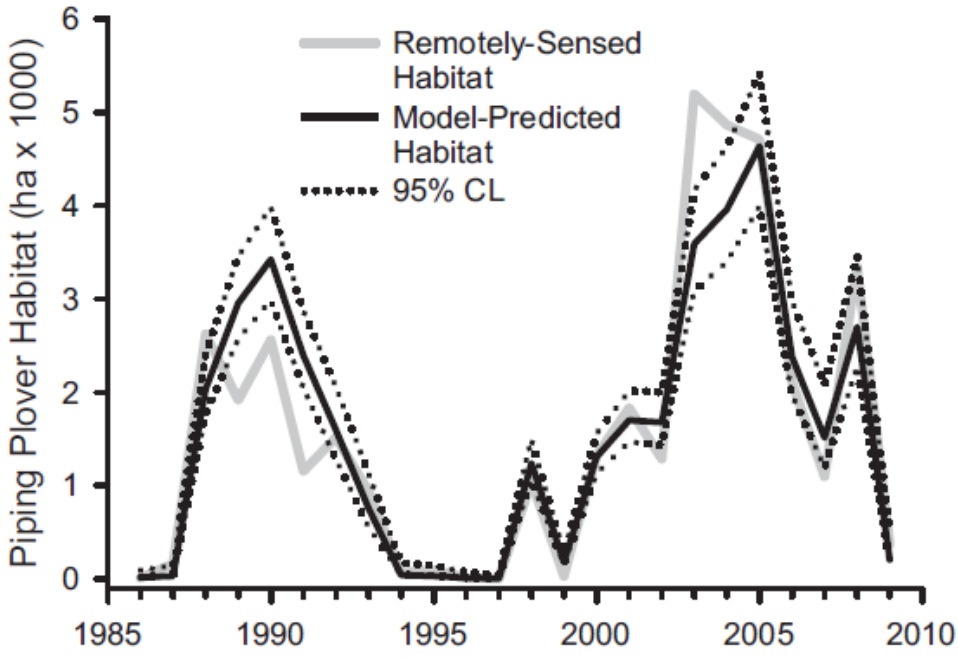
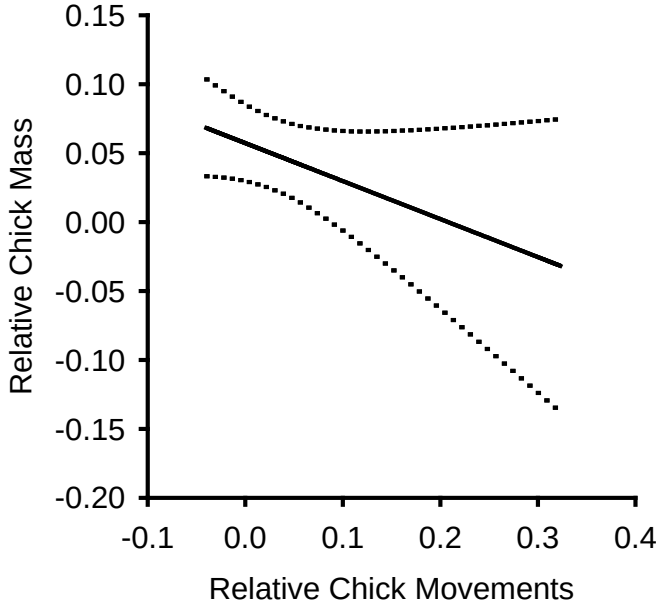


Fig. 11. Remotely-sensed and model-predicted (±95% CL) habitat for piping plovers at Lake Sakakawea, 1986–2009.

F.5.2 Anteau et al. 2014b

Findings:



F.5.3 Goodale et al. 2007

Findings:

Table 1. Logical and hierarchical statements used in the logical filter for the intermediate and final classifications of Johnston's Pond.

Classification	Logical statement
Beach substrate	If mixed substrate overlaps sand and cobble, then classify as mixed
	If sand overlaps cobble, then classify as sand
Intertidal flats	If beach substrate overlaps mudflat, vegetated mudflat, or sandflat, then keep beach substrate classification
	If mudflat overlaps vegetated mudflat and sandflat, then classify as mudflat
	If vegetated mudflat overlaps sandflat, then classify as mudflat
	Merge mudflat and sandflat into one class (mudflat–sandflat)
Thirty metre (30 m) coastline buffer	If vegetated mudflat (from intertidal flats output) is within 30 m of the coastline, then classify as mudflat–sandflat
Intertidal–beach classification	Merge 30 m coastline buffer output with intertidal flats output
Patchy vegetation	If thick vegetation overlaps sand, mixed, or cobble, then classify as patchy vegetation
Final classification	Merge trees–shrubs over patchy vegetation over thick vegetation over water over intertidal–beach classification

Table 2. Logical statements used to classify critical piping plover nesting and feeding habitats.

Piping plover habitat classification	Logical statement
Critical feeding	If slope is less than or equal to 3.0° and beach habitat is classified as mudflat–sandflat or sand, then classify as feeding habitat
Critical nesting	If slope is less than or equal to 3.0° and beach habitat is classified as sand, mixed, or patchy vegetation, then classify as nesting habitat

F.5.4 Haig et al. 2011

Findings:

TABLE 1. Issues addressed by avian conservation genetics.

Taxonomy	(Meta) populations	Landscapes
• What are the evolutionary and geographic boundaries of species, subspecies, and management units? • What is the extent of introgression or hybridization? • Which species are represented in illegal wildlife trade? • What cryptic species have been misclassified? • How is biodiversity changing in response to climate change? • What is the identity of a hybrid, cryptic specimen, or the remains of an ancient or recent specimen? • What is the best avian tree of life for prioritizing biodiversity conservation by phylogenetic diversity?	• What are levels of genetic diversity, population structure, effective population size, or gene flow and how have they changed over time? • What is the best strategy for managing small populations to maximize conservation of genetic diversity (pedigree analyses)? • What is the extent of population connectivity or isolation? • Which populations would be the best source for a translocation or reintroduction? • How do individuals and populations move throughout the annual cycle? • How are disease transmission pathways linked to bird movements?	• What are the effects of landscape features and landscape heterogeneity on genetic diversity and population structure? • How are phylogeographic patterns of birds changing in response to climate change? • Which populations or life stages are most affected by contaminant exposure; are certain genotypes more vulnerable? • In mixed-stock populations, what proportion of each stock is being harvested or affected by environmental perturbations?

F.5.5 Le Fer et al. 2008a

Findings:

Table 1. Mean invertebrate biomass (mg dry wt), invertebrate numbers, Coleoptera biomass (mg), Coleoptera numbers ± standard error per set of sticky traps, and individual invertebrate biomass (mg) in saturated and moist habitats during the chick-rearing period. We collected invertebrates in North and South Dakota, USA, 2001–2003, on paint stirrers covered with Tanglefoot insect coating and placed for 30 minutes. If global model was significant, we show main effects.

	2001			2002			2003			2001–2003		
Site	*n*	$\bar{x}$	SE	*n*	$\bar{x}$	SE	*n*	$\bar{x}$	SE	*n*	$\bar{x}$	SE
Invertebrate biomass[a]												
Alkali	7	35.9 A[b]	17.4	8	40 A	12.9	8	9.5 A	2.4	23	22.2 A	6.1
Hypolimnetic	6	5.3 B	1.4	10	4.5 B	1.4	8	2.9 B	0.3	24	4.1 C	0.7
Epilimnetic	8	24.9 Aa[c]	4.9	11	17.2 Aa	3.7	9	6 Ab	1.2	28	15.4 A	2.3
Reservoir	7	15.7 Ba	5.4	10	4.8 Bb	1.3	4	4 ABb	1.3	21	8.3 B	2.2
Invertebrate no.[d]												
Alkali	7	29.3 Aab	7.3	8	49.8 Aa	15.5	8	12.9 b	3.7	23	30.7 A	6.6
Hypolimnetic	6	4.5 C	0.8	10	5.9 C	1	8	7	1.2	24	5.9 C	0.6
Epilimnetic	8	11.9 ABa	1.4	11	13.8 Ba	3.6	9	5.1 b	0.7	28	10.5 B	1.6
Reservoir	7	10.7 Ba	2.7	10	8.2 BCb	1.4	4	7.6 b	3	21	8.9 BC	1.2
Coleoptera biomass[e]												
Alkali	7	0.4 B	0.2	8	0.1 B	0	8	0	0	23	0.2 B	0.1
Hypolimnetic	6	0.7 B	0.6	10	1.2 B	1.1	8	0.2	0.1	24	0.8 B	0.5
Epilimnetic	8	12.8 Aa	4.4	11	10.5 Aa	3.5	9	1.5 b	0.5	28	8.3 A	2.1
Reservoir	7	2 B	1.6	10	0.5 B	0.5	4	0.1	0	21	0.9 B	0.6
Coleoptera no.[f]												
Alkali	7	0.2 BC	0.1	8	0.1 B	0.1	8	0.1	0.1	23	0.2 B	0
Hypolimnetic	6	0.1 C	0	10	0.1 B	0.1	8	0.1	0	24	0.1 B	0
Epilimnetic	8	2.3 Aa	0.6	11	1.1 Aab	0.3	9	0.7 b	0.3	28	1.3 A	0.3
Reservoir	7	1.6 B	1.1	10	0.2 B	0.1	4	0.3	0.2	21	0.7 B	0.4
Individual invertebrate biomass[g]												
Alkali	7	1.1 a	0.3	8	0.5 Bb	0.1	8	1 ABab	0.3	23	0.9 B	0.2
Hypolimnetic	6	1.3	0.3	10	1 B	0.4	8	0.5 C	0.1	24	0.9 B	0.2
Epilimnetic	8	2 a	0.2	11	1.6 Aab	0.4	9	1.2 Ab	0.2	28	1.6 A	0.2
Reservoir	7	1.5 a	0.2	10	0.6 Bb	0.1	4	0.7 BCb	0.3	21	0.9 B	0.1

[a] Invertebrate biomass = sum of individual invertebrate biomass trapped/30 min. Analysis of variance (ANOVA) on ranks: site: $F_{3,95} = 19.9$, $P < 0.001$; yr: $F_{2,95} = 15.2$, $P < 0.001$; site × yr: $F_{6,95} = 1.4$, $P = 0.21$.

[b] Site means with the same letter within columns are not significantly different ($\alpha = 0.05$).

[c] Site means with the same letter within rows are not significantly different ($\alpha = 0.05$).

[d] ANOVA on ranks: site: $F_{3,95} = 13.1$, $P < 0.001$; yr: $F_{2,95} = 2.9$, $P = 0.06$; site × yr: $F_{6,95} = 2.7$, $P = 0.02$.

[e] ANOVA on ranks: site: $F_{3,95} = 17.5$, $P < 0.001$; yr: $F_{2,95} = 3.3$, $P = 0.04$; site × yr: $F_{6,95} = 0.66$, $P = 0.7$.

[f] ANOVA on ranks: site: $F_{3,95} = 20.3$, $P < 0.001$; yr: $F_{2,95} = 2.1$, $P = 1.3$; site × yr: $F_{6,95} = 1.1$, $P = 0.36$.

[g] Individual invertebrate biomass = sum of individual invertebrate biomass/no. individuals. ANOVA on ranks: site: $F_{3,95} = 13.9$, $P < 0.001$; yr: $F_{2,95} = 14.5$, $P < 0.001$; site × yr: $F_{6,95} = 1.6$, $P = 0.15$.

Table 2. Mean foraging rates (pecks/min), proportion of time foraging, and estimated biomass consumed (mg/min) ± standard error for piping plover chicks aged day 3–10 and day 11 to fledging, in North and South Dakota, USA, 2001–2003. Results of analysis of variance (ANOVA) on ranks comparing sites.

	Day 3–10			Day 11–fledge		
Site	n	$\bar{x}$	SE	n	$\bar{x}$	SE
Foraging rate[a]						
Alkali	21	5.8 B[b]	0.9	15	6.5 B	1.5
Hypolimnetic	16	10.8 A	1.9	11	14.2 A	1.3
Epilimnetic	21	6.7 B	0.7	14	9.2 B	0.7
Reservoir	13	7.2 B	1.3	4	8.4 B	3.4
Proportion of time foraging[c]						
Alkali	21	0.43 B	0.06	19	0.51 B	0.09
Hypolimnetic	16	0.69 A	0.07	12	0.82 A	0.05
Epilimnetic	21	0.78 A	0.06	14	0.83 A	0.04
Reservoir	13	0.64 A	0.08	5	0.53 B	0.18
Biomass consumption index[d]						
Alkali	21	5 C	1	15	5.6 B	1.2
Hypolimnetic	16	9.7 AB	2.1	11	12.7 A	2.6
Epilimnetic	21	10.6 A	1.2	14	15.1 A	1.7
Reservoir	13	6.2 BC	0.9	4	9.6 B	1.8

[a] ANOVA on ranks: day 3–10: $F_{3,70} = 3.3$, $P = 0.02$; day 11 to fledge: $F_{3,43} = 5.1$, $P = 0.004$.
[b] Site means with the same letter within columns are not significantly different ($\alpha = 0.05$).
[c] ANOVA on ranks: day 3–10: $F_{3,70} = 5.4$, $P = 0.002$; day 11 to fledge: $F_{3,49} = 4.7$, $P = 0.007$.
[d] Biomass consumption index = average invertebrate individual biomass × foraging rate; ANOVA on ranks: day 3–10: $F_{3,70} = 3.9$, $P = 0.001$; day 11 to fledge: $F_{3,43} = 5.9$, $P = 0.001$.

Table 3. Slopes of regressions of natural log of mass of piping plover chicks on age in days at 4 sites in North and South Dakota, USA, in 2001–2003. Mass was the mean mass of surviving chicks in each brood for each chick age.

	2001–2003[a]		2001[b]		2002[c]		2003[d]	
Site	Slope	SE	Slope	SE	Slope	SE	Slope	SE
Alkali	0.081 A[e]	0.002	0.086 AB	0.004	0.079 AB	0.002	0.083 A	0.002
Hypolimnetic	0.076 B	0.001	0.078 B	0.003	0.076 B	0.002	0.077 B	0.002
Epilimnetic	0.077 B	0.001	0.092 A	0.003	0.081 A	0.002	0.067 C	0.002
Reservoir	0.078 AB	0.002	0.08 B	0.005	0.078 AB	0.002	0.075 B	0.003

[a] $F_{3,581} = 1641$, $P < 0.001$.
[b] $F_{3,148} = 481$, $P < 0.001$.
[c] $F_{3,237} = 738$, $P < 0.001$.
[d] $F_{3,196} = 747$, $P < 0.001$.
[e]Sites with the same letter within yr (columns) are not significantly different ($\alpha = 0.05$).

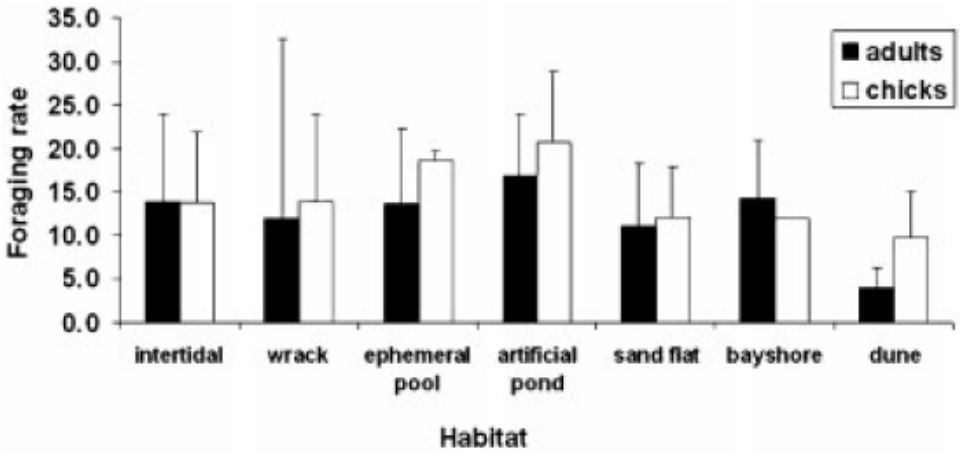
F.5.6 Maslo et al. 2012

Findings:

Table 1. Piping plover productivity for the Lower Cape May Meadows (restoration site), Barnegat Light, North Brigantine Natural Area, and Avalon (reference sites), New Jersey, 2007–2009.

Site	Chicks fledged per nesting pair
Lower Cape May Meadows (restored site)	
2007	2.00
2008	1.75
2009	1.00
Average	1.58
Barnegat Light	
2007	1.75
2008	0.33
2009	1.00
Average	1.03
North Brigantine Natural Area[a]	
2007	0.50
2008	0.50
Average	0.33
Avalon	
2007	0.60
2008	0.25
2009	0.50
Average	0.45

[a] All nests at North Brigantine Natural Area failed due to flooding in 2009; therefore, productivity could not be calculated. In this case, we provided average reproductive success for 2007–2008 (data provided by the New Jersey Division of Fish and Wildlife—Endangered and Nongame Species Program).



F.5.7 Melvin et al. 1991

Findings:

Table 2
Number of Breeding Pairs of Piping Plovers in Atlantic Coast States by Ownership Categories, 1989

Ownership[a]	ME	MA	RI	CT	NY	NJ	DE	MD	VA	NC	Total	(%)
Federal												
Fish and Wildlife Service		6	4		1	25			46	1	83	(11)
National Park Service		15			22	15		20		45	117	(16)
Department of Defense[b]						6			1		7	(1)
NASA									5		5	(1)
State												
Parks or Conservation Department[c]	4	7		5	32	17	3				68	(9)
Other[d]										1	1	(1)
County					26						26	(4)
Municipal		48	5	25	54	37			4		173	(24)
Private												
Conservation Organization		24		3	4	2			60		93	(13)
Individual	12	37	10	1	50	9			5	5	129	(18)
Unknown					2	17					19	(3)
Total	16	137	19	34	191	128	3	20	121	52	721	(100)

[a]Data not available from South Carolina.
[b]Includes U.S. Coast Guard and Army Corps of Engineers.
[c]Includes departments of parks, conservation, and environmental management.
[d]Coastal lands claimed by state.

F.5.8 Sidle et al. 1992

Findings:

Table 1. The number of monitored Least Tern (LT) and Piping Plover (PP) nests inundated along the Platte River between the Loup River confluence and the Missouri River, 1986-90.

Yr	No. Nests Before First Flood	No. Flooded	% Flooded	No. Renests[1]	No. Renests Flooded	Total % Flooded	Maximum Daily Flow (cms) NB/LV[2]
	LT/PP	LT/PP	LT/PP	LT/PP	LT/PP	LT/PP	
1986[3]	136/28	18/6	13/21	0/0	0/0	13/21	421.7/793.0
1987[4]	69/17	5/0	7/0	0/0	0/0	7/0	302.8/484.3
1988[5]	77/32	3/1	4/3	0/0	0/0	4/3	143.2/162.6
1989[5]	61/13	21/7	34/54	24/5	0/0	28/39	331.1/320.0
1990[5]	94/28	94/28	100/100	93/13	4/6	52/85	911.3/1713.3

[1]Least terns and piping plovers renested after losing eggs or chicks from other causes in 1986 and 1988 but initiated no new nests on the river after flooding.
[2]Flows measured at U.S. Geological Survey gauging stations at North Bend and Louisville, Nebraska.
[3]Flooding occurred in late June after most nests had hatched.
[4]Flooding occurred in late May-early June before birds had initiated many nests.
[5]The first flood inundated sandbars along the entire river and occurred during peak nesting. The second flood inundated sandbars below Salt Creek (U.S. Army Corps of Engineers river mile 26) and occurred in late July after most tern but no plover renests had hatched. The second flood killed all chicks on sandbars below river mile 26.

F.5.9 Burger 1994

Findings:

TABLE 3. Habitat comparisons (ocean, dunes, bay) of feeding piping plover and presence of people at three study sites in New Jersey.

	Brigantine	Holgate	Corson's Inlet
Seconds (±SD) devoted to feeding			
Ocean	76 ± 28	106 ± 22	87 ± 36
Dune	50 ± 20	107 ± 22	87 ± 41
Bay	70 ± 24	109 ± 16	75 ± 40
Wilcoxon χ^2 (p)	48.3 (0.0001)	NS	6.23 (0.01)
People within 100 m (mean ± SD)			
Ocean	2.03 ± 0.3	0.09 ± 0.6	2.7 ± 2.1
Dunes	3.3 ± 3.3	0.02 ± 0.3	0.08 ± 0.4
Bay	1.2 ± 1.2	0.10 ± 0.5	3.03 ± 2.0
Wilcoxon χ^2 (p)	19.9 (0.0001)	30.5 (0.0001)	11.8 (0.0006)

F.5.10 Goldin and Regosin 1998

Findings:

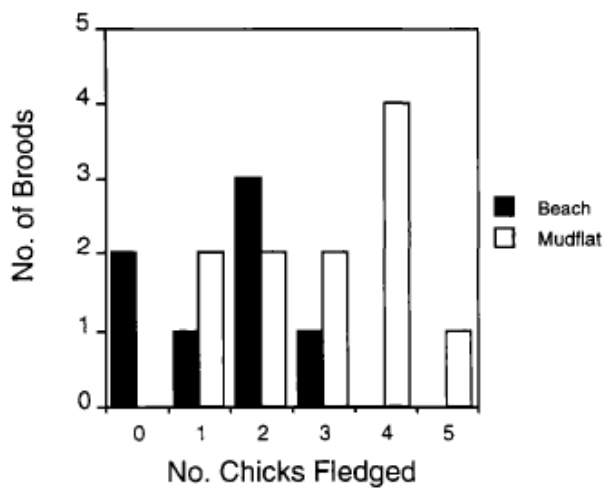


FIGURE 1. Fledging success for Piping Plover broods with and without access to mudflat habitat at Goosewing Beach, Rhode Island. Brood of five chicks resulted from an inter-pair adoption (see Methods).

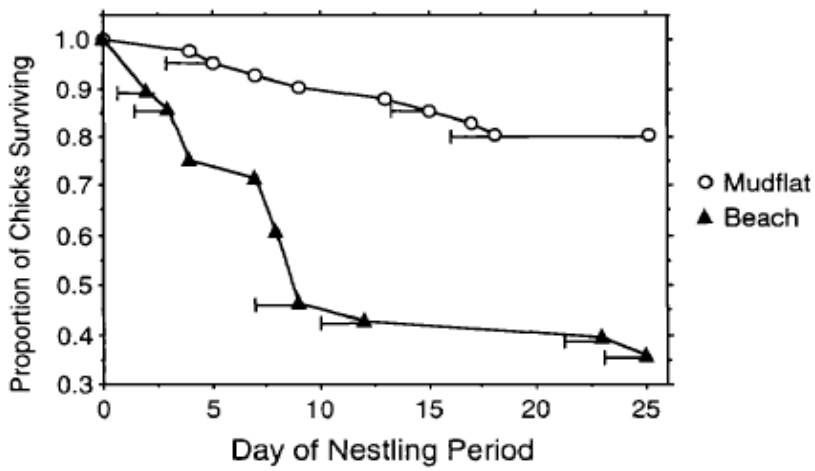


FIGURE 2. Survivorship curves of prefledging Piping Plover chicks with access to mudflat and ocean beach habitat at Goosewing Beach, Rhode Island. Error bars (d) are shown only for mortality events that could not be dated to within two consecutive calendar days.

F.5.11 Knetter et al. 2002

Findings:

Table 2. Mean beach characteristics of piping plover nesting territories for the Lostwood and Stateline regions in northwest North Dakota and northeast Montana during 1999.

	Lostwood (n^a = 45)		Stateline (n^a = 38)			
Habitat variables	Mean	SE	Mean	SE	t	P
% cover vegetation	26.6	2.3	19.5	2.0	–2.22	0.03
Spatial heterogeneity vegetation[b]	108.6	11.0	121.4	14.4	0.72	0.47
Gravel cover (no./0.1 m^2)	13.8	1.1	8.6	0.8	–3.51	0.001
Spatial heterogeneity gravel[b]	99.7	7.0	78.8	6.5	–2.15	0.03
Spatial heterogeneity cobble[b]	97.7	6.7	59.0	8.6	–3.57	<0.001

[a] No. nests.

[b] CV of distance (Roth 1976).

Table 4. Mean beach characteristics of successful and unsuccessful piping plover pairs for both regions in northwest North Dakota and northeast Montana during 1999.

	Successful (n^a=46)		Unsuccessful (n^a=37)			
Habitat variables	Mean	SE	Mean	SE	t	P
% cover vegetation	23.98	2.33	22.67	2.30	–0.40	0.69
Spatial heterogeneity vegetation[b]	116.49	11.61	111.62	13.72	–0.27	0.79
Gravel cover (no./0.1 m^2)	10.65	0.94	12.55	1.34	1.20	0.23
Spatial heterogeneity gravel[b]	89.22	6.14	91.38	8.17	0.22	0.83
Spatial heterogeneity cobble	84.14	7.66	76.28	8.85	–0.67	0.50

[a] No. nests.

[b] CV of distance (Roth 1976).

F.5.12 Roche et al. 2012

Findings:

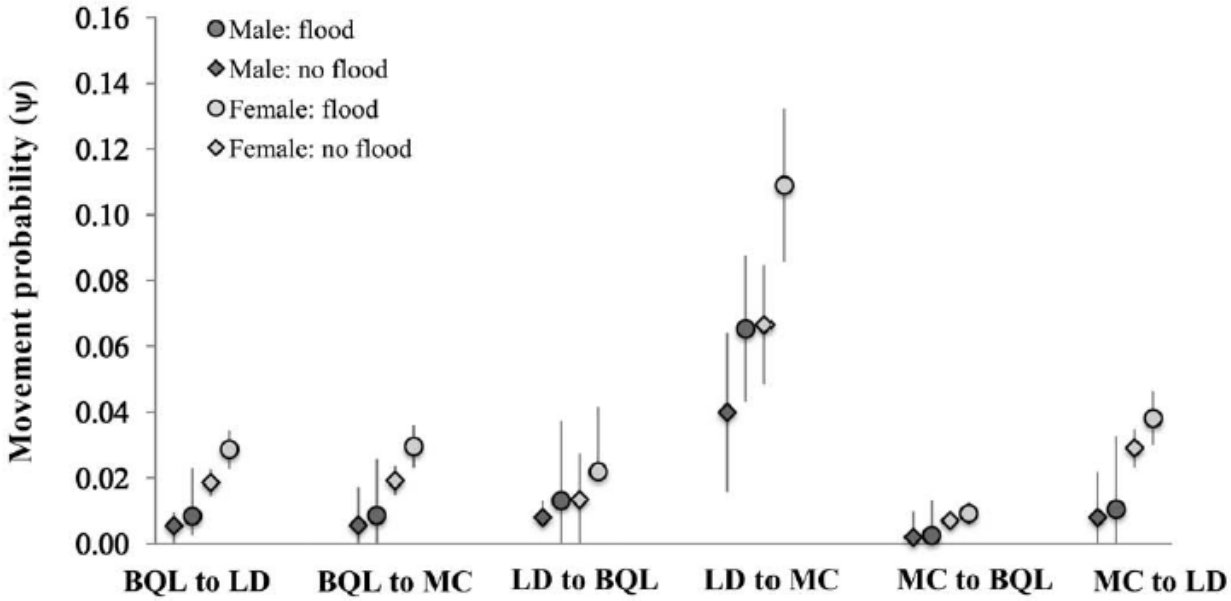


FIG. 3. Probability of interannual breeding-season movement (ψ) following flood and nonflood years among three Piping Plover breeding areas in Saskatchewan (BQL = Big Quill Lake, LD = Lake Diefenbaker, and MC = Missouri Coteau) (2002–2009). See Table 1 for movement-specific sample sizes. Diamonds represent movement following nonflood years; circles represent movement following flood years. Movement probabilities for male Piping Plovers are represented by dark gray markers and those of females by light gray. Vertical lines represent 85% confidence intervals. Estimates were generated from the top-supported model solved at the annual mean covariate value for "fledge," using a sample of 782 marked after-hatch-year Piping Plovers [S(g*t),p(g*t + sex),ψ(g. + sex + flood + fledge)].

F.5.13 Sherfy et al. 2009a

Findings:

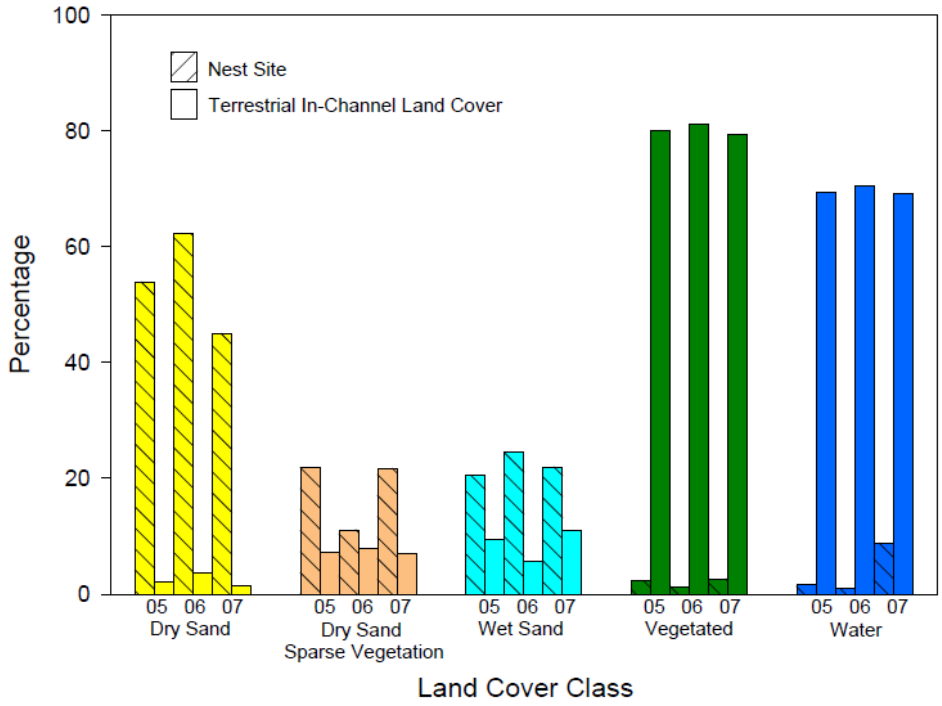


Figure 2. Distribution of least tern and piping plover nests among land cover classes and the percentage of the land cover class for the Gavins Point Reach in 2005, 2006 and 2007.

F.5.14 Espie et al. 1998

Findings:

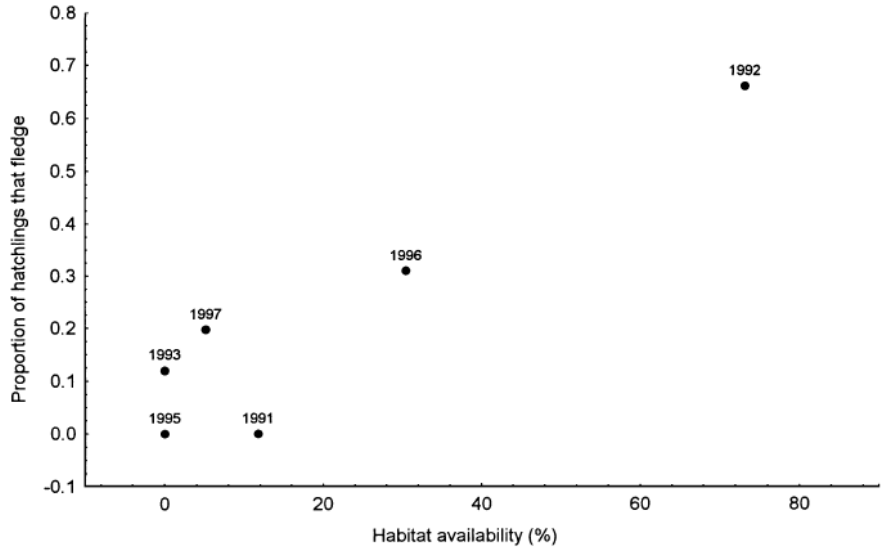


Fig. 2. Scatter diagram of proportion of piping plover hatchlings that fledged and availability of chick-rearing habitat at Lake Diefenbaker on the mean fledge date (7 July). $\tau = 0.643$, $p = 0.07$, $n = 6$.

Table 2
Observed and predicted fledging success at Lake Diefenbaker from 1988–1997

Year	Observed reproduction	Model I predicted	Model II predicted	Model III predicted
1988[a]	0.7	1.7	1.4	1.7
1989[b]	0.6	1.7	0	0.4
1990[c]	0.0	0.2	0	0.1
1991[d]	0.0	1.6	0.2	0.4
1992	2.0	1.7	1.3	1.7
1993	0.1	0.6	0	0.1
1994[e]	0.7	1.5	0.5	0.8
1995[e]	0.0	1.3	0	0.3
1996[e]	0.6	1.1	0.3	0.6
1997[f]	0.3	0.9	0.1	0.2
Mean	0.49	1.23	0.38	0.63

In 1988–1991 and 1994 observed fledging rates are based on brood surveys (chicks of all ages included) and thus likely represent a maximum fledging rate for these years.

[a] Harris, 1988.

[b] Harris and Lamont, 1989.

[c] Harris and Lamont, 1990.

[d] Harris and Lamont, 1991.

[e] D. Duncan, Saskatchewan Wetland Conservation Corporation, unpublished data.

[f] P. Goossen, Canada Wildlife Service, unpublished data.

F.6 Predation

F.6.1 Catlin et al. 2011a

Findings:

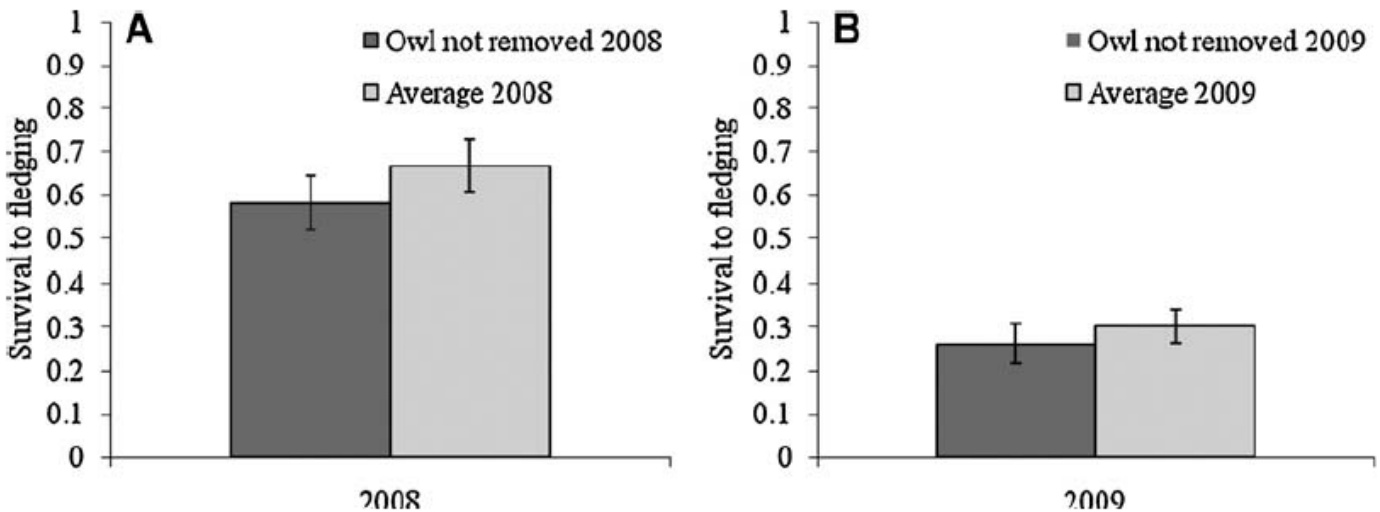


Figure 2. Probability of survival to fledging (approx. 25 days) of piping plover chicks on the Missouri River with average owl removal and without owl removal in (A) 2008 and (B) 2009. We calculated these values for the mean hatch date for a given year. Error bars represent 1 standard error.

F.6.2 Larson et al. 2003

Findings:

Table 1. Financial costs (US$) and reproductive benefits of applying predator-exclusion management on piping plover nesting beaches in the northern Great Plains of North America.

Management technique	Initial cost to protect 4 breeding pairs[a]				AAEV per pair[d]	Fledglings per pair[b]	
	Materials	Transportation	Labor[c]	Sum		Mean	SE
No protection				0	0.00	0.89	0.10
Nest cages	80	133	540	753	180.05	1.28	0.07
Temporary fencing	885	106	592	1,583	224.13	1.02[e]	
Permanent fencing	1,060	106	1,004[f]	2,170	221.43	1.15	0.32
Nest cages and temporary fencing	965	106	608	1,679	230.94	1.78	0.21
Nest cages and permanent fencing	1,140	106	1,020[f]	2,266	228.24	2.25	0.37

[a] The average fence protected approximately 4 breeding pairs. Financial cost data were collected in North Dakota and Montana during the 1996–1998 breeding seasons.

[b] Fledging rates were published previously by Larson et al. (2002) and represent data from throughout the northern Great Plains during various intervals from 1986 to 1999.

[c] Included 13.1 hr/pair for transportation and monitoring. We assumed that removal of cages and temporary fencing required half as long as deployment.

[d] AAEV = average annual equivalent value (Farmer et al. 1988:15) using a 5% annual discount rate. It included the costs of application, maintenance, and replacement of materials over 50 years.

[e] This fledging rate does not represent field data; it was calculated using equation 1.

[f] Time required to erect a permanent fence was estimated by Mayer and Ryan (1991).

F.6.3 Murphy et al. 2003b

Findings:

Table 2. Number of applications of predator exclosure cages at which adult Piping Plovers were killed (N_d) and total number of applications (N_{tot}) for two sizes of cages used to protect plover nests at alkali lakes in the northern Great Plains during 1993-2002.

	Small cages[a]		Large cages[b,c]	
Year	N_d	N_{to}	N_d	N_{tot}
1993	0	8		
1994	0	9		
1995	0	41	1	20
1996	10	89		
1997	4	111		
1998	9	122	0	110
1999	27	198	11	121
2000	0	44	4	116
2001	0	75	1	122
2002	0	73	1	96
Total	50	770	18	585
%	6.5		3.1	

[a]1-1.7m diameter.
[b]3-4m diameter.
[c]Changed from wire mesh top to netting top in 2000.

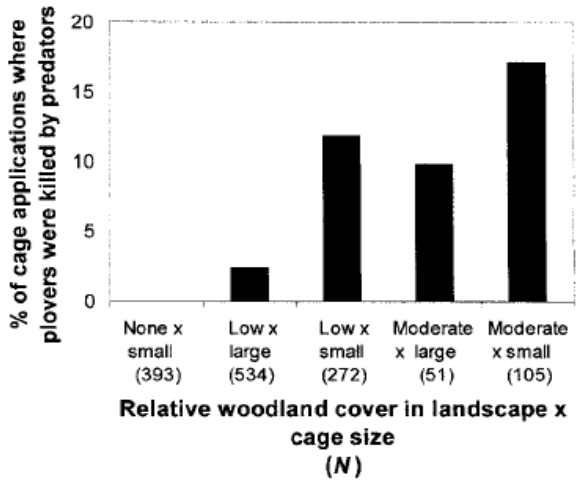


Figure 2. Frequency of predation on adult Piping Plovers at predator exclosure cages as a function of surrounding woodland cover (none: <1% of landscape wooded within 2 km; low: 4%; moderate: 15%) and cage size (small: 1- to 1.7-m diameter; large: 3- to 4-m diameter), at five alkali lake areas in the northern Great Plains during 1993-2002. Numbers of applications are indicated parenthetically.

F.6.4 Smith et al. 2011

Findings:

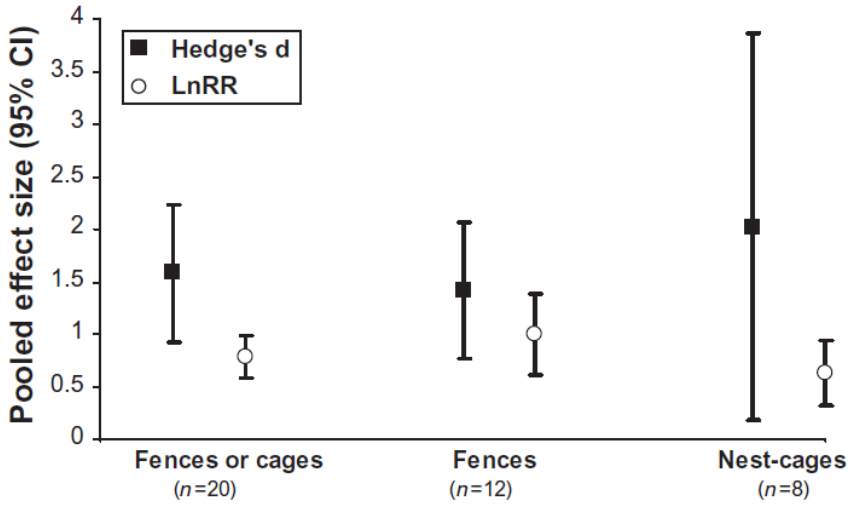


Fig. 1. Pooled effect sizes and 95% confidence intervals for studies of the effectiveness of predator exclosures (fences or nest-cages) for increasing hatching success, where Hedge's d and response ratios (RR) were used as the effect size estimators. Sample sizes (*n*) are also shown.

F.6.5 Doherty and Heath 2011

Findings:

Table 2. Fates of exclosed and unexclosed piping plover nests on Suffolk County Park beaches, New York, in 2006 and 2007. Predator exclosure use increased piping plover hatching success.

		Fate category									
		Successful		Mixed-fate		Abandoned		Depredated		Flooded	
Management	Hatching success	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%
Exclosed	0.68	20	49	11	27	7	17	1	2	2	5
Not exclosed	0.56	26	39	10	15	10	15	14	21	3	5
Total	0.60	46	46	21	17	17	16	15	14	5	5

F.6.6 Larson et al. 2002

Findings:

Table 5. Current and proposed numbers of piping plovers in the Great Plains population receiving predator-exclusion management on alkaline wetland sites and their effect on the mean reproductive success rate for all plovers at alkaline wetland sites.

		Current management			Proposed management		
Management technique[a]	Fledging rate, F[b]	Pairs	Proportion, P_c[c]	FP_c	Pairs[d]	Proportion, P_p[c]	FP_p
None	0.89	902	0.824	0.73	735	0.671	0.60
C	1.28	125[e]	0.114	0.15	234	0.214	0.27
CT	1.78	47[e]	0.043	0.08	105	0.096	0.17
CP	2.25	12[e]	0.011	0.02	21	0.019	0.04
P	1.15	9[e]	0.008	0.01	0	0.000	0.00
Sum				0.99			1.08

[a] None = no protection, C = nest cages, CT = cages and temporary electric fencing, CP = cages and permanent electric fencing, and P = permanent electric fencing.

[b] Fledglings produced per breeding pair per year.

[c] Proportion of all pairs at alkaline wetland sites in the Great Plains (n = 1,095).

[d] Number of pairs based on an effort to protect 360 pairs at alkaline wetlands in North Dakota and Montana assuming no new permanent fences, 35% of pairs would receive fencing, and all protected nests would be caged.

[e] Mean number of protected pairs at the Williams Preserve, State Line, and Mountrail sites in 1986–1999.

F.6.7 Maslo and Lockwood 2009

Findings:

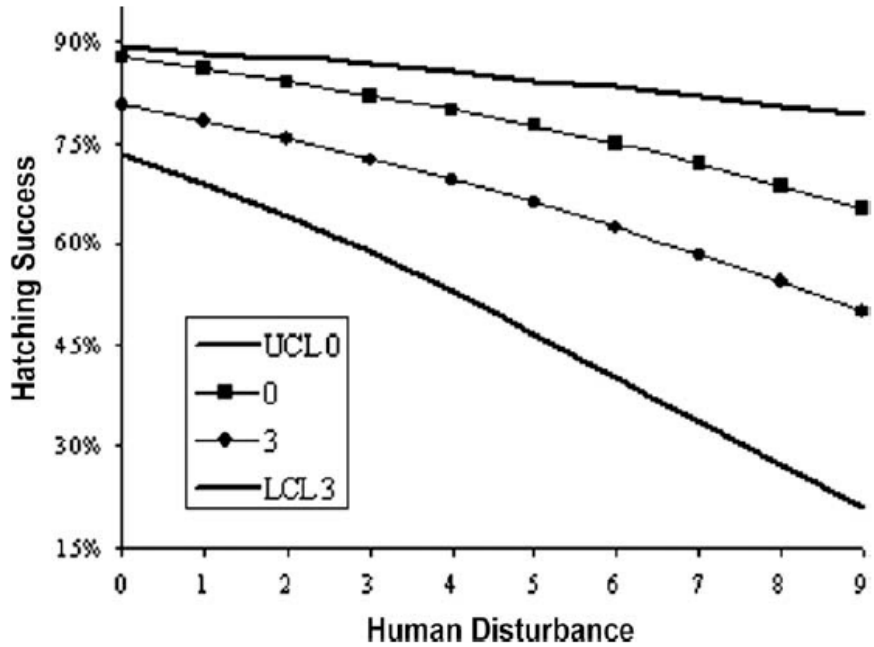


Fig. 3. Effect of human disturbance on hatching success of electrified exclosures at mammalian predator pressure levels of 0 and 3. Predator pressure levels of 1 and 2 show the same trend and have been excluded to increase clarity of the figure. UCL represents the upper 95% confidence level for mammalian predator pressure level of 0. LCL represents the lower 95% confidence level for mammalian predator pressure level of 3.

F.6.8 Maxson and Haws 2000

Findings:

Some management activities at PCI (e.g., predator trapping, predator exclosures) have been effective in increasing hatching success of Piping Plover nests. Use of gull deterrents has caused nesting Ring-billed Gulls to abandon their colony sites making those areas available to Piping Plovers. Nevertheless, low fledging rates continue to be a serious problem for Piping Plovers at LOTW.

F.6.9 Patterson et al. 1991

Findings:

Table 2. Probable causes of piping plover nest loss, Assateague Island, 1986–87.

Area	Nest losses (n)	Cause of loss (n)			
		Predator	ORV[a]	Other[b]	Unknown
Maryland					
Northern 11 km	25	17	0	1	7
ORV section	1	0	0	0	1
Virginia					
Wild Beach	18	14	0	0	4
Wash Flats	2	0	0	1	1
Tom's Cove Hook	21	11	1	1	8
Island					
Total losses	67	42	1	3	21

[a] Off-road vehicle.
[b] Other causes of nest loss were high tides (n = 1), wild pony (*Equus caballus*) (n = 1), and destruction of own eggs (n = 1).

F.6.10 Espie et al. 1996

Findings:

Table 1. Description of habitat variables used in analysis of Piping Plover nest-site selection.

Variable	Description
Near water	Distance in metres from the nest or random site to the nearest water source
Main water	Distance in metres from the nest or random site to Lake Diefenbaker
Elevation	Elevation of the nest or random site
Vegetation	Distance in metres from the nest or random site to the vegetation line
Pebbles	Number of pebbles per square metre at the nest or random site
Stones	Number of stones per square metre at the nest or random site
Rocks	Number of rocks per square metre at the nest or random site
% veg	Percent vegetative cover at the nest or random site
Sticks	Number of sticks >10 cm long within 1 m of the nest or random site

Table 5. Sample means and standard deviations of habitat variables for successful, flooded, and depredated nests.

				F (ANOVA)[a]	
Variable	Successful nests	Flooded nests	Depredated nests	1	2
Near water	68.23±40.10 (30)	43.92±26.45 (24)	53.00±20.53 (13)	6.06*	0.53
Main water	131.70±93.64 (30)	69.29±51.44 (24)	81.62±69.13 (13)	8.66**	3.30
Vegetation	120.83±142.12 (30)	92.87±82.32 (24)	23.83±15.28 (13)	0.06	5.05*
Pebbles	1025.67±354.38 (21)	1051.92±390.11 (20)	946.89±21.74 (11)	2.53*	0.25
Stones	37.05±26.27 (21)	47.03±18.89 (20)	31.83±21.74 (11)	1.94	0.32

Note: Numbers in parentheses are sample sizes.

[a]1, successful vs. flooded nests; 2, successful vs. depredated nests; *, $p < 0.05$; **, $p < 0.01$; others $p > 0.05$.

F.6.11 Catlin 2009

Findings:

Table 2: Raw nest success (% total) for piping plover nests on the Missouri River during the 2005 – 2007 nesting seasons. Known successful nests had one or more chicks associated with the nest or the parents of the nest. Possibly successful nests did not have chicks associated with the nest or adults but eggs disappeared within 2 days of the projected hatch date. Failed nests did not have chicks associated with the nest or adults and disappeared outside the window of reasonable hatch dates (excluding abandoned nests).

Year	Total Nests	Known Successful	Possibly Successful	Failed Predation	Failed Abandoned	Failed Flooding	Failed Other [a]	Failed Unknown
2005	183	97 (53.0%)	12 (6.6%)	18 (9.8%)	7 (3.8%)	1 (0.5%)	5 (2.7%)	43 (23.5%)
2006	205	105 (51.2%)	6 (2.9%)	17 (8.3%)	21 (10.2%)	10 (4.9%)	4 (2.0%)	42 (20.5%)
2007 [b]	211	107 (50.7%)	18 (8.5%)	48 (22.7%)	15 (7.1%)	9 (4.3%)	2 (0.9%)	12 (5.7%)
Total	599	309 (51.6%)	36 (6.0%)	83 (13.9%)	43 (7.2%)	20 (3.3%)	11 (1.8%)	97 (16.2%)

[a] Includes bank erosion, weather events, and nests stepped on by researchers.

[b] Change in protocols led to more failures classified as predation. If eggs were missing before the hatch date and none of the other causes of failure were implicated, predation was thought the likely cause. The other frequent causes of nest loss, abandonment and flooding were obvious to observers. Many of the unknown nest losses from 2005 and 2006 were probably as a result of predation.

F.7 Stage Survival

F.7.1 Cohen et al. 2006

Adult Survival – Study Design – Sample Size:

Table 1. Number of adult Piping Plovers captured and number of resightings at West Hampton Dunes and Westhampton Beach, Long Island, NY, from 2001 to 2005.

Year	New captures[a]	New recruits[b]	Breeders resighted[c]	Nonbreeders resighted[d]	Nonbreeders missed[e]	Total population[f]
2001	36	0	0	0	0	36
2002	14	0	26	1	0	41
2003	16	3	21	10	0	50
2004	0	1	24	6	2	33
2005	0	1	6	6	—	13

[a]Number of birds trapped on the nest and marked in year i.

[b]Number of birds marked as chicks prior to year i that nested at our site for the first time in year i.

[c]Number of marked birds breeding at our site in year i that were marked prior to year i and nested at our site in year $i - 1$.

[d]Number of marked birds resighted in year i that did not breed at our site in year i, were marked prior to year i, and bred at the site in at least one year prior to year i.

[e]Number of marked birds not resighted in year i but resighted in at least 1 year after year i that were marked prior to year i, and bred at the site in at least 1 year prior to year i.

[f]Total number of marked birds known to be in the population in year i.

Table 3. Model-averaged estimates and unconditional confidence intervals on apparent survival rate (S), resighting rate (R), and apparent site fidelity (F) of adult Piping Plovers, West Hampton Dunes and Westhampton Beach, Long Island, NY 2001–2005 ($N = 71$ adults).

		Survival			Resighting rate			Site fidelity		
R'[a]	Year[b]	S	SE	95% CI	R	SE	95% CI	F	SE	95% CI
0.025[c]	2002	0.714	0.091	0.551–0.856	0.984	0.022	0.807–0.999	0.945	0.082	0.443–0.997
	2003	0.751	0.095	0.527–0.891	0.984	0.022	0.807–0.999	0.707	0.117	0.443–0.880
	2004	0.643	0.087	0.462–0.791	0.948	0.051	0.660–0.994	0.830	0.084	0.601–0.940
	Mean	0.703	0.032	0.637–0.762	0.972	0.012	0.936–0.988	0.827	0.069	0.650–0.925
0.000[d]	2002	0.731	0.092	0.521–0.871	0.984	0.022	0.797–0.999	0.904	0.094	0.532–0.987
	2003	0.753	0.091	0.539–0.888	0.984	0.022	0.797–0.999	0.702	0.116	0.443–0.874
	2004	0.651	0.087	0.468–0.798	0.946	0.060	0.635–0.994	0.812	0.084	0.595–0.927
	Mean	0.712	0.031	0.648–0.769	0.971	0.013	0.931–0.988	0.806	0.058	0.668–0.896
0.046[e]	2002	0.721	0.119	0.447–0.892	0.984	0.022	0.813–0.999	0.947	0.078	0.464–0.997
	2003	0.738	0.098	0.512–0.884	0.984	0.021	0.813–0.999	0.721	0.119	0.447–0.892
	2004	0.638	0.087	0.457–0.787	0.950	0.053	0.677–0.994	0.844	0.084	0.607–0.950
	Mean	0.699	0.031	0.635–0.756	0.973	0.011	0.941–0.988	0.837	0.065	0.669–0.929

[a] R' = The probability that a bird sighted between 15 March and 15 April died later in that interval without being reported dead. Because we fixed r (the probability of a bird being found dead and reported) at 0.0, R' equals the mortality rate of resighted birds in the territory establishment period (15 March to 15 April).

[b] Year of resighting.

[c] Assumes an annual survival rate of 0.74 (USFWS 1996) and constant monthly survival rates.

[d] Assumes no mortality of resighted birds in the territory establishment period.

[e] Assumes an annual survival rate of 0.74 and that mortality during the territory establishment period is twice as high as in other months.

F.7.2 Cohen and Gratto-Trevor 2011

First Winter Survival – Adult Survival:

Table 2. Mean Saskatchewan Piping Plover true survival (S), breeding/natal site fidelity (F), breeding site recapture probability (p), wintering grounds resighting probability (R = birds that survived to breed, R' = birds that did not), and re-immigration rate of emigrants (F'), 2002–2009.

Parameter	Age[a]	Estimate	SE	95% LCL	95% UCL
S	Adult	0.80	0.03	0.74	0.84
	Subadult	0.57	0.05	0.47	0.67
F	Adult	0.86	0.04	0.76	0.92
	Subadult	0.46	0.11	0.27	0.66
p	Adult	0.90	0.03	0.83	0.94
	Subadult	0.84	0.16	0.34	0.98
R	Adult	0.24	0.03	0.18	0.30
	Subadult	0.20	0.04	0.13	0.30
R'	Adult	0.09	0.01	0.07	0.11
	Subadult	0.07	0.03	0.02	0.17
F'	Adult	0.15	0.02	0.12	0.20
	Subadult	0.19	0.10	0.07	0.45

[a]Adult = AHY, Subadult = fledge to second year.

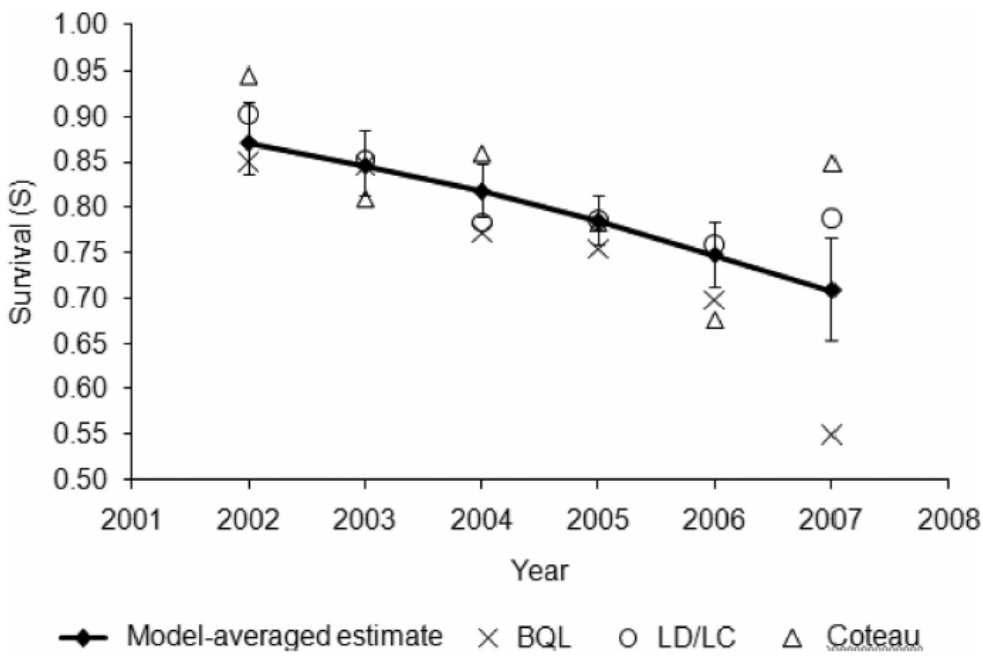


Fig. 2. Model-averaged estimates (with SE bars) and global model estimates (dots) of annual adult survival of Piping Plovers in Saskatchewan, 2002–2007. The global model, $S(g^{*}t)$, was not well supported based on information-theoretic criteria, probably because sample sizes were not adequate to estimate survival for each site x year combination precisely. However, we provide the estimates for reference. LD/LC = Lake Diefenbaker/Lake Chaplin and BQL = Big Quill Lake.

Sample Size:

Table 1. Number of Piping Plovers captured in Saskatchewan and number resighted in Saskatchewan and on the wintering grounds, 2002–2009.

Age	Year	New captures	Study area resightings	Winter resightings
Adult	2002	221	0	31
	2003	172	140	65
	2004	145	245	74
	2005	122	297	117
	2006	122	246	140
	2007	0	161	76
	2008	0	114	104
	2009	0	66	27
Subadult	2002	21	0	2
	2003	45	3	14
	2004	35	18	9
	2005	53	14	23
	2006	28	22	23
	2007	0	12	15
	2008	0	8	17
	2009	0	6	9

F.7.3 Larson et al. 2000

Adult Survival:

TABLE 2. Apparent local survival (ϕ) and resighting (p) probabilities for adult Piping Plovers at the Williams Preserve, North Dakota, 1984–1994, excluding 1991.

Year[a]	ϕ	SE(ϕ)	95% C.I.[b]	p	SE(p)	95% C.I.[b]
1984	1.000	0.354[c]	0.540–1.000[d]			
1985	0.868	0.142	0.369–0.987	0.125	0.139	0.012–0.633
1986	0.585	0.077	0.431–0.724	0.372	0.097	0.208–0.571
1987	0.506	0.079	0.355–0.656	0.664	0.084	0.485–0.805
1988	0.690	0.128	0.407–0.878	0.667	0.093	0.467–0.820
1989	1.000	0.498[c]	0.341–1.000[d]	0.495	0.109	0.294–0.697
1990	0.948[e]	0.240	0.001–1.000	0.315	0.092	0.166–0.514
1992	0.300	0.233	0.051–0.775	0.375	0.204	0.099–0.767
1993				0.500	0.297	0.089–0.911
Mean	0.737	0.092[f]	0.526–0.877	0.439	0.064[f]	0.319–0.566

[a] For values of ϕ, year corresponds to the time interval year to year + 1.

[b] Confidence intervals were calculated using a logit transformation (Lebreton et al. 1992).

[c] Using the sine link function in MARK resulted in SE(ϕ) = 0 for 1984–1985 and 1989–1990 (see Methods). Using the identity link function with the same model resulted in more realistic estimates of SE(ϕ) (reported in table) but biologically impossible estimates of ϕ (1.234 for 1984–1985 and 1.317 for 1989–1990).

[d] Confidence intervals for ϕ in 1984–1985 and 1989–1990 were taken from the model run using the identity link function (see footnote c and Methods) and were truncated at a maximum of 1. The statistical but biologically impossible upper C.I. limits were 1.927 and 2.292 for 1984–1985 and 1989–1990, respectively.

[e] This value of ϕ is for a 2-yr interval because there was no resighting effort in 1991.

[f] The SE of the mean ϕ and p represent total variability. The variance in ϕ and p over time (τ^2; i.e., with the sampling variance and covariance removed) derived from variance components models (see Methods) was 0.0445 and 0.0399 for ϕ when using the sine and identity link functions, respectively, and 0.020 for p.

Sample Size:

TABLE 1. Number of Piping Plovers captured and number of band resightings at the Williams Preserve, North Dakota, 1984–1994.

	Adults		Juveniles	
Year	New captures	Resightings	New captures	Resightings
1984	8			
1985	54	1	44	
1986	111	20	34	4
1987	31	64	65	7
1988		43		11
1989		22		6
1990		14		3
1991		0[a]		0[a]
1992		15		3
1993		6		3
1994		4		3
Total	204	189	143	40

[a] No attempt was made to observe color-banded plovers in 1991.

F.7.4 Ledee et al. 2010

Sample Size:

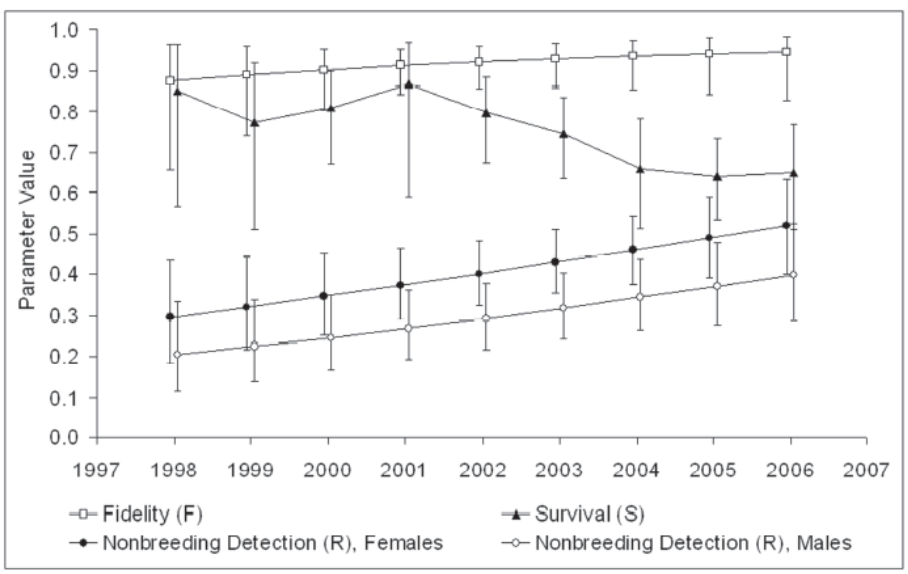


FIGURE 1. Model-averaged parameter estimates and 95% confidence intervals from Barker model for fidelity (F), survival (S), and nonbreeding detection probability (R) of Piping Plovers ($n = 150$) at the Great Lakes from 1998 to 2006.

F.7.5 Plissner and Haig 2000a

First Winter Survival – Adult Survival:

Table 1
Baseline metapopulation structure of Atlantic Coast piping plovers[a]

Population	Description	*K*	Initial population size	1st year/adult mortality	Mean no. fledged per pair
Atlantic metapopulation		4000	2540	51.64/26.13	1.37
1. Atlantic Canada	Newfoundland, Nova Scotia, Prince Edward Island, New Brunswick, Quebec, St. Pierre and Miquelon (France)	800	384	51.64/26.13	1.28
2. New England	Maine, Massachusetts, Rhode Island, Connecticut	1250	1118	51.64/26.13	1.75
3. Mid-Atlantic	New York, New Jersey	1150	718	51.64/26.13	1.07
4. Southern Region	Delaware, Maryland, Virginia, North Carolina, South Carolina	800	320	51.64/26.13	0.98

[a] Sources: personal communications/unpublished annual reports of authors listed in Acknowledgements.

Table 2
Baseline metapopulation structure of Great Lakes/Northern Great Plains piping plovers[a]

Population	Description	*K*	Initial population size	1st year/adult mortality	Mean no. fledged per pair
Metapopulation		5891	3332	56.8/34.0	1.25
1. Great Lakes	L. Superior, L. Michigan, L. Erie	300	48	56.8/34.0	1.46
2. Manitoba and Lake of the Woods	Manitoba and Lake of the Woods	250	73	56.8/34.0	0.53
3. Northern Prairie	Alberta, Saskatchewan Montana, and Dakotas, N and S Saskatchewan Rivers, include. L. Diefenbaker	4700	2786	56.8/34.0	1.28
4. Nebraska Rivers	Platte, Loup, Elkhorn, Niobrara, and Missouri below Fort Randall Dam	500	412	56.8/34.0	1.11
5. Colorado	Reservoirs in SE quadrant of state	40	13	56.8/34.0	0.73

[a] Sources: personal communications/unpublished annual reports of authors listed in Acknowledgements.

F.7.6 Roche et al. 2010c

First Winter Survival – Adult Survival:

Table 3. Geometric mean and 95% confidence intervals of apparent survival and resighting probabilities for each study in North America (Big Quill Lake: 2002–2006, Lake Diefenbaker: 2002–2008, Prairie Coteau: 2002–2008, Great Lakes: 1998–2008, Atlantic Canada: 1998–2003; Long Island, New York: 2001–2005). We made estimates from real parameter estimates generated via model-averaging and comparing Φ_{AHY} among breeding populations of piping plovers. We excluded terminal estimates of Φ_{AHY} for Atlantic Canada, the Canadian Prairie populations, and Long Island when generating mean estimates.

	Annual survival (Φ)			Resighting (p)		
Population	Estimate	Lower CI	Upper CI	Estimate	Lower CI	Upper CI
Big Quill Lake, SK	0.81	0.72	0.87	0.90	0.75	0.96
Prairie Coteau, SK	0.69	0.62	0.75	0.88	0.83	0.92
Lake Diefenbaker, SK	0.76	0.72	0.02	0.81	0.71	0.88
Missouri River, SD	0.80	0.74	0.84	0.97	0.94	0.99
Great Lakes, MI	0.71	0.63	0.76	0.97	0.89	0.99
Atlantic Canada	0.66	0.56	0.75	0.42	0.25	0.62
Long Island, NY	0.58	0.48	0.68	1.00	1.00	1.00

F.7.7 Root et al. 1992

Adult Survival:

TABLE 2. Jolly-Seber survival (S) and sighting (*P*) probabilities for adult Piping Plovers at the John E. Williams Memorial Nature Preserve, North Dakota, 1984–1990.

Year[a]	S	SE (S)	95% CL	*P*	SE (*P*)
1984	0.586	0.222	0.150–1.022		
1985	0.933	0.134	0.671–1.200	0.640	0.267
1986	0.563	0.063	0.439–0.686	0.305	0.074
1987	0.619	0.097	0.429–0.810	0.670	0.070
1988	0.617	0.159	0.306–0.928	0.523	0.087
1989				0.482	0.122
Mean	0.664	0.057	0.552–0.775	0.524	0.021

[a] For values of S, year corresponds to the time interval year–year + 1.

Study Design – Sample Size:

TABLE 1. Program JOLLY capture-recapture input data (Leslie Method B Table format, Brownie et al. 1986) for 214 adult Piping Plovers at the John E. Williams Memorial Nature Preserve, North Dakota, 1984–1990. Entries show the number of plovers resighted in year *j* that were last resighted in year *i*.

Year last resighted (*i*)	Year resighted (*j*)						
	1984	1985	1986	1987	1988	1989	1990
1984	0	3	1	0	0	0	0
1985		0	16	12	6	0	0
1986			0	51	4	3	1
1987				0	30	8	3
1988					0	15	5
1989						0	8
Resighted	0	3	17	63	40	26	17
Captured	8	55	111	29	10	1	0

F.7.8 Ryan et al. 1993

First Winter Survival – Adult Survival:

Table 1. Means of Great Plains Piping Plover reproductive rate, adult survival rate, and four levels of immature survival rate used in population simulations, with projected extirpation times.

Reproductive rate ($\bar{x} \pm SE$)[a]	*Adult survival ($\bar{x} \pm SE$)*[b]	*Immature survival ($\bar{x} \pm SE$)*[c]	*Years to extirpation (min–max)*[d]
0.86 ± 0.09	0.66 ± 0.06	0.46 ± 0.06	44 (25–54)
		0.53 ± 0.06	56 (28–77)
		0.60 ± 0.06	81 (29–105)
		0.66 ± 0.06	120 (36–148)

[a] *n = 32; sources: Haig, personal communication; L. Weber, personal communication; Wiens & Cuthbert 1984; Whyte 1985; Wiens 1986; Haig 1987; Haig & Oring 1987; Prindiville-Gaines & Ryan 1988; Schwalbach 1988; Mayer 1990, 1991.*
[b] *Source: Root et al. 1992.*
[c] *Mean values = 70%, 80%, 90%, and 100% of adult survival; see text for additional explanation.*
[d] *Value is the first year the mean population size = 0, based on simulations with 500 replications/year; minimum is first year one of the 500 replications = 0; maximum is first year all 500 replications = 0.*

Table 2. Reproductive and survival rates necessary for stabilization and growth of the Great Plains Piping Plover population, with reproductive rate or survival rates alternately held constant (dashes within columns).

Population status	*Chicks fledged per pair (% increase)*	*Adult survival (% increase)*	*Immature survival (% increase)*
Recent[a]	0.86	0.66	0.60
Stable	1.13 (31.4)	——	——
Stable	——	0.72 (8.7)	0.65 (8.7)
1% Growth	1.16 (34.8)	——	——
1% Growth	——	0.73 (9.6)	0.66 (9.7)
2% Growth	1.19 (38.4)	——	——
2% Growth	——	0.74 (10.7)	0.67 (10.9)

[a] *Recent population decline = 7.6%; see Table 1 for sources.*

F.7.9 Catlin et al. 2011a

Nest Survival:

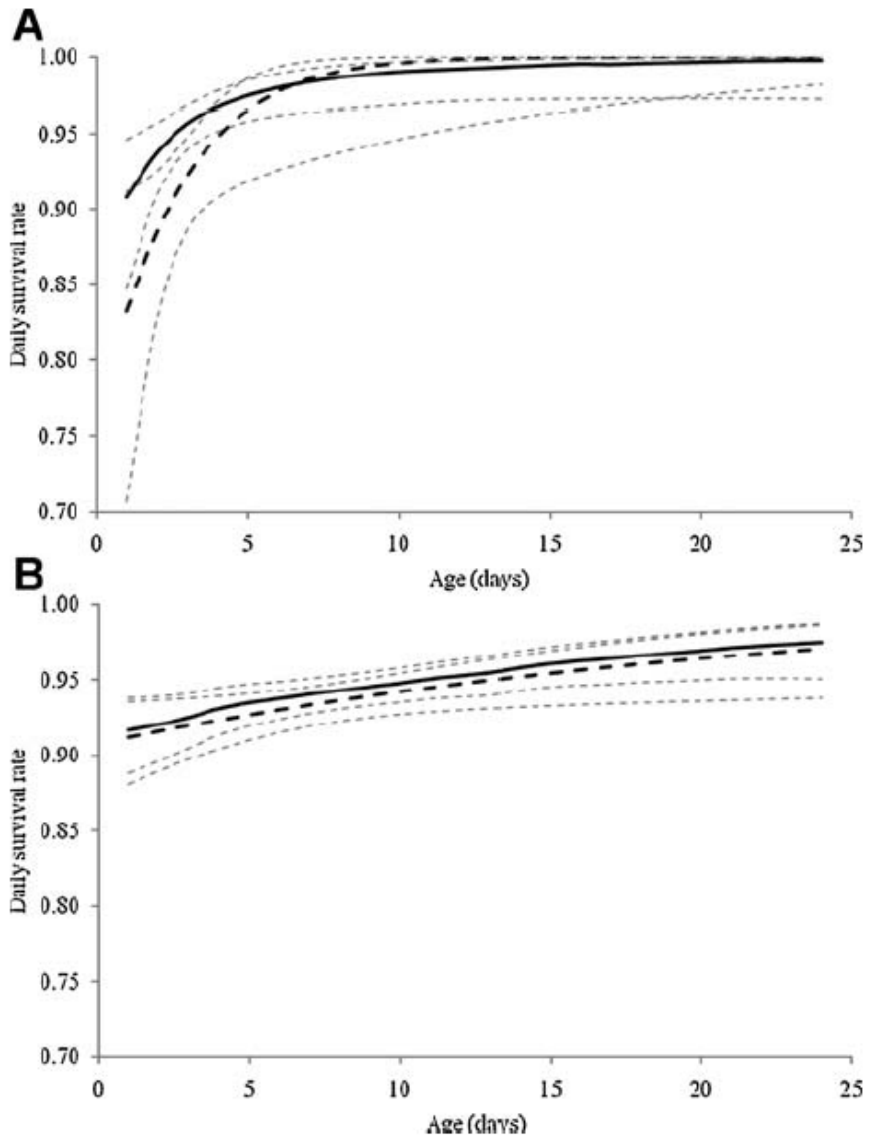


Figure 1. Survival of prefledge piping plover chicks on the Missouri River in 2008 (A) and 2009 (B). Estimates are for chicks where no owl was captured prior to fledging (black dashed line) and for those experiencing average owl removal for the year (black solid line). Estimates and 95% confidence intervals (gray dashed lines) are from the top-ranked survival model. We calculated these values for the mean hatch date for a given year.

F.7.10 Le Fer et al. 2008a

Sample Size:

Table 4. Piping plover mean chick daily survival rates ± standard error in North and South Dakota, USA, 2001–2003. Results of 2-way analyses of variance comparing daily survival rates among sites.[a] Sample size is the number of broods.

Site	n	$\bar{x}$	SE
Alkali	23	0.932 AB[b]	0.024
Hypolimnetic	22	0.985 A	0.005
Epilimnetic	29	0.853 B	0.042
Reservoir	20	0.923 AB	0.025

[a] Site: $F_{3,94} = 3.25$, $P = 0.03$; yr: $F_{2,94} = 0.01$, $P = 0.99$; site × yr: $F_{6,94} = 0.97$, $P = 0.45$.

[b] Sites with the same letter are not significantly different ($\alpha = 0.05$).

F.7.11 Murphy et al. 2003b

Adult Survival (Predation):

Table 1. Predation on adult Piping Plovers at predator exclosure cages at alkali lakes in five areas across the northern Great Plains during 1993-2002: number of applications of cages at which plovers were killed (N_d) and total number of applications of cages (N_{tot}).

	Hanna		Reflex Lakes		Stateline		Lostwood		Williams Preserve	
Year	N_d	N_{tot}	N_d	N_{tot}	N_d	N_{tot}	N_d	N_{tot}	N_d	N_{tot}
1993							0	8		
1994							0	9		
1995			1	20	0	2	0	30	0	9
1996			0	17	0	21	10	51		
1997			1	15	0	35	0	46	3	15
1998	0	12	3	28	0	44	0	114	6	34
1999	13	27	14	41	0	103	11	125	0	23
2000	0	4	4	35	0	40	0	56	0	25
2001					0	75	1	78	0	44
2002					0	73	1	56	0	40
Total	13	43	23	156	0	393	23	573	9	190
%	30		15		0		4		5	

F.7.12 Roche et al. 2010a

Adult Survival:

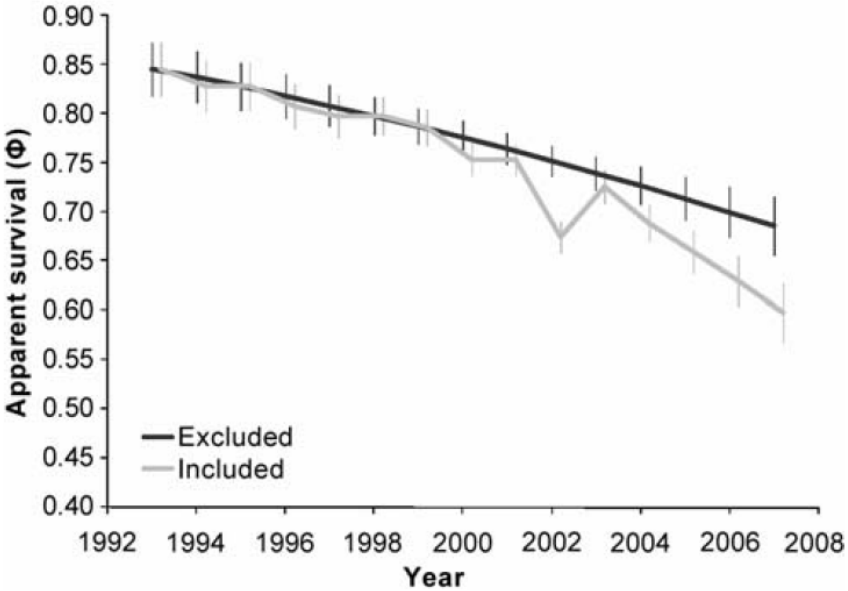


FIG. 3. Model-based estimates and 95% confidence intervals of annual survival of Great Lakes Piping Plovers when average annual losses due to disappearance and desertion are included (gray lines) versus excluded (black lines) from calculations, based on the best-supported model from Table 3.

F.7.13 Roche et al. 2008

First Winter Survival:

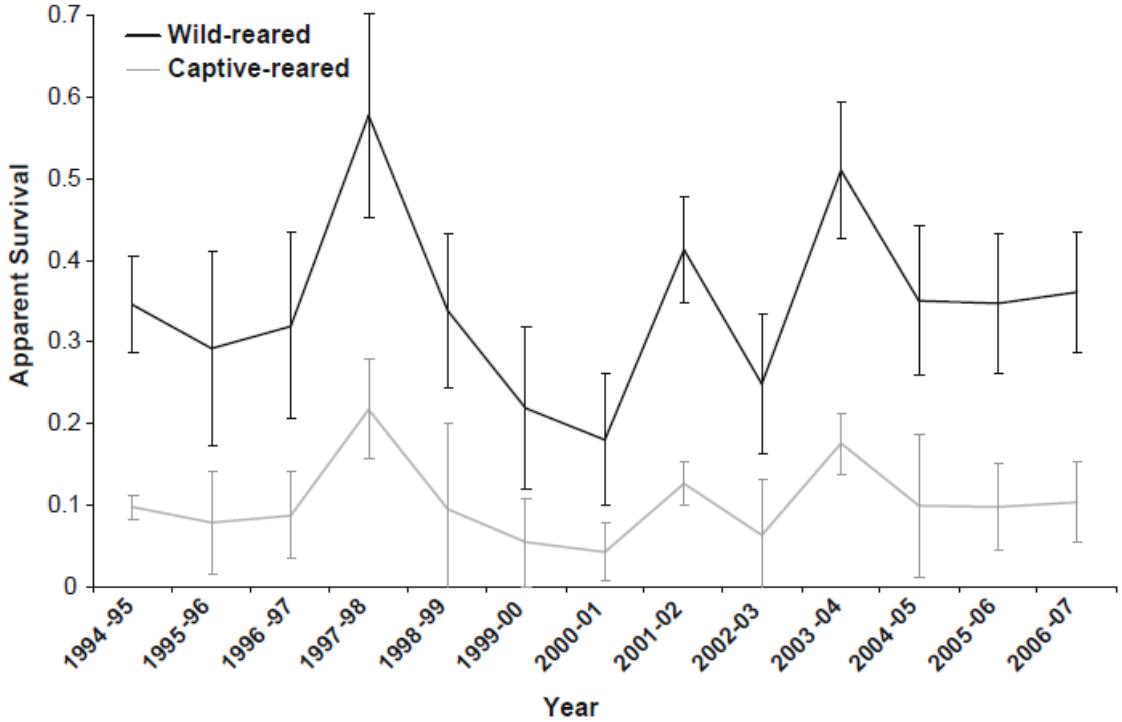


Fig. 1 – Model-based estimates of annual apparent survival (Φ_1, 95% CI) from the best-supported MARK model for first year captive-reared versus wild-reared piping plovers banded at 25 and 15 days of age, respectively.

Adult Survival:

	C				W			
	1	2	3	4+	1	2	3	4+
Breeding Propensity	0.78	0.58	0.83	0.96	0.63	0.58	0.83	0.96
Fledging Rate	0.58	0.58	0.58	0.58	0.91	0.91	0.91	0.91
Stage 1 Survival	0.08	0.08	0.08	0.08	0.29	0.29	0.29	0.29
Stage 2+ Survival	0.68	0.68	0.68	0.68	0.78	0.78	0.78	0.78
Fecundity	0.04	0.23	0.33	0.38	0.17	0.41	0.59	0.68
Lambda				0.71				1.00

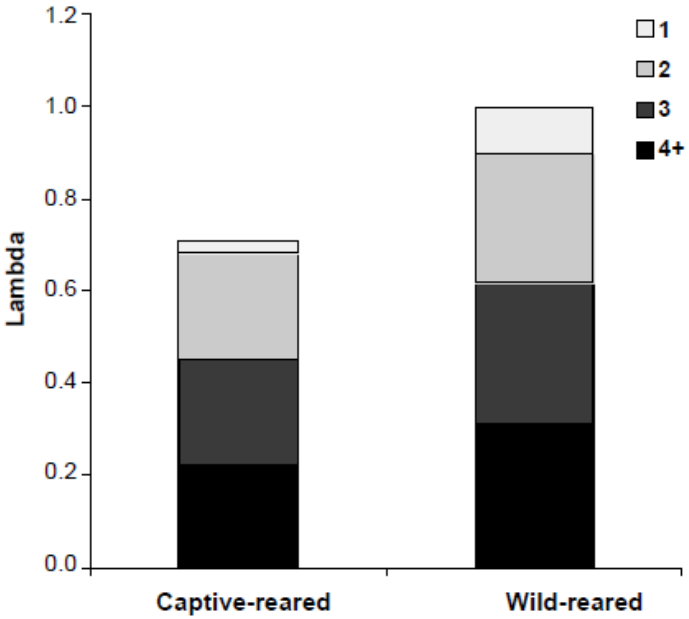


Fig. 2 – Age-specific reproductive contributions of captive-reared and wild-reared piping plovers in the first four years following hatch. Values are depicted as age-specific proportions of the total contribution to the finite rate of population change (lambda) for each group.

Study Design (Model-based estimates):

Table 3 – Mean parameter estimates from the best-supported MARK model comparing apparent annual survival and detection rates for captive-reared (C) and wild-reared (W) piping plovers

	Apparent survival (Φ)				Detection (ρ)							
	1		2+		2		3		4		5+	
	Est.	SE	Est.	SE	Est.	SE	Est.	SE	Est.	SE	Est.	SE
C	0.081	0.050	0.682	0.052	0.808	0.153	0.580	0.105	0.834	0.062	0.962	0.019
W	0.293	0.087	0.780	0.030	0.293	0.093	0.580	0.105	0.834	0.062	0.959	0.021

Stage 1 survival estimates are based on W plovers banded at 15 days of age and C plovers released at 25 days of age.

F.7.14 Sidle et al. 1992

Nest Survival:

Table 1. The number of monitored Least Tern (LT) and Piping Plover (PP) nests inundated along the Platte River between the Loup River confluence and the Missouri River, 1986-90.

Yr	No. Nests Before First Flood	No. Flooded	% Flooded	No. Renests[1]	No. Renests Flooded	Total % Flooded	Maximum Daily Flow (cms) NB/LV[2]
	LT/PP	LT/PP	LT/PP	LT/PP	LT/PP	LT/PP	
1986[3]	136/28	18/6	13/21	0/0	0/0	13/21	421.7/793.0
1987[4]	69/17	5/0	7/0	0/0	0/0	7/0	302.8/484.3
1988[5]	77/32	3/1	4/3	0/0	0/0	4/3	143.2/162.6
1989[5]	61/13	21/7	34/54	24/5	0/0	28/39	331.1/320.0
1990[5]	94/28	94/28	100/100	93/13	4/6	52/85	911.3/1713.3

[1]Least terns and piping plovers renested after losing eggs or chicks from other causes in 1986 and 1988 but initiated no new nests on the river after flooding.
[2]Flows measured at U.S. Geological Survey gauging stations at North Bend and Louisville, Nebraska.
[3]Flooding occurred in late June after most nests had hatched.
[4]Flooding occurred in late May-early June before birds had initiated many nests.
[5]The first flood inundated sandbars along the entire river and occurred during peak nesting. The second flood inundated sandbars below Salt Creek (U.S. Army Corps of Engineers river mile 26) and occurred in late July after most tern but no plover renests had hatched. The second flood killed all chicks on sandbars below river mile 26.

F.7.15 Catlin et al. 2011b

Nest Survival:

Table 2. Raw nest success (% total) for piping plover nests on the Missouri River during the 2005–2007 nesting seasons. Known successful nests had ≥1 chick associated with the nest or the parents of the nest. Possibly successful nests did not have chicks associated with the nest or adults but eggs disappeared within 2 days of the projected hatch date. Failed nests did not have chicks associated with the nest or adults and disappeared outside the window of reasonable hatch dates (excluding abandoned nests).

			Known successful		Possibly successful		Failed predation		Failed abandoned		Failed flooding		Failed other[a]		Failed unknown	
Yr	Habitat type	Total nests	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%
2005	Natural	76	33	43.4	8	10.5	6	7.9	3	3.9	0	0.0	0	0.0	26	34.2
	Managed	36	12	33.3	2	5.6	12	33.3	2	5.6	0	0.0	0	0.0	8	22.2
	Engineered	71	52	73.2	2	2.8	0	0.0	2	2.8	1	1.4	5	7.0	9	12.7
2006	Natural	47	14	29.8	1	2.1	7	14.9	5	10.6	9	19.1	0	0.0	11	23.4
	Managed	31	7	22.6	1	3.2	8	25.8	3	9.7	1	3.2	0	0.0	11	35.5
	Engineered	127	84	66.1	4	3.1	2	1.6	13	10.2	0	0.0	4	3.1	20	15.7
2007	Natural	54	21	38.9	7	13.0	14[b]	25.9	2	3.7	6	11.1	1	1.9	3	5.6
	Managed	16	6	37.5	4	25.0	4[b]	25.0	1	6.3	0	0.0	0	0.0	1	6.3
	Engineered	141	80	56.7	8	5.7	29[b]	20.6	12	8.5	3	2.1	1	0.7	8	5.7
Total		599	309	51.6	37	6.2	82	13.7	43	7.2	20	3.3	11	1.8	97	16.2

[a] Includes bank erosion, weather events, and nests stepped on by researchers.
[b] Change in protocols led to more failures classified as predation. If eggs were missing before the hatch date and none of the other causes of failure were implicated, we concluded that predation was the cause of failure. The other frequent causes of nest loss, abandonment and flooding, were obvious to observers. Many of the unknown nest losses from 2005 and 2006 were probably as a result of predation.

F.7.16 Elias et al. 2000

Table 4. Number of piping plover pairs, chicks hatched and fledged, and modified Mayfield daily survival rate (DSR) estimates of chick survival, by beach segment type, on New York barrier islands, 1992 and 1993.

		Chicks					
Beach segment type[a]	No. pairs	Hatched	Fledged	% Fledged	Fledged/pair	DSR[b] (±SE)	S[c]
1992							
Ephemeral pools available	30	69	41	59%	1.37	0.979 (0.004) A	58%
Bay tidal flats available	2	1	0	0%	0.00	0.000	0%
Ephemeral pools and bay tidal flats unavailable	47	91	38	42%	0.81	0.959 (0.008) B	35%
1993							
Ephemeral pools available	38	101	55	54%	1.45	0.975 (0.006) A	53%
Bay tidal flats available	8	23	13	57%	1.63	0.972 (0.017) A	49%
Ephemeral pools and bay tidal flats unavailable	36	42	21	50%	0.58	0.970 (0.008) A	45%
1992–93 totals	161	327	168	51%	1.04	0.971 (0.003)	48%

[a] Habitats available on all beach segment types were: ocean intertidal zone, wrack, backshore, open vegetation, and interdune, except on bay tidal flat segments.
[b] Chi-square test for homogeneity of survival among habitats: 1992: $\chi^2 = 4.26$, 1 df, $P < 0.05$; 1993: $\chi^2 = 0.33$, 2 df, $P > 0.05$. Means with the same letters are not significantly different.
[c] The interval from hatching to fledging is 25 days; interval survival (S) = $DSR^{25} \times 100$.

F.7.17 Wemmer et al. 2001

First Winter Survival – Adult Survival:

Table 1
Survival estimates for piping plovers

Survival rate	Adult	Fledgling–age 1
Empirical for Michigan[a]	0.73	0.24–0.312
Average other sources[b]	0.74	0.32–0.34
Min all sources	0.64	0.17
Max all sources	0.94	0.48

[a] Wemmer, unpublished data.
[b] Loegering (1992), Root et al. (1992), Wiens (1986), Cross, MacIvor, in USFWS (1996).

F.7.18 Cohen et al. 2009

Survival:

Table 5. Pearson's correlation (r = coeff.) of mean reproductive variables with population density (pairs/ha) for piping plovers at West Hampton Dunes (WHD) and the reference area (REF), New York, USA, 1994–2004 (n = 11 yr).

	WHD		REF	
Variable	***r***	***P***	***r***	***P***
First nest clutch size	−0.08	0.820	−0.460	0.155
Second nest clutch size	−0.38	0.256	−0.15	0.681
Renest probability	0.30	0.370	0.10	0.792
Nest survival	−0.07	0.766	0.05	0.879
Chick survival	−0.03	0.834	0.39	0.230
Chicks fledged/pair	−0.05	0.891	0.11	0.743

First Winter Survival:

Table 9. Chick survival (chicks fledged/eggs hatched, yearly means) of piping plovers (Mayfield interval) for broods foraging on the bay side, ocean side, or both the ocean and bay side at West Hampton Dunes, New York, USA, 1994–2004.

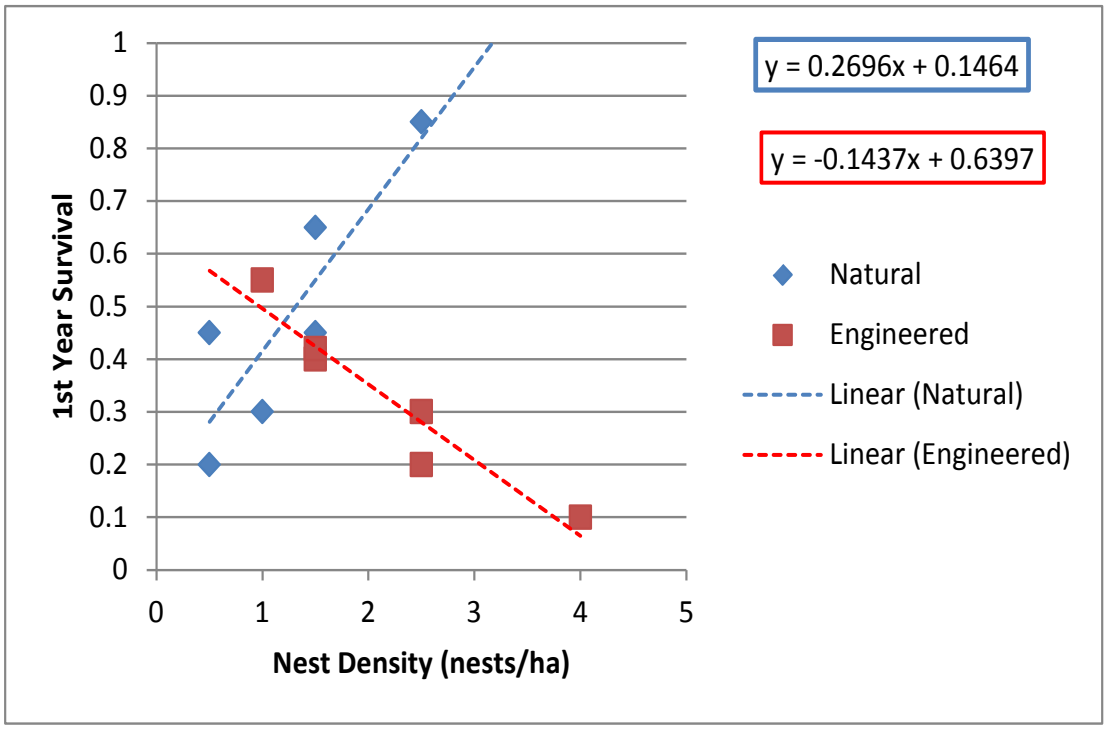
	Brood foraging location[b]								
	Ocean side only			Bay side only			Both sides		
Yr[a]	*n*	$\bar{x}$	SE	*n*	$\bar{x}$	SE	*n*	$\bar{x}$	SE
1994–1996	5	0.29 Bb	0.10	28	0.52 Ba	0.10	0		
1997–2000	30	0.56 Ab	0.08	34	0.65 Aa	0.07	33	0.49 Ac	0.09
2001–2004	50	0.50 Aa	0.08	11	0.01 Cb	0.02	21	0.36 Aa	0.10

[a] 1994–1996 = prior to United States Army Corps of Engineers (USACE) Interim Project; 1997–2000 = USACE Interim Project, predators trapped; 2001–2004 = after first renourishment, predator trapping curtailed.

[b] Within columns, means with the same capital letter are not different. Within rows, means with the same lowercase letter are not different (Mixed Mayfield logistic regression with random yr effect, $F_{7,109} = 18.43$, $P < 0.001$. We evaluated pairwise differences by least-squares means contrasts).

F.7.19 Catlin 2009

First Winter Survival:



Adult Survival:

Table 10: Estimates of true survival for breeding piping plovers.

Study	Estimates (CI_{95})	Years	Population
LeDee (2008)	0.91 (0.63 – 0.98)	1998 – 2005	Increasing[A]
This study	0.822 (0.747 – 0.878)	2005 – 2006	Stationary[C]
	0.824 (0.736 – 0.887)	2006 – 2007	
Cohen et al. (2006)	0.714 (0.551 – 0.856)	2002 – 2003	Decreasing[B]
	0.751 (0.527 – 0.891)	2003 – 2004	
	0.643 (0.462 – 0.791)	2004 – 2005	

[A] Haig et al. (2005)

[B] Cohen et al., in press

[C] G. Pavelka, USACE, personal communication

F.7.20 Brudney et al. 2013

Nest Survival – First Winter Survival:

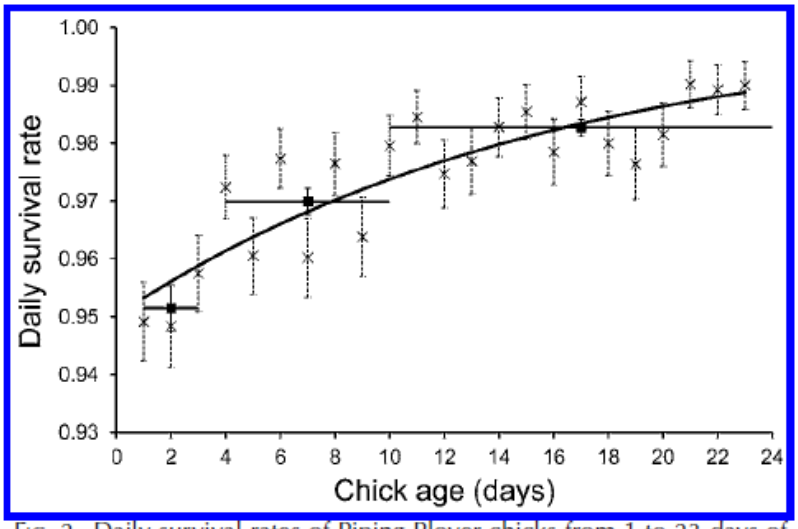


FIG. 2. Daily survival rates of Piping Plover chicks from 1 to 23 days of age in the Great Lakes region, 1992–2011. Individual points denoted by "×" represent daily survival rates under the fully age-dependent model $S_{(1,2,3...23)}$ with their associated 85% confidence intervals (CIs), filled squares are from the three-stage model $S_{(1-3,4-10,11-23)}$, and the curvilinear line represents the logistic regression model: logit(S) = $\beta_0 + \beta_1$*age, where β_0 = 2.948 (85% CI: 2.868–3.028) and β_1 = 0.067 (85% CI: 0.059–0.075).

F.8 Site Fidelity, Immigration, and Emigration

F.8.1 Gratto-Trevor et al. 2012

Findings:

Table 1. Banding locations and numbers of piping plovers uniquely color marked, for each breeding population.

Breeding population	Provinces or states	Chief investigators	Years of marking	No. chicks uniquely color marked	No. adults uniquely color marked	% adults resighted in winter[a]	2006 population estimate[b]
Eastern Canada	NS, NB, PE, NL, QC	D. Amirault-Langlais and F. Schaffer	1998–2003	259	224	12	457
Great Lakes	MI, WI	F. Cuthbert	1993–2008	0[c]	347	43	109
Great Plains	SD, NE	D. Catlin and J. Fraser	2005–2008	1,140	451	25	2,959
Prairie Canada	SK	C. Gratto-Trevor	2002–2006	455	792	33	1,703

[a] Birds marked as adults and chicks were used in all analyses, except % resighted in winter (see Methods).
[b] Elliott-Smith et al. (2009).
[c] Great Lakes chicks were not uniquely color marked.

Table 2. Percentage of individually marked piping plovers from each breeding population observed in different wintering areas along the east coast of the United States and into Mexico up to December 2008. Each individual was counted only once.

	% (*N*) of piping plovers from a breeding location at different wintering areas							
Winter area	Eastern Canada		Great Lakes		Great Plains		Prairie Canada	
Atlantic Central	43	(20)	19	(28)	0	(0)	1	(3)
Atlantic South	34	(16)	56	(84)	7	(12)	3	(12)
Gulf Florida South	23	(11)	19	(29)	14	(23)	4	(15)
Gulf Florida North and Alabama	0	(0)	4	(6)	8	(13)	4	(16)
Mississippi and Louisiana	0	(0)	1	(1)	8	(13)	5	(17)
Texas North	0	(0)	0	(0)	12	(21)	17	(61)
Texas South and Mexico	0	(0)	1	(2)	51	(87)	66	(241)
Total		(47)		(150)		(169)		(356)

F.8.2 Haig and Oring 1988a

Findings:

TABLE 1. Breeding-site fidelity and natal philopatry in Piping Plovers.

Study location	Nesting adults banded	*n* return (%)	Fledged chicks banded	*n* return (%)	Focal sites	Years of study	Source
Southern Manitoba	65	44 (67.7)	90	5 (5.5)[a] 11 (12.2)[b]	5	4	This study
Cape Cod, Massachusetts	16	11 (68.9)	28	0 (0.0)[a] 12 (42.9)[b]	12	3	MacIvor et al. 1987
Waugoshance Pt., Michigan	16	9 (56.3)	35	1 (2.9)[a] 8 (2.9)[b]	1–10	11	Pike 1985, pers. comm.
Lake of the Woods, Minnesota	47	32 (68.0)	70	15 (21.0)	4	3	Wiens 1986
Long Island, New York	1,173	288 (24.6)	979	34 (3.4)[a] 47 (4.8)[b]	3	20	Wilcox 1959
Chain of Lakes, North Dakota	111	32 (55.2)	123	7 (6.0)	7	2	Mayer & Ryan 1986
Cadden Beach, Nova Scotia	19	7 (36.8)	39–57	1 (1.6–2.6)	1	2	Cairns 1982
Big Quill Lake, Saskatchewan	14	6 (42.0)	12	1 (8.3)	1	2	Whyte (1985)
Total	1,461	429 (29.4)	1,376–1,394	64 (4.7)[a] 102 (7.3–7.4)[b]			

[a] Return to natal site.
[b] Return to local area (including natal site). In all cases, except Whyte 1984, local sites were surveyed in addition to focal sites.

TABLE 2. Dispersal of breeding Piping Plovers between years.[a]

State or province	Breeding site 1	Breeding site 2	Sex[b] M	F	U	Km dispersed
Manitoba	Clandeboye Bay, Lake Manitoba	Stony Beach, Lake Manitoba		1		3
	Clandeboye Bay, Lake Manitoba	West Shoal Lake	3	3		32
	Stony Beach, Lake Manitoba	Twin Lakes Beach, Lake Manitoba		1		15
	S. West Shoal Lake	N. West Shoal Lake			8	8
	West Shoal Lake	Clandeboye Bay, Lake Manitoba	2	3		32
	West Shoal Lake	Grand Marais, Lake Winnipeg	1			70
	West Shoal Lake	Twin Lakes Beach, Lake Winnipeg	2	2		25
Massachusetts	Plymouth Beach	Sandy Neck		1		37
Minnesota	Lake of the Woods	Clandeboye Bay, Lake Manitoba	1			314
	Lake of the Woods	Long Point, Lake Winnipeg	1			546
	Lake of the Woods	West Shoal Lake	1	1		273
New York	Moriches	Shinnecock		1		18
	Shinnecock	Moriches		1		25
	Shinnecock	Mecox	1			14
Ontario	Long Point, Lake Erie	Waugoshance Pt., Lake Michigan			1	595

[a] Sources: this study (Manitoba, Minnesota), Wilcox 1959 (New York), MacIvor et al. 1987 (Massachusetts), Pike 1985 (Michigan).

[b] Values represent the number of males, females, and unknown-sex birds that moved from one breeding site to another in successive years.

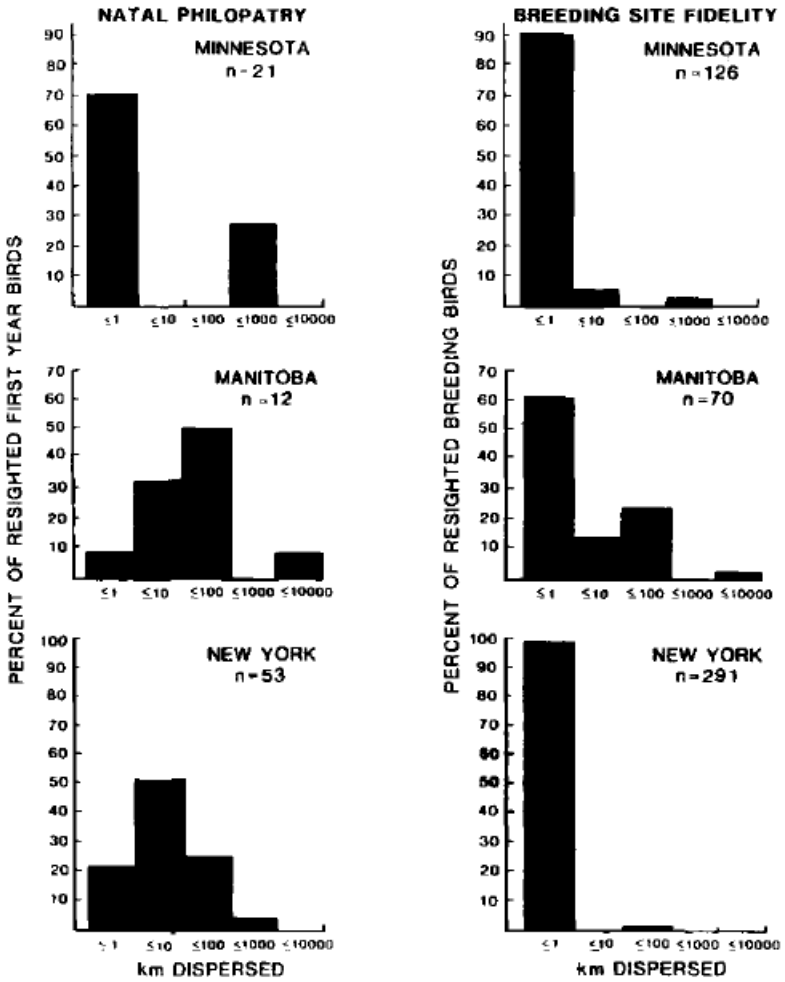


Fig. 2. Comparison of distances dispersed from natal sites in Piping Plover chicks resighted their second year (left column) and distances adults dispersed between breeding sites in successive years (right column). Distance intervals (km) on bars are: 0–1.0, 1.1–10.0, 10.1–100.0, 100.1–1,000.0, 1,000.1–10,000.0. Sources: this study, Wilcox 1959, Wiens 1986.

TABLE 3. Dispersal of Piping Plovers from natal sites to breeding sites.[a]

Hatch site	Breeding site	Sex[b] M	F	U	Km dispersed
Manitoba					
Grand Marais, Lake Winnipeg	West Shoal Lake	1	1	—	70
Stony Beach, Lake Manitoba	West Shoal Lake	—	1	1	35
Twin Lakes Beach, Lake Manitoba	West Shoal Lake	1	—	1	25
West Shoal Lake	West Shoal Lake	2	2	1	5
West Shoal Lake	Long Point, Lake Erie	1	—	—	1,500
Massachusetts					
Harding Beach	Monomoy	—	—	1	22
Michigan					
Waugoshance Pt., Lake Michigan	Cathead Bay, Lake Michigan	—	—	1	74
Waugoshance Pt., Lake Michigan	Grand Marais, Lake Superior	—	—	6	112
Minnesota					
Lake of the Woods	West Shoal Lake	—	2	—	273
Lake of the Wood	Grand Marais, Lake Winnipeg	1	2	1	222
New York					
Atlantic Beach	Shinnecock	—	1	—	101
Mecox	Shinnecock	2	2	—	9
Moriches	Shinnecock	2	3	—	25
Oak Beach	Shinnecock	—	1	—	62
Tobay Beach	Moriches	1	—	—	66
Penn Yan	Long Point, Lake Erie	—	—	1	240

[a] Sources: This study (Manitoba, Minnesota), MacIvor et al. 1985 (Massachusetts), Pike 1985 (Michigan), Wilcox 1959 (New York).
[b] Values represent numbers of males, females and unknown-sex first-year adults that dispersed from their natal site to a non-natal site to breed.

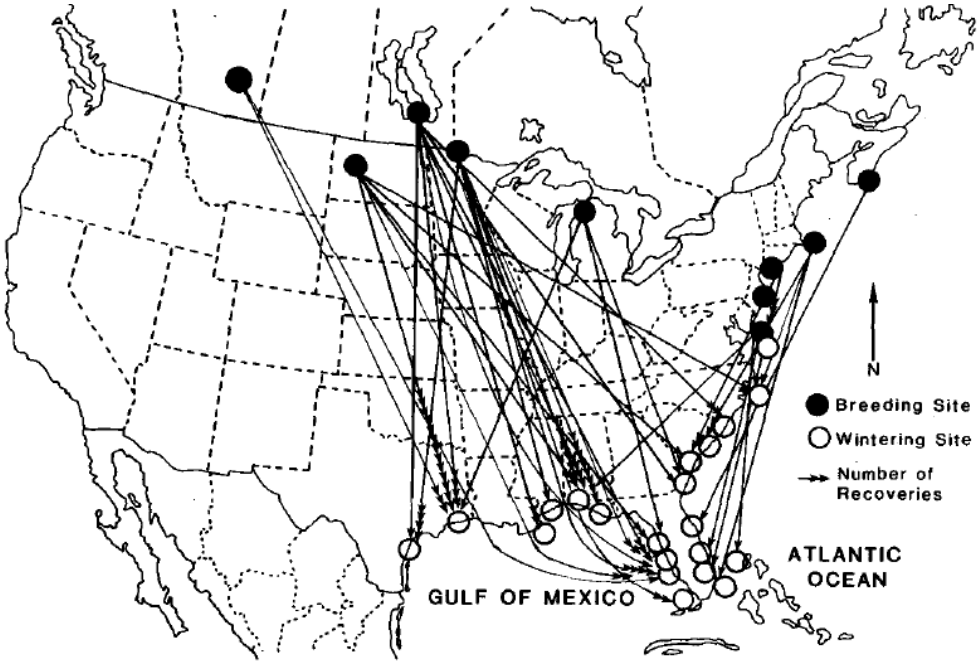


Fig. 3. Dispersal of Piping Plovers from breeding sites to wintering sites, based on 91 sightings of marked birds.

F.8.3 Haig and Oring 1988b

Findings:

TABLE 1. Annual individual reproductive effort of Piping Plovers breeding in southern Manitoba, 1982–1985.

	n	Mean	Range	SD
No. of mates	71	1.3	0–2	0.5
No. of nests	80	1.6	0–3	0.6
Eggs laid/female	78	6.2	0–12	2.3
Chicks hatched	92	1.7	0–4	1.7
Chicks fledged	94	0.9	0–4	1.4

TABLE 2. Population reproductive success of Piping Plovers at Lake Manitoba and West Shoal Lake, Manitoba, 1982–1985.

	West Shoal Lake	Lake Manitoba	Total
Years	1984–1985	1982–1985	1982–1985
Birds	61	37	98
Pairs	33	20	53
Nests	45	27	72
Chicks fledged per:			
Breeding pair	1.5	0.3	1.0
Pair with brood	3.1	1.3	2.7
Percentage pairs fledging chicks	48.5	20.0	37.7

TABLE 3. Breeding-site fidelity of Piping Plovers in southern Manitoba by previous year's reproductive success. Sample sizes (in parentheses) are the number of individuals monitored.

Previous success	Percentage return		
	Males	Females	Overall
Chicks hatched	73.3 (15)	54.5 (11)	65.4 (26)
Nests failed	76.5 (17)	57.1 (14)	67.7 (31)

TABLE 4. Interyear mate fidelity and territory retention of Piping Plovers breeding in southern Manitoba, 1982–1985. Sample sizes (in parentheses) are the number of individuals monitored. Asterisks indicate significant differences between males and females (χ^2, $P < 0.01$).

Pattern	Males (%)	Females (%)	Overall (%)
Changed mate	73.1 (26)	78.6 (14)	75.0 (40)
Changed territory	41.0 (39) *	85.7 (14)	52.8 (53)
Changed mate			
Changed territory	26.3 (19) *	100.0 (11)	53.3 (30)
Kept territory	73.7	0.0	46.7
Kept mate			
Changed territory	42.9 (7)	33.3 (3)	40.0 (10)
Kept territory	57.1	66.7	60.0

TABLE 5. Comparison of previous hatching success among Piping Plovers that retained territories in southern Manitoba with those that changed territories. Sample sizes (in parentheses) are the number of individuals monitored.

	Percentage that hatched chicks in previous year[a]		
Pattern	Males	Females	Combined
Kept territory	28.6 (14)	100.0 (2)	37.5 (16)
Changed territory	44.4 (9)	33.3 (12)	38.1 (21)

[a] Differences within sexes were not significant ($P \leq 0.05$).

TABLE 6. Intrayear mate and territory fidelity of Piping Plovers breeding in southern Manitoba, 1982–1985.[a] Sample sizes (in parentheses) are number of individuals monitored.

Pattern	Males (%)	Females (%)	Overall (%)
Changed territory	63.6 (22)	70.8 (24)	64.7 (46)
Changed mate	40.7 (27)	28.0 (25)	34.6 (52)
Changed mate			
Changed territory	62.5 (8)	100 (19)	88.9 (27)
Kept territory	37.5	0.0	11.1
Kept mate			
Changed territory	62.5 (16)	62.5 (16)	62.5 (32)
Kept territory	37.5	37.5	37.5

[a] Data represent birds that renested following nest failure.

TABLE 7. Intrayear pair-bond retention following destruction of Piping Plover nests in southern Manitoba, 1982–1985.[a]

Year	Pair bond: Dissolved (%)	Pair bond: Persisted (%)	n
1982	58.3	41.7	12
1983	33.3	66.7	6
1984	42.9	57.1	14
1985	15.0	85.0	20
1982–1985	34.6	65.4	52

[a] Data represent pairs in which at least one member of pair renested on a study site in Manitoba.

F.8.4 Cohen et al. 2006

Findings:

Table 1. Number of adult Piping Plovers captured and number of resightings at West Hampton Dunes and Westhampton Beach, Long Island, NY, from 2001 to 2005.

Year	New captures[a]	New recruits[b]	Breeders resighted[c]	Nonbreeders resighted[d]	Nonbreeders missed[e]	Total population[f]
2001	36	0	0	0	0	36
2002	14	0	26	1	0	41
2003	16	3	21	10	0	50
2004	0	1	24	6	2	33
2005	0	1	6	6	—	13

[a]Number of birds trapped on the nest and marked in year i.

[b]Number of birds marked as chicks prior to year i that nested at our site for the first time in year i.

[c]Number of marked birds breeding at our site in year i that were marked prior to year i and nested at our site in year $i - 1$.

[d]Number of marked birds resighted in year i that did not breed at our site in year i, were marked prior to year i, and bred at the site in at least one year prior to year i.

[e]Number of marked birds not resighted in year i but resighted in at least 1 year after year i that were marked prior to year i, and bred at the site in at least 1 year prior to year i.

[f]Total number of marked birds known to be in the population in year i.

F.8.5 Cohen and Gratto-Trevor 2011

Findings:

Table 2. Mean Saskatchewan Piping Plover true survival (S), breeding/natal site fidelity (F), breeding site recapture probability (p), wintering grounds resighting probability (R = birds that survived to breed, R' = birds that did not), and re-immigration rate of emigrants (F'), 2002–2009.

Parameter	Age[a]	Estimate	SE	95% LCL	95% UCL
S	Adult	0.80	0.03	0.74	0.84
	Subadult	0.57	0.05	0.47	0.67
F	Adult	0.86	0.04	0.76	0.92
	Subadult	0.46	0.11	0.27	0.66
p	Adult	0.90	0.03	0.83	0.94
	Subadult	0.84	0.16	0.34	0.98
R	Adult	0.24	0.03	0.18	0.30
	Subadult	0.20	0.04	0.13	0.30
R'	Adult	0.09	0.01	0.07	0.11
	Subadult	0.07	0.03	0.02	0.17
F'	Adult	0.15	0.02	0.12	0.20
	Subadult	0.19	0.10	0.07	0.45

[a]Adult = AHY, Subadult = fledge to second year.

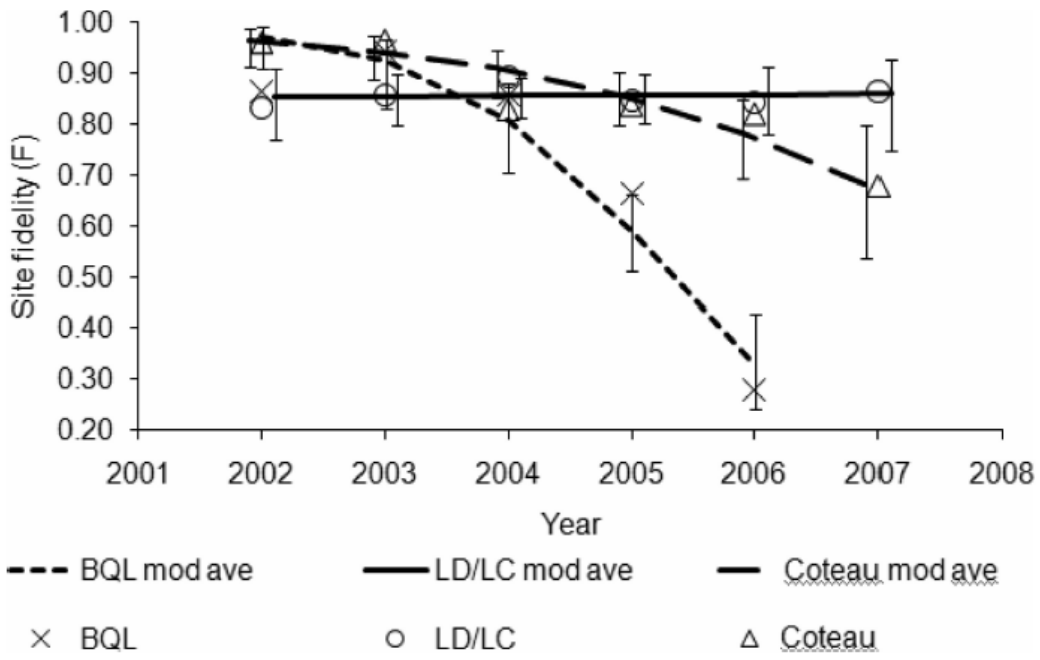


Fig. 3. Model-averaged (mod ave) estimates (with SE bars) and global model estimates (dots) of annual adult site fidelity of Piping Plovers in Saskatchewan, 2002–2009. The global model, S(g*t), was not well supported based on information-theoretic criteria, probably because sample sizes were not adequate to estimate survival for each site × year combination precisely. However, we provide the estimates for reference. LD/LC = Lake Diefenbaker/Lake Chaplin and BQL = Big Quill Lake.

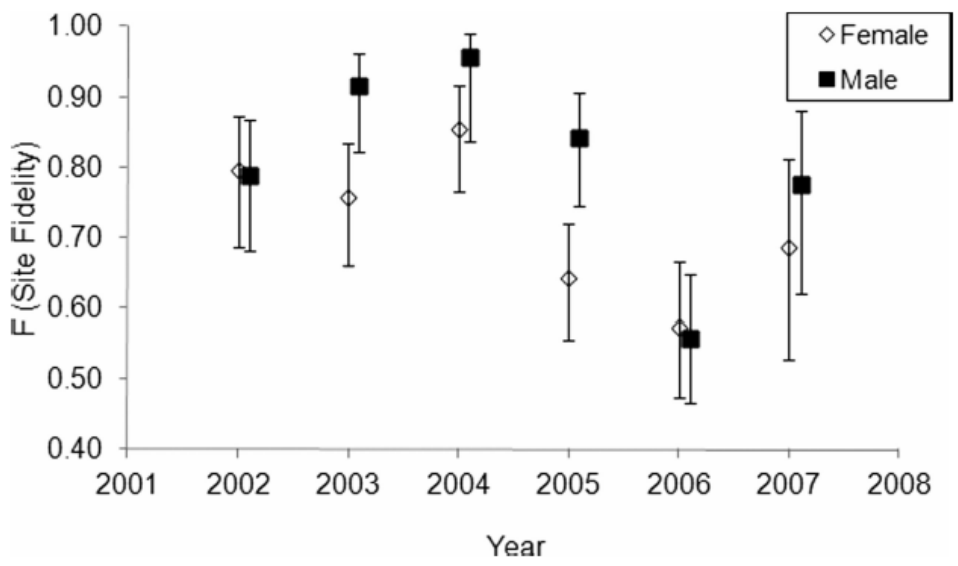


Fig. 4. Model-averaged estimates (with SE bars) of annual site fidelity of adult male and female Piping Plovers in Saskatchewan, 2002–2007.

F.8.6 Ledee et al. 2010

Findings:

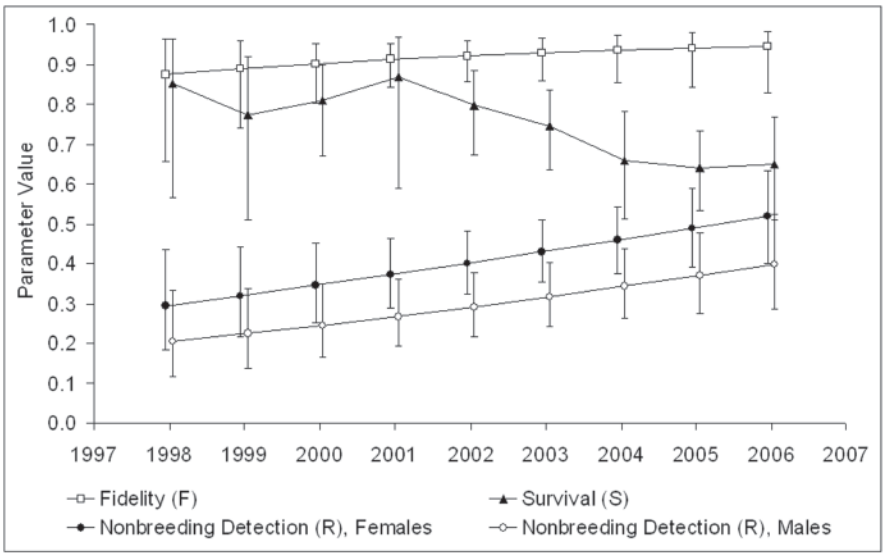


FIGURE 1. Model-averaged parameter estimates and 95% confidence intervals from Barker model for fidelity (*F*), survival (*S*), and nonbreeding detection probability (*R*) of Piping Plovers ($n = 150$) at the Great Lakes from 1998 to 2006.

F.8.7 Roche et al. 2012

Findings:

TABLE 2. Movement among breeding areas for 782 individually marked Piping Plovers in Saskatchewan, 2002–2009. The category "Origin" designates the population from which an individual Piping Plover emigrated, and "Destination" the population it immigrated to (e.g., there were 364 cases in which a Piping Plover "moved" between Big Quill Lake and Big Quill Lake, and 10 in which a Piping Plover moved between Big Quill Lake and Lake Diefenbaker). Each integer value represents the number of interannual movements that occurred between two areas; values in parentheses represent the percentage of total interannual movements per transition.

	Origin			
Destination	Big Quill Lake	Chaplin Lake	Missouri Coteau	Lake Diefenbaker
Big Quill Lake	364 (95%)	2 (1%)	2 (0.5%)	8 (1%)
Chaplin Lake	6 (2%)	140 (84%)	6 (1%)	57 (8%)
Missouri Coteau	2 (1%)	2 (1%)	398 (98%)	8 (1%)
Lake Diefenbaker	10 (3%)	22 (13%)	2 (0.5%)	651 (90%)

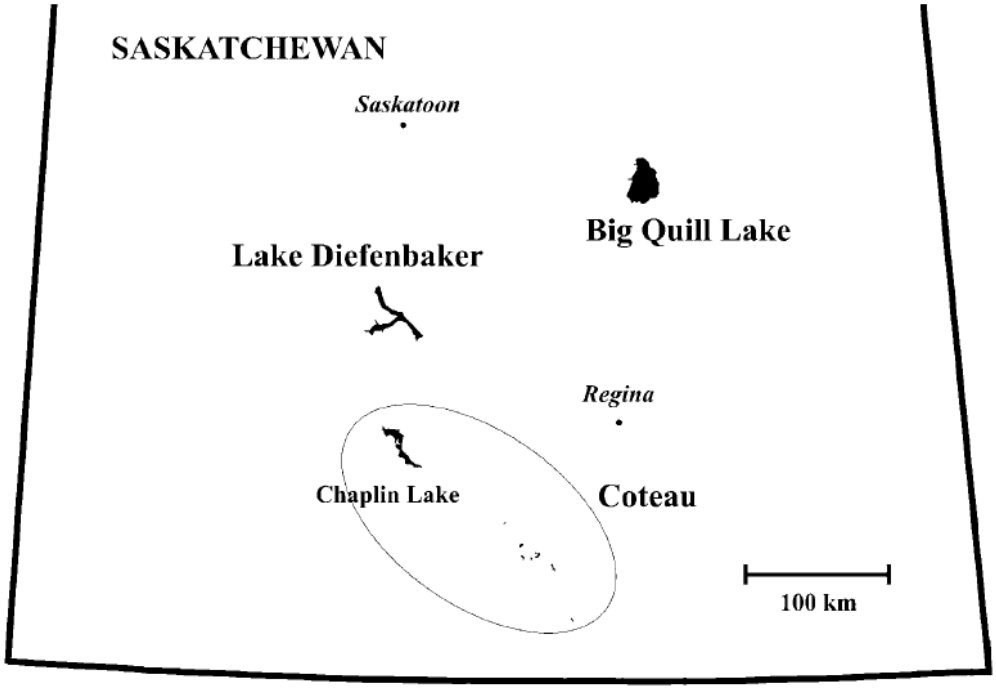


FIG. 1. Map of the three Piping Plover breeding-population study areas in Saskatchewan: Lake Diefenbaker, Big Quill Lake, and the Missouri Coteau (composed of Chaplin Lake and multiple small alkali wetlands in the Coteau).

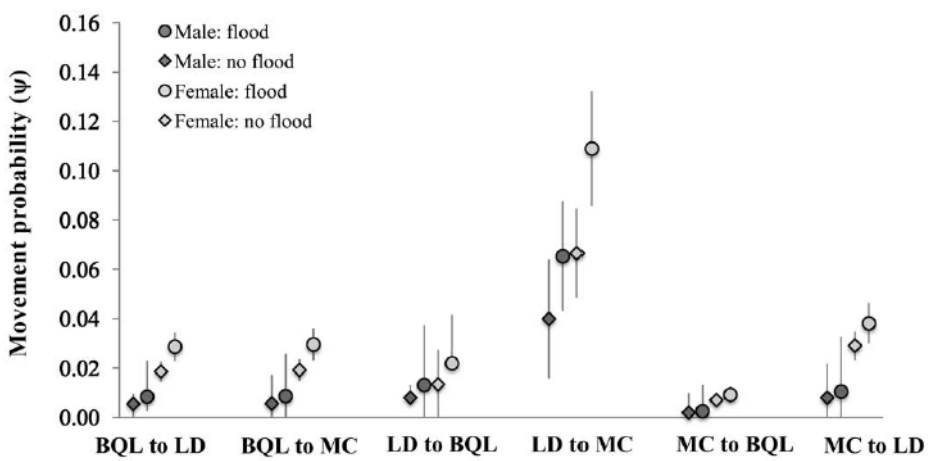


FIG. 3. Probability of interannual breeding-season movement (ψ) following flood and nonflood years among three Piping Plover breeding areas in Saskatchewan (BQL = Big Quill Lake, LD = Lake Diefenbaker, and MC = Missouri Coteau) (2002–2009). See Table 1 for movement-specific sample sizes. Diamonds represent movement following nonflood years; circles represent movement following flood years. Movement probabilities for male Piping Plovers are represented by dark gray markers and those of females by light gray. Vertical lines represent 85% confidence intervals. Estimates were generated from the top-supported model solved at the annual mean covariate value for "fledge," using a sample of 782 marked after-hatch-year Piping Plovers [S(g*t),p(g*t + sex),ψ(g. + sex + flood + fledge)].

F.8.8 Catlin 2009

Findings:

Table 5: Monte Carlo estimates from model-averaged parameter estimates and unconditional standard errors from the Barker (1997) model for piping plovers banded as adults (N = 357) on the Gavins Point Reach of the Missouri River and Lewis and Clark Lake from 2005 – 2008.

Parameter[A]	Period	Estimate[B]	Standard Error	Lower 95% Confidence Limit	Upper 95% Confidence Limit
S	2005 - 2006	0.822	0.033	0.747	0.878
	2006 - 2007	0.824	0.038	0.736	0.887
p	2005 - 2007	0.971	0.015	0.941	1.000
R	2005 - 2007	0.213	0.032	0.151	0.275
R'	2005 - 2007	0.118	0.028	0.063	0.174
F	2005 - 2006	0.999	0.010	0.980	1.000
	2006 - 2007	0.891	0.052	0.742	0.959

[A] True survival (S), resight rate on the breeding grounds (p), resight rate during the non-breeding season (R), the mortality rate of birds that were sighted during the non-breeding season (R'), and the fidelity rate (F).

[B] Estimates and standard errors for estimates that are not both year- and group-specific were derived from Monte-Carlo procedure with 10,000 iterations to combine year- and group-specific estimates.

Table 9: Monte Carlo estimates from model-averaged parameter estimates and unconditional standard errors from the Barker (1997) model for piping plovers banded as chicks (N = 685) on the Gavins Point Reach of the Missouri River and Lewis and Clark Lake from 2005 – 2008.

Parameter[A]	Period	Estimate[B]	Standard Error	Lower 95% Confidence Limit	Upper 95% Confidence Limit
$S_{\text{adult, engineered}}$	2006 – 2007	0.793	0.078	0.602	0.907
$S_{\text{adult, natural}}$	2006 – 2007	0.764	0.215	0.239	0.971
$p_{\text{chick, engineered}}$	2005 – 2007	0.777	0.074	0.601	0.890
$p_{\text{chick, natural}}$	2005 – 2007	0.531	0.120	0.305	0.745
$p_{\text{adult, all}}$	2006 – 2007	0.890	0.074	0.650	0.972
$R_{\text{chick, all}}$	2005 – 2007	0.146	0.028	0.010	0.210
$R_{\text{adult, engineered}}$	2006 – 2007	0.263	0.049	0.178	0.369
$R_{\text{adult, natural}}$	2006 – 2007	0.128	0.067	0.044	0.321
R'_{all}	2005 – 2007	0.000	0.000	0.000	0.749
$F_{\text{adult, engineered}}$	2006 – 2007	0.792	0.093	0.557	0.920
$F_{\text{adult, natural}}$	2006 – 2007	0.716	0.305	0.118	0.979
$\beta_{\text{hatch date}}$		-0.087	0.016	-0.118	-0.057
$\beta_{\text{age at banding}}$		-0.013	0.072	-0.155	0.128
$\beta_{\text{age at banding*hatch date}}$		0.005	0.003	-0.001	0.011
$\beta_{\text{natural nesting density}}$		1.158	0.583	0.017	2.300

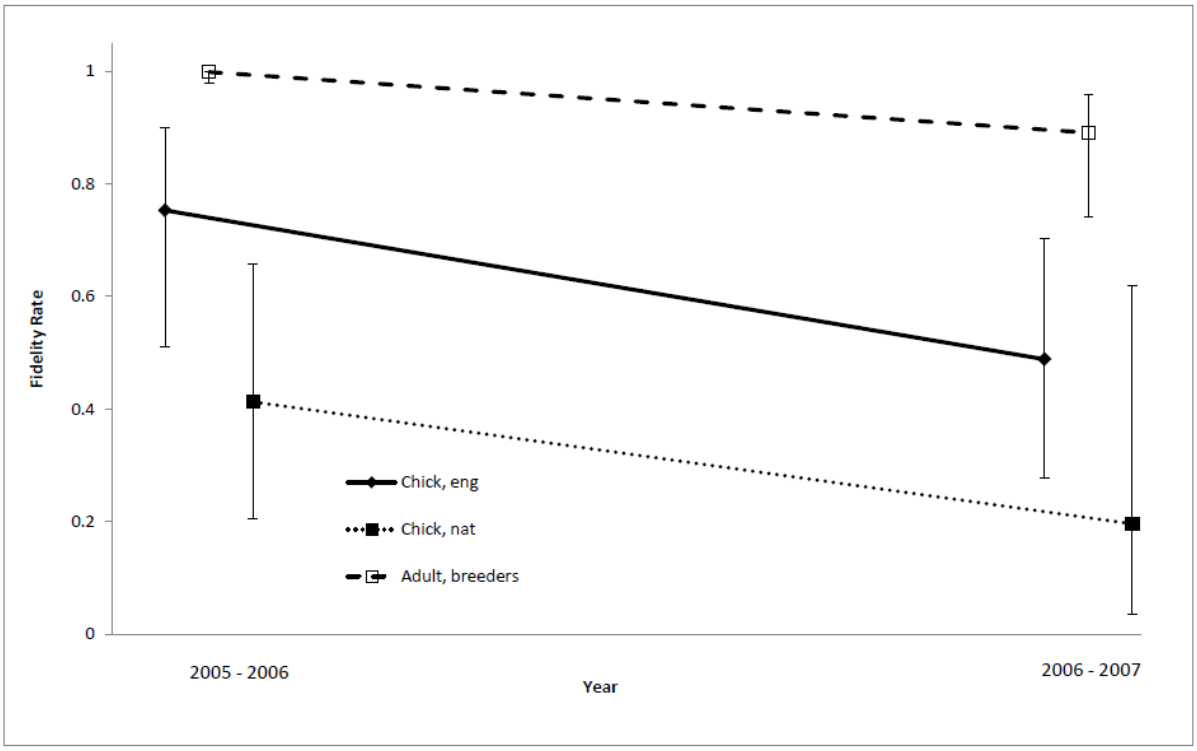


Figure 2: Site fidelity of piping plovers banded as adults and as chicks to the Gavins Point Reach of the Missouri River and Lewis and Clark Lake from 2005 – 2007. Bars represent 95% CIs.

F.9 Overwintering Habitat

F.9.1 Haig et al. 2005

Findings:

Table 1. Wintering piping plovers (PIPL) recorded and winter survey effort in 2001.

State/Country	PIPL	% of census	Sites surveyed	Km surveyed	Participants[a]	Coordinator
North Carolina	87	3.6	39	330	43	D. Allen
South Carolina	78	3.3	32	146	5	T. Murphy
Georgia	111	4.6	16	155	71	B. Winn
Florida	416	17.4	122	900	162	P. Kelly/B. Brooks
Atlantic	111	4.6	34	242	n/a[b]	
Gulf	305	12.8	88	658	n/a[b]	
Alabama	30	1.3	13	47	10	R. Clay
Mississippi	18	0.8	12	167	9	M. Woodrey
Louisiana	511	21.4	26	202	23	S. Shively
Texas	1,042	43.6	50	1,075	53	P. Glass/R. Cobb/ J. Rupert
Puerto Rico	6	0.3	5	15	3	S. Earsom
Cuba	55	2.3	29	105	8	F. Shaffer
Bahamas	35	1.5	8	n.r.[c]	n.r.[c]	none
Total	2,389	n/a[b]	352	3,142	387	

[a] Numbers reported are minimum estimates as not all observers recorded this information.
[b] n/a = Not applicable.
[c] n.r. = Not reported.

Table 2. International Piping Plover Winter Census results for 1991, 1996, and 2001, and percentage of birds counted relative to subsequent international breeding censuses.

	Piping plovers		
State/Country	1991	1996	2001
North Carolina	20	50	87
South Carolina	51	78	78
Georgia	37	124	111
Florida	551	375	416
Atlantic	70	31	111
Gulf	481	44	305
Alabama	12	31	30
Mississippi	59	27	18
Louisiana	750	398	511
Texas	1,904	1,333	1,042
Puerto Rico	0	0	6
Mexico	27	16	n.s.[a]
Cuba	11	66	55
Bahamas	29	17	35
Grand total	3,451	2,515	2,389
% of Breeding Census	62.9	42.4	40.2

[a] n.s. = not surveyed.

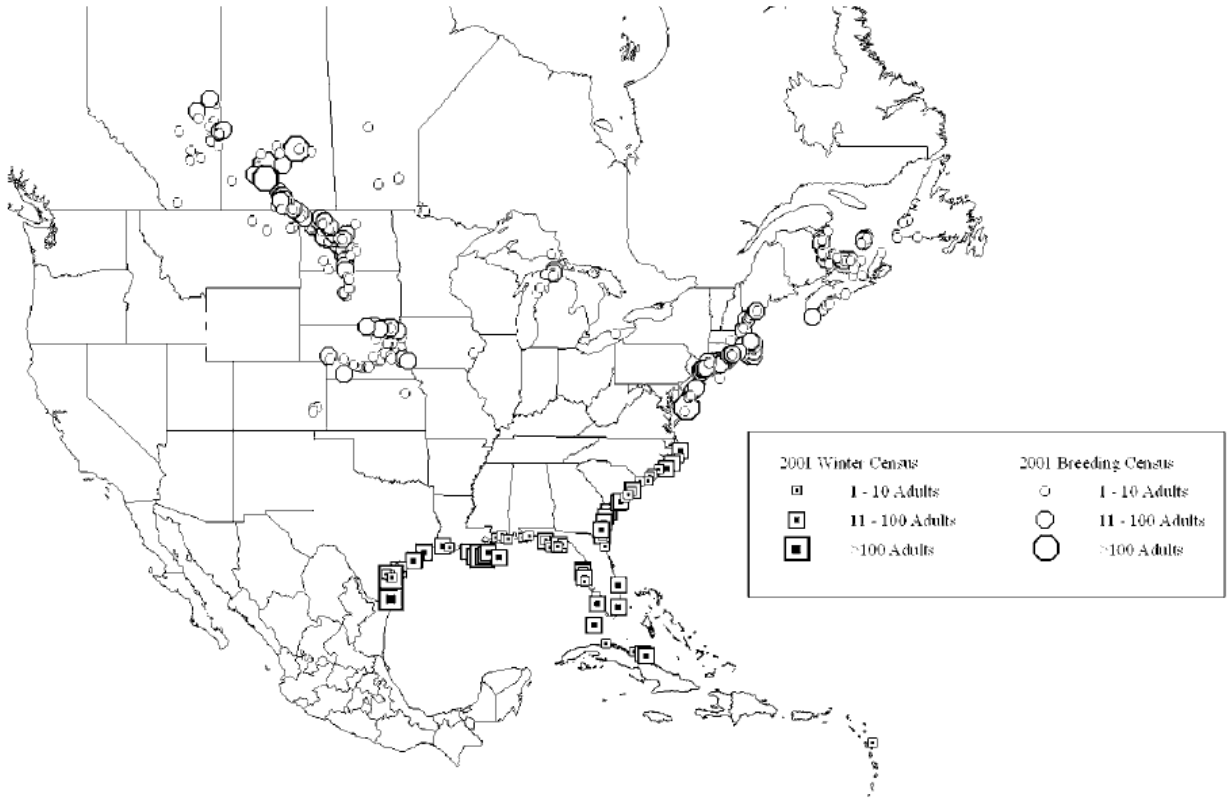


Fig. 1. Breeding and winter distribution and abundance of piping plovers in 2001. Breeding birds recorded in Canada, the United States, and St. Pierre and Miquelon, France; winter birds recorded in the southern United States, Gulf Coast of Mexico, Cuba, the Bahamas, and Caribbean.

F.9.2 Haig and Oring 1985

Findings:

TABLE 2. Winter distribution and population estimate of Piping Plovers in the United States.

State	Estimate	Year
Alabama	80	1984
Florida	135	1984
Georgia	15	1983
Louisiana	33	1984
Mississippi	51	1984
North Carolina	100	1983[a]
South Carolina	20	1984
Texas	400+	1984
Total	834+	1984

[a] J. Fussell, pers. comm.

TABLE 3. Occurrence of Piping Plovers throughout their winter range.

Location	Estimate	Year	Source
Bahamas	Rare	1981	R. Norton[a]
Barbados	Rare	1984	M. Hutt[a]
Bermuda	6–8	1983	D. Wingate[a]
Dominican Republic	Rare	1931	Wetmore and Swales 1931
Cuba	10+	1984	O. Garrido[a]
Ecuador	1 seen	1956	Marchant 1956
Haiti	2 seen	1929	U.S. National Museum[b]
Jamaica	0–2	1983	I. Goodbody, R. Sutton[a]
Mexico	20+	1984	This study
Netherlands-Antilles	1	1983	Voous 1983
Puerto Rico	Rare	1983	Raffaele 1983
United States	834+	1984	This study
Virgin Islands	0–5	1983	J. Ynetema[a]
West Indies	Rare	1982	R. Norton[a]

[a] Pers. comm.
[b] Museum skin.

FIGURE 2. Breeding and winter distribution of Piping Plovers in 1984.

F.9.3 Haig and Plissner 1993

TABLE 1. Numbers of wintering Piping Plovers and winter sites censused in 1991.

Location	Birds	Sites[1]	Plover sites[2]	Census coordinator
U.S. Atlantic	178	132	30	
North Carolina	20	32	7	Henson
South Carolina	51	27	8	Spinks/Wilkinson/Murphy
Georgia	37	14	6	Johnson
Florida	70	59	9	Kale
U.S. Gulf	3,206	197	126	
Florida	481	66	31	Runde
Alabama	12	6	1	Clay
Mississippi	59	13	7	McDearman
Louisiana	750	29	23	Martin
Texas	1,904	83	64	Eubanks
Mexico Gulf	27	18	4	Sada
Caribbean	40	11+	2	
Bahamas	29	1	1	Fettig
Turks and Caicos	0	1	0	Bradley/Fabian/Fabian
Cuba	11	2	1	Blanco/Garrido
Jamaica	0	n.a.	0	Levy
Puerto Rico	0	5	0	Lee
Cayman Islands	0	n.a.	0	Marsden
Total	3,451	358+	162	

[1] Total number of sites censused for Piping Plovers.
[2] Number of sites where Piping Plovers occurred.

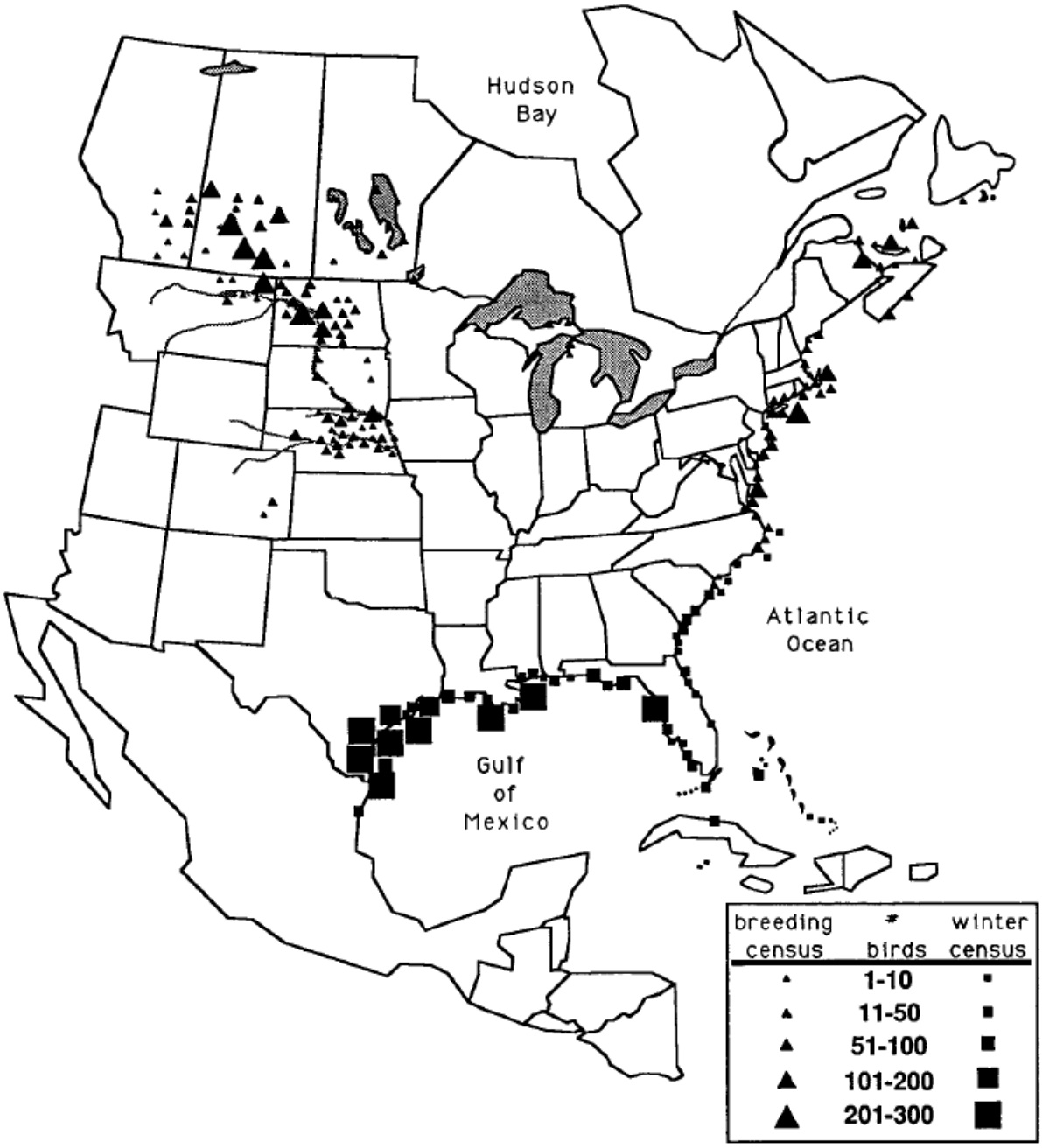


FIGURE 1. Breeding and winter distribution of Piping Plovers in 1991.

TABLE 2. Major Piping Plover winter sites in 1991.

Site	County	State	No. birds	% total census
Mustang Island	Nueces	Texas	281	8.1
Port Mansfield Pass	Willacy	Texas	247	7.1
St. Joseph Island	Aransas	Texas	210	6.0
Matagorda Peninsula South	Matagorda	Texas	154	4.4
Brazos Island/South Bay	Cameron	Texas	151	4.3
Chandeleur Islands	St. Bernard	Louisiana	131	3.8
Matagorda Island	Calhoun	Texas	116	3.3
Honeymoon Island	Pinellas	Florida	102	2.9
West Timbalier Island	Terrebonne	Louisiana	89	2.6
Breton Island	St. Bernard	Louisiana	88	2.5
Western East Island	Terrebonne	Louisiana	86	2.5
Curlew Island	St. Bernard	Louisiana	85	2.4
Redfish Bay islands	Nueces	Texas	82	2.4
South Padre Island	Cameron	Texas	82	2.4
Padre Island	Kleberg	Texas	77	2.2
Bolivar Flats	Galveston	Texas	72	2.1
Three Rooker Bar	Pinellas	Florida	59	1.7
Matagorda Peninsula North	Matagorda	Texas	53	1.5
Buena Vista Ranch	Cameron	Texas	46	1.3
Anna Maria Island	Manatee	Florida	45	1.3
Last Island	Terrebonne	Louisiana	43	1.2

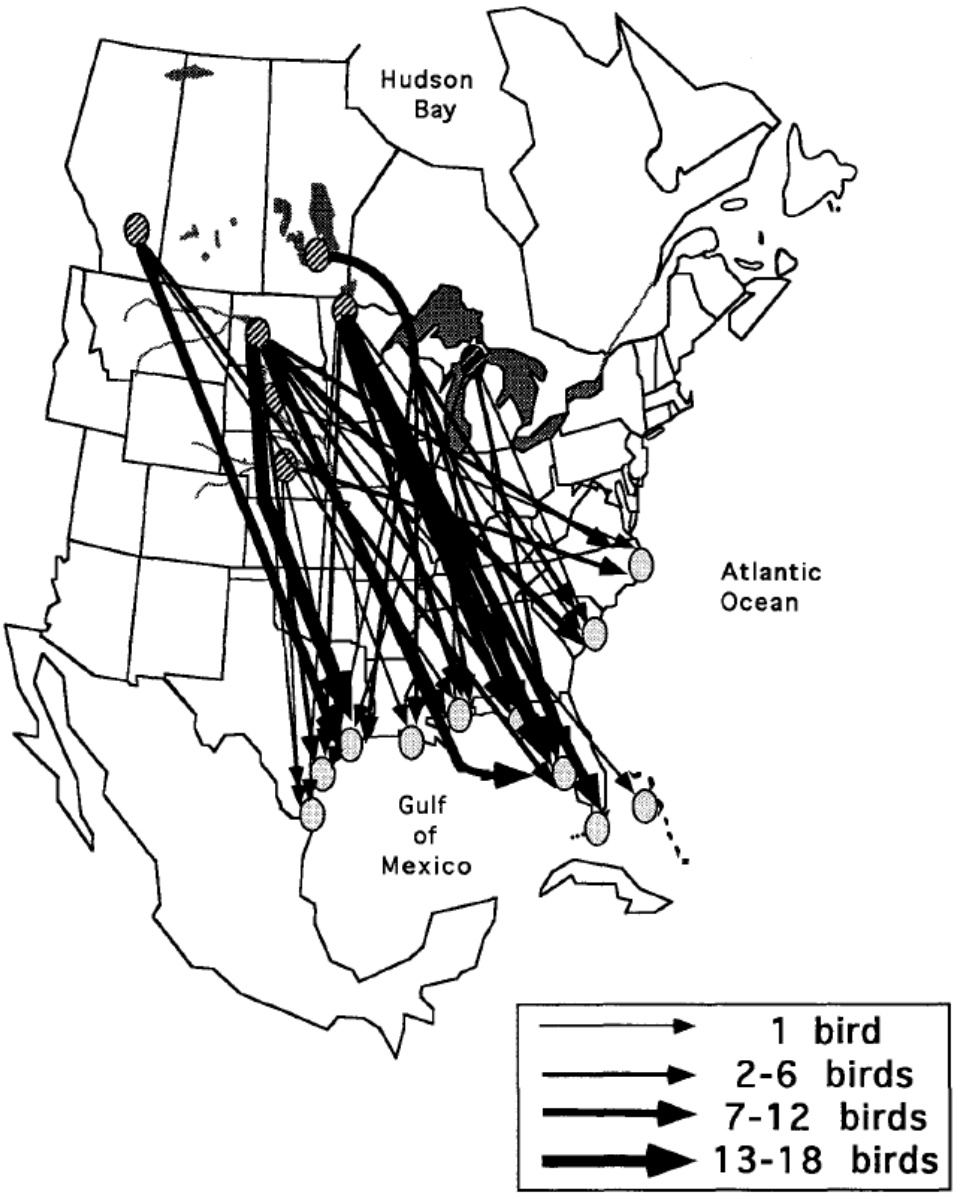


FIGURE 2. Winter sightings of Piping Plovers banded on the Great Lakes, Northern Great Plains, or Prairies during the breeding season (1983–1991).

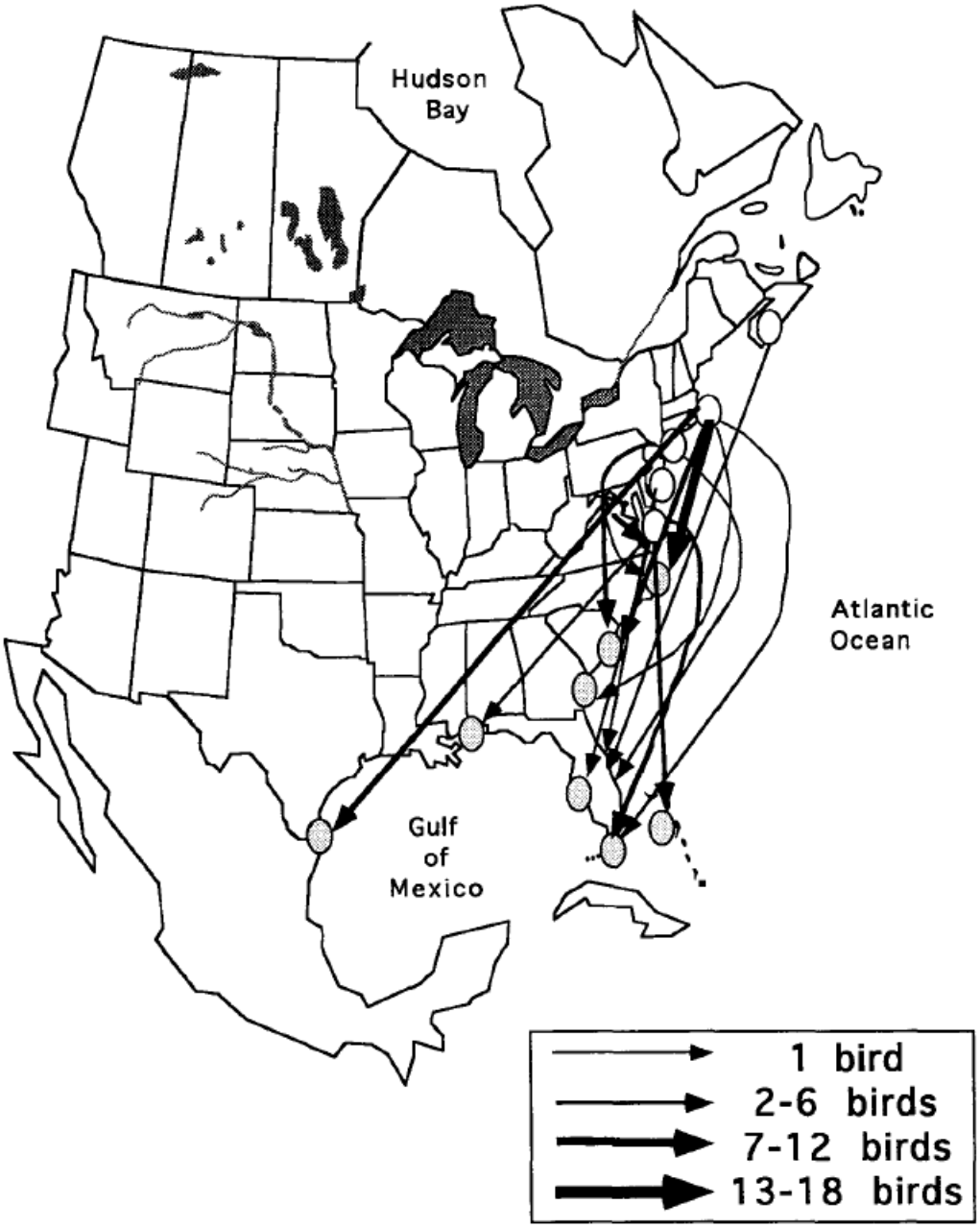


FIGURE 3. Winter sightings of Piping Plovers banded on the Atlantic Coast during the breeding season (1983–1991).

TABLE 4. Major Piping Plover breeding sites in 1991.

Site	County/area or river system	m	State/province	No. birds	% regional total	% total breeding pop.
Northern Great Plains/Prairie sites						
Lake Diefenbaker	S. Saskatchewan R.		Saskatchewan	276	8.0	5.0
Gavins Pt.–Sioux City	Missouri River		South Dakota/Nebraska	165	4.8	3.0
J. E. Williams Preserve	McLean Co.		North Dakota	162	4.7	3.0
Lake Sakakawea	Missouri River		North Dakota	162	4.7	3.0
Niobrara River	Niobrara River		Nebraska	152	4.4	2.8
Big Quill Lake	Big Quill Lake		Saskatchewan	151	4.4	2.8
L. Sakakawea–L. Oahe	Missouri River		North Dakota	124	3.6	2.3
Chaplin Lake	Missouri Coteau		Saskatchewan	113	3.3	2.1
Manitou Lake	North West		Saskatchewan	111	3.2	2.0
Platte River sand pits	Platte River		Nebraska	107	3.1	2.0
Lake Oahe	Missouri River		North Dakota/South Dakota	101	2.9	1.8
Lostwood NWR	Missouri Coteau		North Dakota	78	2.2	1.4
Medicine Lake	Sheridan Co.		Montana	65	1.9	1.2
Lake McConnaughy	N. Platte River		Nebraska	64	1.8	1.2
Ft. Randall–Gavins Pt.	Missouri River		South Dakota/Nebraska	57	1.6	1.0
Atlantic Coast sites						
Chincoteague NWR	Accomack Co.		Virginia	87	4.4	1.6
North Metomkin Island	Accomack Co.		Virginia	72	3.6	1.3
Breezy Point	Queens Co.		New York	49	2.5	0.9
Forsythe NWR	Ocean Co.		New Jersey	44	2.2	0.8
Little Beach Island	Atlantic Co.		New Jersey	43	2.2	0.8
Gateway NRA	Monmouth Co.		New Jersey	39	2.0	0.7
Assateague Island NS	Worcester Co.		Maryland	35	1.8	0.6
Brigantine Jetty	Atlantic Co.		New Jersey	25	1.3	0.5
Cobb Island	Northampton Co.		Virginia	24	1.2	0.4
Crane Beach	Essex Co.		Massachusetts	24	1.2	0.4
Orient Point SP	Suffolk Co.		New York	20	1.0	0.4
Great Lakes sites						
Grand Marais Inner	Alger Co.		Michigan	8	23.5	0.1
Cross Village North	Emmet Co.		Michigan	6	17.6	0.1
Point Island	Emmet Co.		Michigan	4	11.8	<0.1
Vermillion SP	Chippewa Co.		Michigan	3	8.8	<0.1

F.9.4 Ledee et al. 2008

Findings:

Table 1. *Results from simple linear regression. From the set of 11 parameters, the five parameters in bold were found to be significant* predictors of piping plover abundance on the U.S. Gulf of Mexico coast.*

Landscape Measure	Adjusted *R*-squared	*P* Value	df
Landform category	**0.39**	**<0.01**	30
Urban area	**0.18**	**0.01**	30
Total area	**0.17**	**0.01**	30
Intertidal area	**0.16**	**0.01**	30
Roads	**0.10**	**<0.05**	30
Boat launches	0.09	0.10	21
Access points	0.08	0.11	21
Lagoon	0.05	0.17	30
Lagoon perimeter	0.03	0.17	30
Beach	−0.02	0.47	30
River	−0.03	0.62	30

* $p < 0.05$.

F.9.5 Gratto-Trevor et al. 2012

Findings:

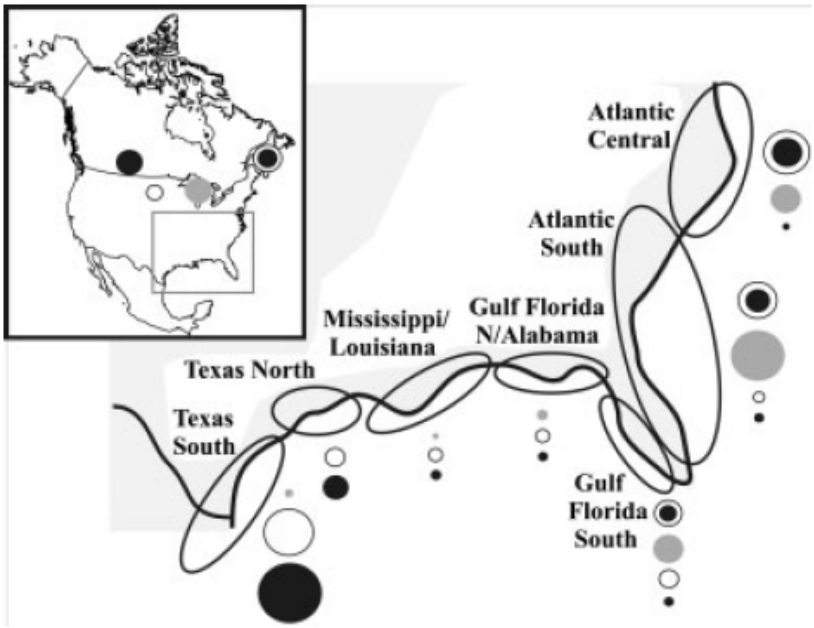


Figure 1. Four breeding locations (inset) included in our study of wintering piping plovers in North America, 1998–2008, including eastern Canada (white circle with a large black spot in the center), Great Lakes (gray circle), Great Plains (white circle), and Prairie Canada (black circle). The North American wintering area is expanded to the right, divided into different wintering regions. The size of the adjacent circles relative to others represents the percentage of individual piping plovers from a specific breeding area reported in each wintering region up to December 2008.

Table 3. Percentage of marked individual piping plovers from each breeding population observed at specific locations along the eastern coast of the United States in the winters of 2006–2007 and 2008–2009.

Winter	Location	No. banded individuals seen	Percentage from each breeding area[a]			
			Eastern Canada	Great Lakes	Great Plains	Prairie Canada
2006–2007	South Carolina	31	14	77	7	2
	Louisiana	19	0	22	45	33
	Texas North	9	0	0	32	68
	Texas South	42	0	0	26	74
2008–2009[b]	Florida Gulf South and Alabama	34	—	62	24	13
	Mississippi and Louisiana	29	—	30	36	34
	Texas North	42	—	0	30	70
	Texas South	99	—	0	36	64

[a] Corrected for the number of banded birds estimated alive in each breeding population.
[b] By 2008 few, if any, eastern Canada plovers were individually recognizable due to removal of color bands and abrasion of codes on remaining bands.

F.9.6 Stucker et al. 2010

Findings:

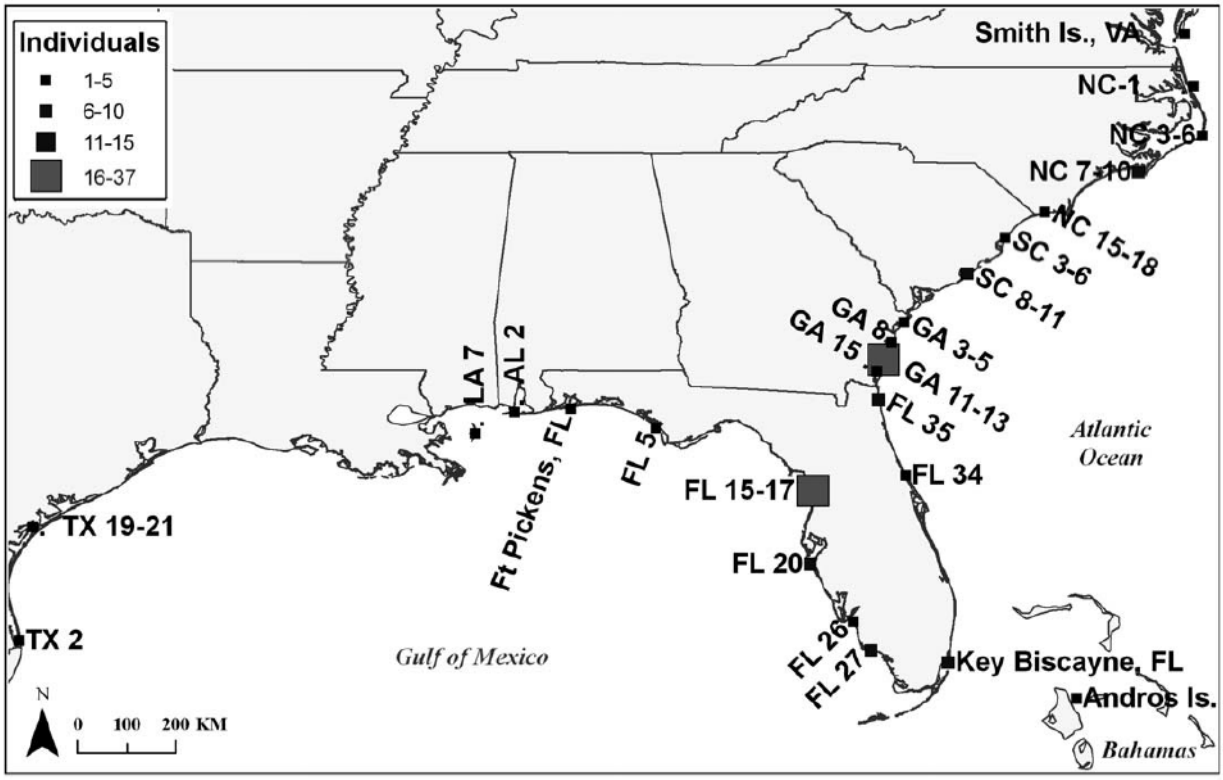


Figure 1. Counts of Great Lakes Piping Plover individuals at coastal wintering locations November-February, 1995-2005. Sites are labeled by name or Critical Habitat Unit (State - #).

F.10 Population Trends through Time

F.10.1 Calvert et al. 2006

Observed Trend:

Fig. 1. Location of beaches used by eastern Canada Piping Plovers breeding in the Gulf of St. Lawrence (squares) and South Nova Scotia (circles) from 1998 to 2003, with associated population abundance trends between 1991 and 2005 as indicated by the standardized annual population survey (black) and international census (gray) data (see Amirault 2005). Recovery goals for each population segment are also noted (dashed lines; Amirault 2006). Regions within the Gulf population segment are: AC (Acadian peninsula, northern New Brunswick), NBNS (southern New Brunswick and northern Nova Scotia), NF (Newfoundland), PEI (Prince Edward Island), and QC (Magdalen Islands, Quebec).

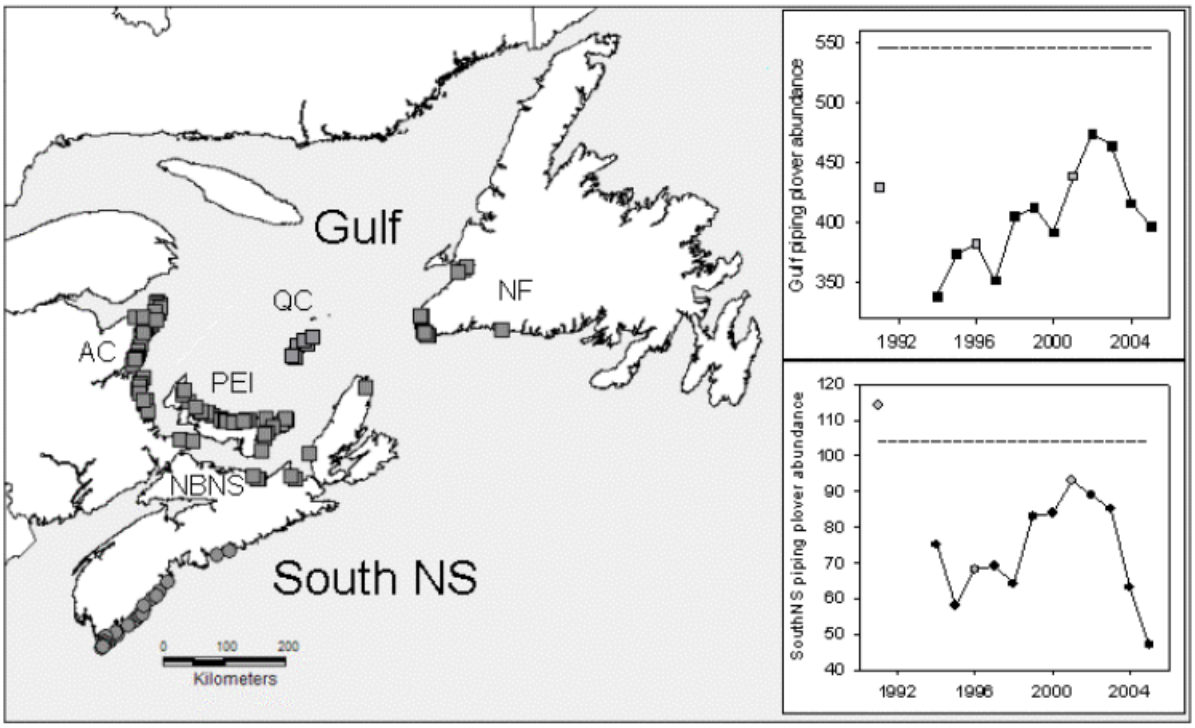


Table 1. Component parameters for the eastern Canada Piping Plover projection matrix model, and estimates for each breeding population segment (South NS: southern Nova Scotia; Gulf: Gulf of St. Lawrence); see text for details of parameter estimation.

Parameter	*Notation*	*Definition*	*South NS*	*Gulf*
Adult survival	Φ_A	SY$^+$ bird survival 1 yr after census	0.7324	0.7331
Juvenile survival (from hatch)	Φ_J	HY bird survival 1 yr after census	0.3279	0.2395
Fledging success	f	Probability hatchling survives to fledge	0.6171	0.7014
Juvenile survival (post-fledge)	$\Phi_J^w = \Phi_J/f$	HY bird survival from fledge to 1 yr old	0.5314	0.3415
Second-year recruitment	y_S	Probability that SY bird builds nest	0.8095	0.8504
Third-year recruitment	y_T	Probability that TY bird builds nest	0.9910	0.9823
Number of eggs laid	E	Mean number eggs laid per nest	3.8065	3.9389
Hatching success	h	Probability that an egg hatches	0.4603	0.5120

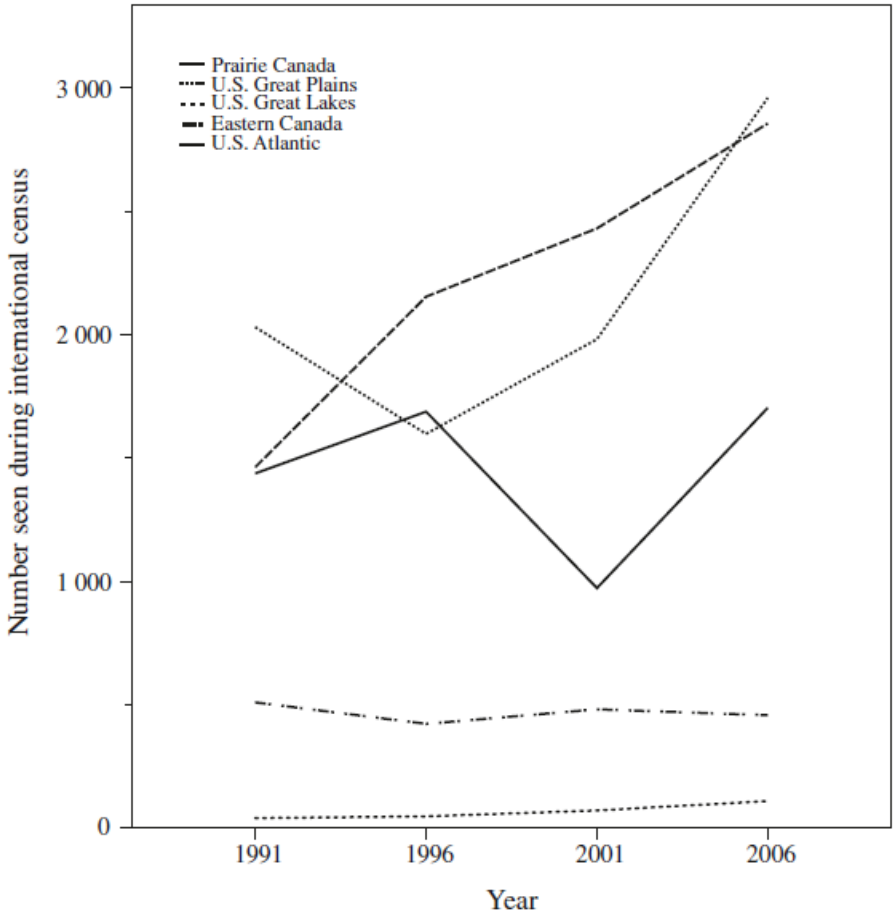
Table 3. Deterministic growth rate estimates, stable age distribution, age-specific reproductive values, sensitivities, and elasticities for southern Nova Scotia and Gulf of St Lawrence population segments of Piping Plovers in eastern Canada, 1998–2003.

	South NS (λ_d = 1.0043)		**Gulf (λ_d = 0.9651)**	
Parameter	*Sensitivity*	*Elasticity*	*Sensitivity*	*Elasticity*
Φ_A	1.0140	0.7398	1.0090	0.7666
Φ_J	0.7970	0.2602	0.9404	0.2334
f	0.4235	0.2602	0.3212	0.2334
$\Phi_J^* = \Phi_J/f$	0.4918	0.2602	0.6596	0.2334
y_S	0.0748	0.0603	0.0564	0.0497
y_T	0.0545	0.0538	0.0428	0.0436
E	0.0687	0.2602	0.0572	0.2334
h	0.5678	0.2602	0.4400	0.2334
Age Class	*Stable Age Distribution*	*Reproductive Contribution*	*Stable Age Distribution*	*Reproductive Contribution*
Hatch-year (HY)	0.4533	0.1750	0.4921	0.1359
Second-year (SY)	0.1480	0.4120	0.1221	0.4311
Third-year or older (TY^+)	0.3986	0.4130	0.3858	0.4330

F.10.2 Gratto-Trevor and Abbott 2011

Observed Trend:

Fig. 1. Regional counts from the International Piping Plover (*Charadrius melodus*) Breeding Census during 1991, 1996, 2001, and 2006 (Haig and Plissner 1993; Plissner and Haig 2000*a*; Haig et al. 2005; Elliott-Smith et al. 2009).



F.10.3 Haig and Plissner 1993

Observed Trend:

TABLE 5. Changes in numbers of Piping Plovers at specific breeding areas.

Location	Date 1st est.	1st est.[1]	Date 2nd est.	2nd est.[1]	1991[2] census	% Change 1st est. →1991	% Change 2nd est. →1991
Atlantic Coast							
Newfoundland	1968	30	1984	4	7	−72	+75
Cadden Beach, Nova Scotia	1976	56	1983	28	20	−64	−29
Maine	1976	48	1982	12	38	−21	+217
Rhode Island	1945	80	1983	20	47	−41	+135
Connecticut	1980	40	1983	34	67	+68	+97
Long Island, New York	1939	1,000	1983	200	338	−66	+69
New Jersey	1980	118	1983	64	280	+137	+338
Delaware	1978	80	1984	18	10	−88	−44
Maryland	1972	85	1984	25	35	−59	+40
Great Lakes							
Michigan	1979	77	1982	14	39	−49	+179
Wisconsin	1900	140	1983	6	1	−99	−83
Northern Great Plains/Prairie							
Big Quill Lake, Saskatchewan	1978	210	1984	186	151	−28	−19
Chain Lakes, Alberta	1976	50	n.a.	n.a.	9	−72	n.a.
Lake Manitoba, Manitoba	1980	27	1984	9	3	−89	−67
Lake of the Woods, Minnesota	1982	44	1986	32	13	−70	−59
Niobrara River, Nebraska	1981	92	1985	100	110	+20	+10

[1] Sources are listed in Haig and Oring (1985).
[2] Sources are listed in Table 2.

F.10.4 Hecht and Melvin 2009a

Observed Trend:

Table 1. Number of nonfederal cooperators, total number of sites occupied by United States Atlantic Coast breeding piping plovers, number of sites with ≤5 pairs, and percent breeding pairs at sites with ≤5 pairs, 2002.

State	No. nonfederal cooperators[a]			No. occupied breeding sites		% breeding pairs at sites with ≤5 pairs
	State agency	Local government	Private	Total	With ≤5 pairs	
ME	2	2	7	20	17	71
NH	1	0	0	2	2	100
MA	2	10	6	109	84	28
RI	1	1	2	13	7	14
CT	1	0	1	8	7	77
NY	2	7	4	84	64	35
NJ	2	3	1	28	18	29
DE	3	0	0	2	2	100
MD	1	0	0	1	0	0
VA	2	0	2	7	1	1
NC	2	1	2	7	6	35
All states	19	24	25	281	208	30

[a] State and local government agencies, nongovernmental organizations, or private landowners that 1) paid full- or part-time staff to monitor or protect piping plovers, or 2) contributed funds, equipment, or supplies to on-site protection efforts.

F.10.5 Hecht and Melvin 2009b

Observed Trend:

Table 1. Distribution, population growth (λ) 1989-2006, and productivity (chicks fledged per pair), by state and recovery units, of breeding pairs of Atlantic Coast Piping Plovers.

RECOVERY UNIT/State	Number (%) breeding pairs 1989	Number (%) breeding pairs 2006	λ[a] (1989-2006)	Number (%) breeding pairs Overall (range)
EASTERN CANADA	233 (24.3)	256 (14.6)	1.10	1.61 (0.69-2.10)
Maine	16 (1.7)	40 (2.3)	2.50	1.65 (0.55-2.50)
New Hampshire	0 (0.0)	3 (0.2)	—	1.40 (0.00-2.67)
Massachusetts	137 (14.3)	482 (27.6)	3.52	1.41 (1.00-2.03)
Rhode Island	19 (2.0)	72 (4.1)	3.79	1.41 (0.77-2.00)
Connecticut	34 (3.6)	37 (2.1)	1.09	1.48 (0.38-2.14)
NEW ENGLAND	206 (21.5)	634 (36.2)	3.08	1.44 (1.04-1.91)
New York	191 (20.0)	422 (24.2)	2.21	1.29 (0.80-1.62)
New Jersey	128 (13.4)	116 (6.6)	0.91	1.00 (0.39-1.40)
NY-NJ REGION	319 (33.3)	538 (30.8)	1.69	1.18 (0.88-1.49)
Delaware	3 (0.3)	9 (0.5)	3.00	1.48 (0.50-2.50)
Maryland	20 (2.1)	64 (3.7)	3.20	1.33 (0.41-2.41)
Virginia	121 (12.6)	202 (11.6)	1.67	1.35 (0.59-2.23)
North Carolina	55 (5.7)	46 (2.6)	0.84	0.54 (0.07-0.92)
SOUTHERN REGION	199 (20.8)	321 (18.4)	1.61	1.19 (0.62-1.95)
U.S.TOTAL	724 (75.7)	1,493 (85.4)	2.06	1.31 (1.06-1.56)
ATLANTIC COAST TOTAL	957 (100.0)	1,749 (100.0)	1.83	1.35 (1.16-1.54)

[a]$\lambda = N_{t+1}/N_t$.
[b]productivity = chicks fledged per pair.

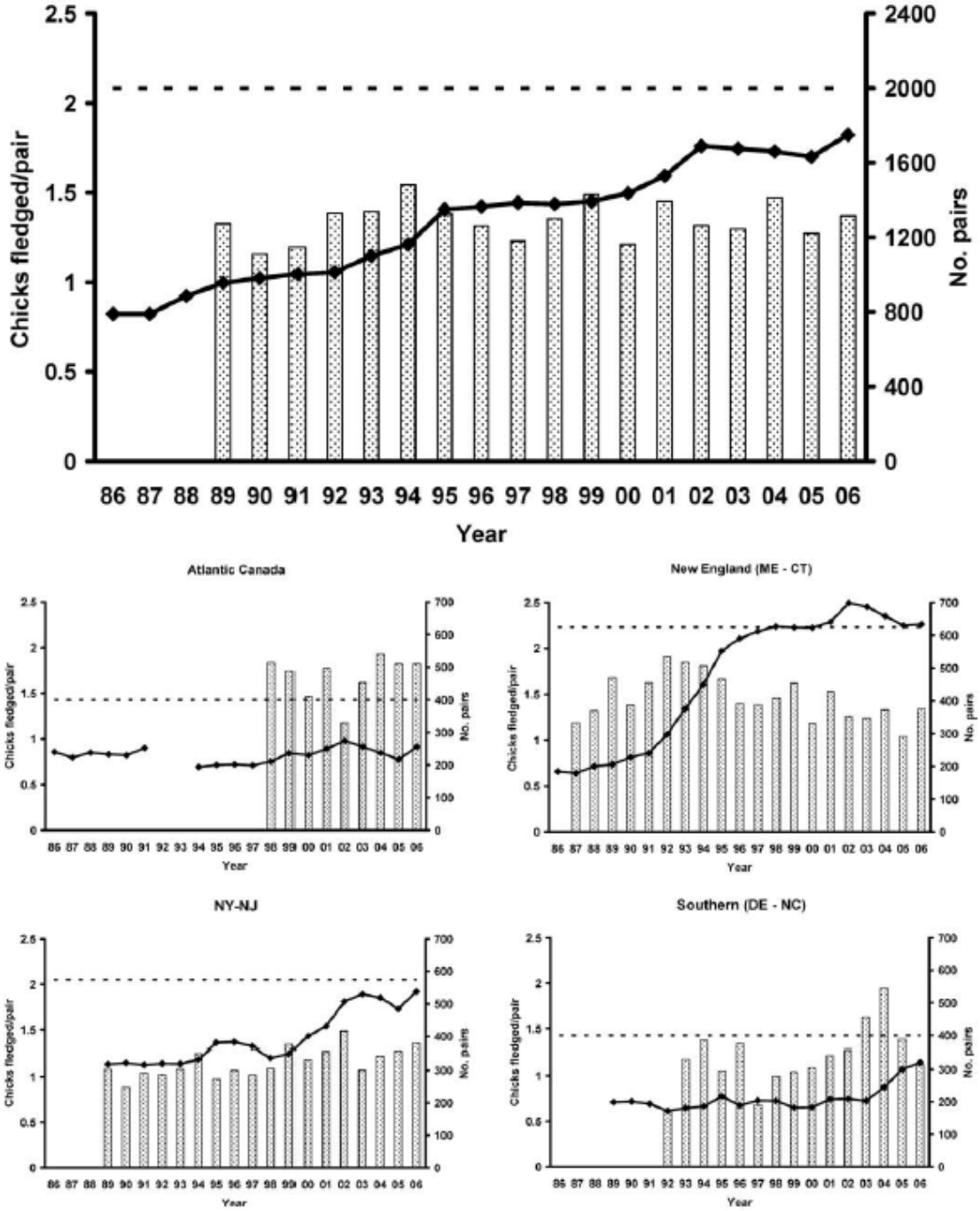


Figure 1. Trends in abundance (breeding pairs) and productivity (chicks fledged per pair) for the Atlantic Coast population of Piping Plovers and individual recovery units established in the U.S. recovery plan for the Atlantic Coast population of Piping Plovers (U.S. Fish and Wildlife Service 1996). Abundance data are indicated by lines connecting points, productivity data are indicated by bars. Dashed lines indicate abundance objectives established for the Atlantic Coast population as a whole and for individual recovery units in the U.S. recovery plan (U.S. Fish and Wildlife Service 1996). Abundance data are not plotted for Eastern Canada in 1992 and 1993 and for the New York-New Jersey and Southern recovery units in 1986-1988 because of incomplete census efforts in those years. Productivity data are not plotted in 1986-1997 for Eastern Canada, 1986-87 for New England, 1986-88 for New York-New Jersey, and 1986-91 for the Southern recovery unit because percent of pairs for which productivity data were reported was ≤60%.

F.10.6 McGowan et al. 2011

Observed Trend:

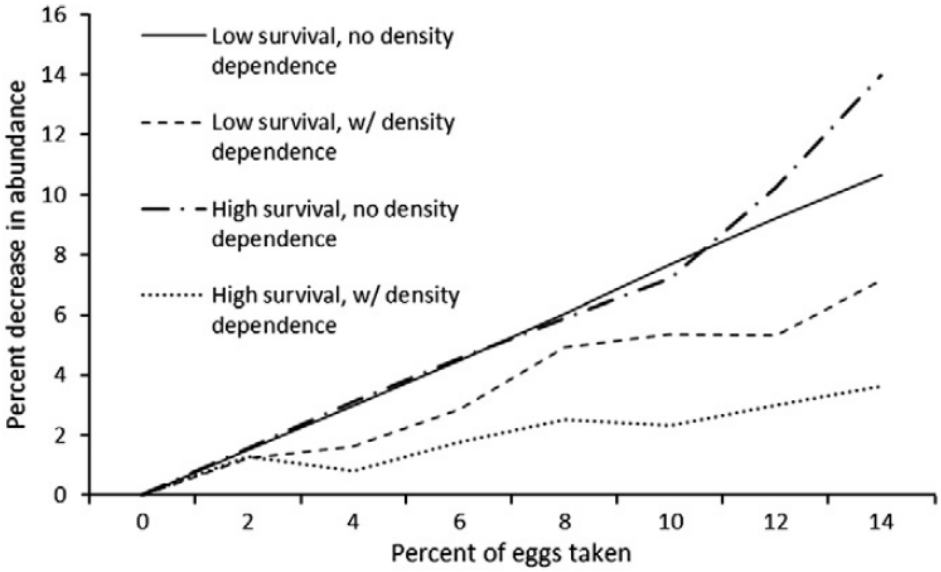


Fig. 2. The simulated percent decline in median abundance at 30 years, relative to the no-take scenario, under 0% (baseline), 2%, 4%, 6%, 8%, 10%, 12%, and 14% of eggs incidentally taken for piping plovers in the Great Plains, with low or high adult survival, and with or without density-dependent juvenile survival.

F.10.7 Plissner and Haig 2000b

Observed Trend:

Table 3
Results of Great Lakes/Great Plains/Prairie baseline model

	Population					
	Metapopulation	Great Lakes	Manitoba and Lake of the Woods	N. Missouri River and Coteau	Nebraska rivers	Colorado
Baseline model						
Probability of survival for 100 years (SE)	0.002 (0.002)	0	0	0.002 (0.002)	0	0
Mean final population size [a] (SE)	18 (0)	0	0	18(0)	0	0
Mean years to first extinction [b] (SE)	55.39 (0.56)	30.64 (0.75)	32.27 (0.44)	51.15 (0.56)	46.71 (0.52)	22.05 (0.39
Population growth rate (r)	−0.136	−0.103	−0.115	−0.142	−0.113	−0.091

[a] For populations persisting 100 years.
[b] For populations going extinct.

Table 4
Results of Atlantic Coast piping plover metapopulation models

	Population				
	Atlantic metapopulation	Atlantic Canada	New England	Mid-Atlantic	Southern Region
Baseline model					
Probability of survival for 100 years	1.000	1.000	1.000	1.000	0.996
Mean final extant population size ±SE)	2172 ± 6.19	579 ± 8.28	1159 ± 7.81	367 ± 9.11	68 ± 2.25
Population growth rate (r)	0.073	0.032	0.126	−0.006	−0.015
Fecundity = 1.25 fledged/year					
Probability of survival for 100 years	1.000	1.000	1.000	1.000	1.000
Mean final extant population size (± SE)	2571 ± 29.07	547 ± 8.85	751 ± 14.68	741 ± 13.89	532 ± 9.55
Population growth rate (r)	0.023	0.022	0.010	0.014	0.021

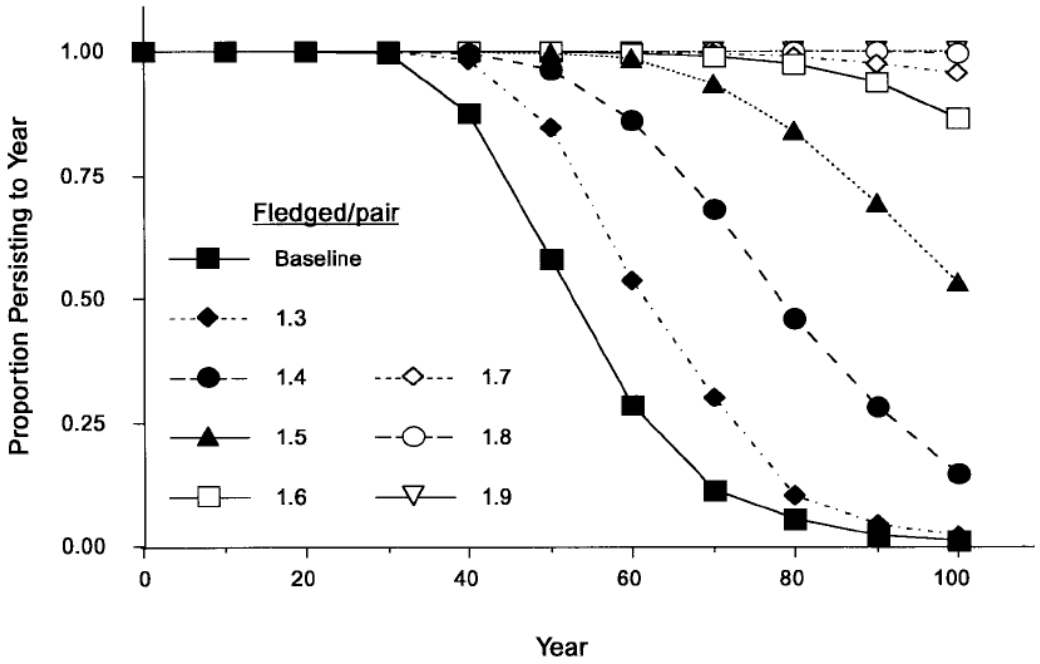


Fig. 2. Probability of Great Lakes/Great Plains piping plover metapopulation persistence relative to mean number of offspring fledged per pair.

Table 5
Persistence and size of Great Lakes/Great Plains populations at reproductive success (RS) = 1.7 and RS = 2.0

Population	RS = 1.7 fledged per pair		RS = 2.0 fledged per pair	
	Probability of survival for 100 years (SE)	Mean final population size[a] (SE)	Probability of survival for 100 years (SE)	Mean final population size[a] (SE)
Metapopulation	0.958 (0.009)	694.84 (40.35)	0.998 (0.002)	2398.77 (72.91)
Great Lakes	0.328 (0.021)	84.73 (7.02)	0.798 (0.018)	181.01 (5.00)
Manitoba and Lake of the Woods	0.870 (0.015)	104.08 (3.95)	0.996 (0.003)	185.10 (3.26)
Missouri River/Coteau	0.904 (0.013)	417.23 (34.69)	0.998 (0.002)	1671.19 (66.91)
Platte River, NE	0.884 (0.014)	171.88 (7.32)	0.998 (0.002)	366.43 (6.60)
Colorado	0.758 (0.019)	23.44 (0.64)	0.982 (0.006)	32.06 (0.44)

[a] For populations persisting 100 years.

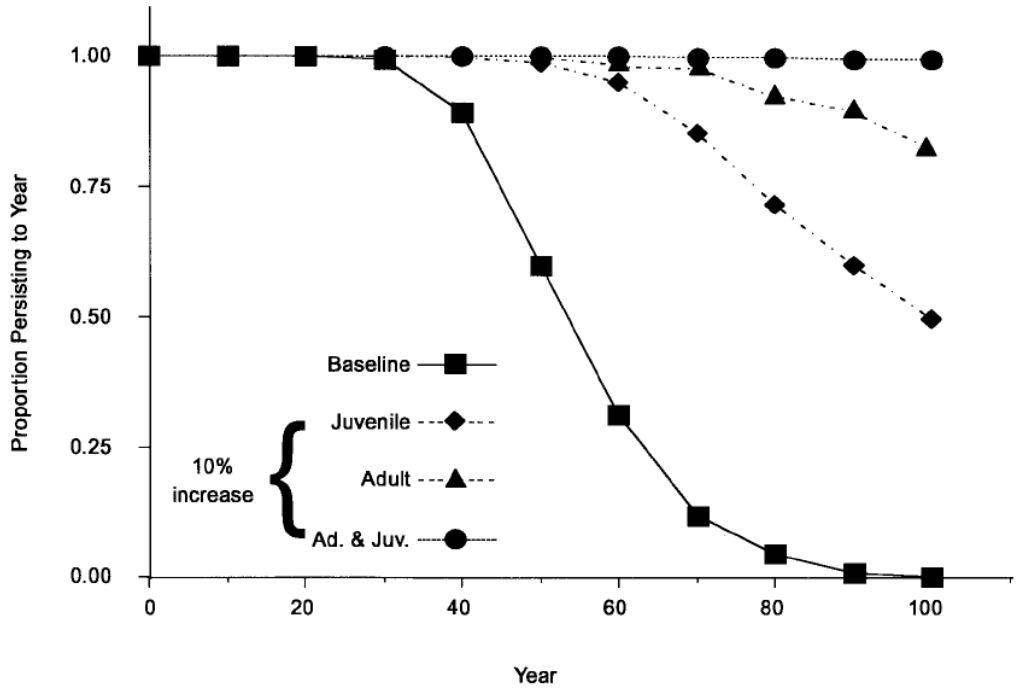


Fig. 3. Probability of Great Lakes/Great Plains piping plover metapopulation persistence as an effect of a 10% increase over baseline adults and/or juvenile survivorship.

F.10.8 Haig et al. 2005

Observed Trend:

Table 3. Piping plovers recorded and census effort for the 2001 international piping plover breeding census.

State/Province	Adults	% of census	% of region	Pairs	Sites[a]	Km[b]	Observers[b]	Coordinator(s)
Prairie Canada and U.S. Northern Great Plains	**2,953**	**49.7**	**n/ac**	**1,291**	**958**	**6,235**	**414**	
Prairie Canada	972	16.3	32.9	392	424	3,506	240	
Alberta	150	2.5	5.1	72	115	905	66	D. Prescott
Saskatchewan	805	13.5	27.3	313	282	2,552	165	L. Dunlop
Manitoba	16	0.3	0.5	7	23	37	7	K. De Smet
Ontario (Lake of Woods)	1	0.0	0.0	0	4	12	2	L. Heyens/S.Jones
U.S. Northern Great Plains	1,981	33.3	67.1	899	534	2,729	174	
Minnesota (Lake of Woods)	7	0.1	0.2	3	4	10	2	K. Haws
Montana	137	2.3	4.6	57	64	431	21	L. Hanebury
Missouri River	7[d]	n/a[c]	n/a[c]	3[d]	3[d]	1[d]	n/a[c]	
North Dakota	1,112	18.7	37.7	522	246	902	51	K. Kreil
Missouri River	643[d]	n/a[c]	n/a[c]	298[d]	116[d]	36[d]	n/a[c]	
South Dakota	390	6.6	13.2	172	77	178	22	N. McPhillips
Missouri River	390[d]	n/a[c]	n/a[c]	172[d]	67[d]	136[d]	n/a[c]	
Nebraska	308	5.2	10.4	133	117	1,081	44	J. Dinan
Missouri River	8[d]	n/a[c]	n/a[c]	3[d]	3[d]	1[d]	n/a[c]	
Missouri River: MT, ND, SD	1,048[d]	17.6[d]	35.5[d]	476[d]	189[d]	174[d]	28	C. Kruse/G. Pavelka
Iowa	11	0.2	0.4	5	2	2	2	D. Howell
Kansas	3	0.1	0.1	2	1	2	3	D. Mulhern
Colorado	13	0.2	0.4	5	23	123	1	J.Yost/D. Nelson
Great Lakes	**72**	**1.2**	**n/a[c]**	**29**	**108**	**361**	**71**	
Canada (Ontario)	1	0.0	1.4	0	33	114	18	L. Heyens/S.Jones
United States	71	1.2	98.6	29	75	247	53	J. Dingledine
Michigan	65	1.1	90.3	27	58	166	24	F. Cuthbert
Wisconsin	6	0.1	8.3	2	8	29	16	J. Trick
IL/IN/OH/PA/NY	0	0.0	0.0	0	9	52	13	J. Dingledine
Atlantic	**2,920**	**49.1**	**n/a[c]**	**1,427**	**826**	**2,098**	**492**	
St. Pierre and Miquelon France	9	0.2	0.3	4	4	20	3	R. Etcheberry
Eastern Canada	481	8.1	16.5	240	350	936	206	
Newfoundland	39	0.7	1.3	23	39	73	26	J. Brazil
Quebec	70	1.2	2.4	35	40	213	39	F. Shaffer
Prince Edward Island	112	1.9	3.8	54	87	188	59	J. Waddell
New Brunswick	167	2.8	5.7	83	66	303	39	D. Amirault
Nova Scotia	93	1.6	3.2	45	118	159	43	P. Mills/A. Bond
U.S. Atlantic	2,430	40.9	83.2	1,183	472	1,142	283	
Maine	96	1.6	3.3	48	32	48	19	J. Jones
New Hampshire	14	0.2	0.5	7	2	2	5	C. Dudley
Massachusetts	962	16.2	32.9	481	161	n.r.e	105	S. Melvin
Rhode Island	93	1.6	3.2	46	19	28	7	C. Raithel
Connecticut	45	0.8	1.5	23	27	25	6	J. Victoria
New York	624	10.5	21.4	309	119	311	29	M. Gibbons
New Jersey	228	3.8	7.8	109	44	127	36	D. Jenkins/T.Pover
Delaware	10	0.2	0.3	5	13	34	2	A. Doolittle
Maryland	112	1.9	3.8	28	3	46	10	D. Brinker
Virginia	198	3.3	6.8	106	23	194	26	R. Boettcher
North Carolina	48	0.8	1.6	21	28	327	37	D. Allen
South Carolina	0	0.0	0.0	0	1	n.r.[e]	1	T. Murphy
Totals								
United States	4,482	75.4	n/ac	2,111	1,081	4,118	510	
Canada	1,454	24.5	n/ac	632	807	4,556	464	
France	9	0.2	n/ac	4	4	20	3	
Grand total	5,945	n/ac	n/ac	2,747	1,892	8,694	977	

[a] 52 additional sites were not officially ground surveyed due to lack of habitat and/or access.
[b] Numbers reported serve as minimum estimates. Not all observers included this information.
[c] n/a = not applicable.
[d] Subtotals for reference only. Missouri River results by state are included in state results.
[e] n.r. = not reported.

Table 4. Adult piping plovers recorded in the 1991, 1996, and 2001 international piping plover breeding censuses.

State/Province	Adults 1991	Adults 1996	Adults 2001	% change (1991–2001)	% change (1996–2001)
Northern Great Plains/Prairies	3,469	3,286	2,953	–14.9	–10.1
Prairie Canada	1,437	1,687	972	–32.4	–42.4
Alberta	180	276	150	–16.7	–45.7
Saskatchewan	1,172	1,348	805	–31.3	–40.3
Manitoba	80	60	16	–80.0	–73.3
Ontario	5	3	1	–80.0	–66.7
U.S. Northern Great Plains	2,032	1,599	1,981	–2.5	23.9
Minnesota	13	10	7	-46.2	–30.0
Montana	308	153	137	–55.5	–10.5
Missouri River	26[e]	24[e]	7[e]	n/a[b]	n/a[b]
North Dakota	992	1,004	1,112	12.1	10.8
Missouri River	307[e]	125[e]	643[e]	n/a[b]	n/a[b]
South Dakota	295	29	390	32.2	1,244.8
Missouri River	292[ef]	29[eg]	390[e]	n/a[b]	n/a[b]
Nebraska	398	375	308	–22.6	–17.9
Missouri River	0[e]	9[e]	8[e]	n/a[b]	n/a[b]
Missouri River (MT, ND, SD, NE)	625[e]	187[e]	1,048[e]	67.7	460.4
Iowa	13	14	11	–15.4	–21.4
Kansas	0	1	3	300.0	200.0
Colorado	13	13	13	0.0	0.0
Oklahoma	0	n.s.[a]	n.s.[a]	n/a[b]	n/a[b]
Great Lakes	40	48	72	80.0	50.0
Canada (Ontario)	0	1	1	100.0	0.0
United States	40	47	71	77.5	51.1
Michigan	39	47	65	66.7	38.3
Wisconsin	1	0	6	500.0	600.0
IL/IN/OH/PA/NY	n.s.[a]	n.s.[a]	0	n/a[b]	n/a[b]
Atlantic	1,645	2,597	2,920	77.5	12.4
St. Pierre and Miquelon (France)	4	6	9	125.0	50.0
Eastern Canada	509	422	481	–5.5	14.0
Newfoundland	7	27	39	457.1	44.4
Quebec	76	104	70	–7.9	–32.7
Prince Edward Island	110	66	112	1.8	69.7
New Brunswick	203	146	167	–17.7	14.4
Nova Scotia	113	79	93	–17.7	17.7
U.S. Atlantic	1,462	2,169	2,430	66.2	12.0
Maine	38	114	96	152.6	–15.8
New Hampshire	n.s.[a]	n.s.[a]	14	n/a[b]	n/a[b]
Massachusetts	293	877	962	228.3	9.7
Rhode Island	47	91	93	97.9	2.2
Connecticut	67	42	45	–32.8	7.1
New York	334[c]	493	624	86.8	26.6
New Jersey	280	225[d]	228	–18.6	1.3
Delaware	10	8	10	0.0	25.0
Maryland	35	91	112	220.0	23.1
Virginia	270	155	198	–26.7	27.7
North Carolina	86	73	48	–44.2	–34.2
South Carolina	2	0	0	–100.0	0.0
Totals					
U.S.	3,534	3,815	4,482	26.8	17.5
Canada	1,946	2,110	1,454	–25.3	–31.1
France	4	6	9	125.0	50.0
Grand total	5,484	5,931	5,945	8.4	0.2

[a] n.s. = not surveyed.
[b] n/a = not applicable.
[c] Adjusted 1991 New York tally due to previous error (originally reported as n = 338; revised n = 334).
[d] Adjusted 1996 New Jersey tally due to previous error (originally reported as n = 209; revised n = 225).
[e] Subtotals for reference only. Missouri River results by state are included in state results.
[f] Adjusted 1991 South Dakota tally due to previous error (originally reported as n = 290; revised n = 292).
[g] Adjusted 1996 South Dakota tally due to previous error (originally reported as n = 27;revised n = 29).

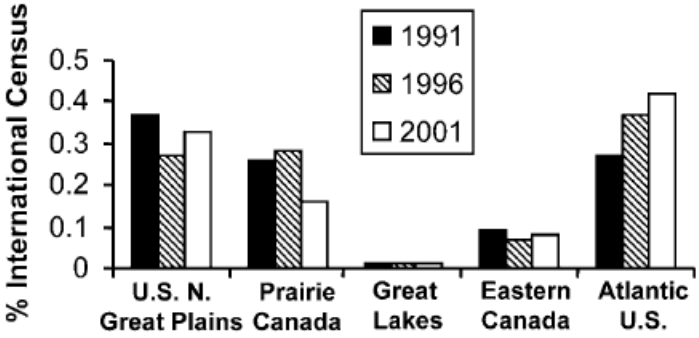


Fig. 2. Changes in breeding distribution and abundance in piping plovers as recorded in international censuses in 1991, 1996, and 2001.

F.10.9 Haig and Oring 1985

Observed Trend:

TABLE 5. Documentation of Piping Plover decline throughout North America (values represent number of adults).

Location	Date 1st estimate	1st estimate	Source	Date 2nd estimate	2nd estimate	% decline	Source
Connecticut	1980	40	Cairns/McLaren 1980	1983	34	15	J. Zickefoose[a]
Delaware	1978	80	L. Master/T. French[a]	1984	18	78	Sidle 1984
Illinois	1876	250	Russell 1983	1983	0	100	Russell 1983
Indiana	1900	100	Russell 1983	1983	0	100	Russell 1983
Lake Manitoba, Manitoba	1980	27	R. Phillips[a]	1984	9	67	Haig 1985
Maryland	1972	85	Cairns/McLaren 1980	1984	25	71	Sidle 1984
Maine	1976	48	J. Arbuckle[a]	1982	12	75	Sidle 1984
Michigan	1979	77	Lambert/Ratcliff 1981	1982	14	82	Russell 1983
Newfoundland	1980	20–30	Cairns/McLaren 1980	1984	4	80–87	J. Brazil[a]
New Hampshire	1958	25	L. Master/T. French[a]	1976	0	100	Master/French[a]
New Jersey	1980	118	A. Galli[a]	1983	64	46	A. Galli[a]
Cadden Beach, Nova Scotia	1976	56	Cairns 1982	1983	28	50	Flemming 1984
Lake Ontario, New York	1935	54	Bull 1974	1974	0	100	Bull 1974
Long Island, New York	1939	1000	Wilcox 1959	1983	200	80	Sidle 1984
Ohio	1935	50	Hicks 1935	1940	0	100	Russell 1983
Long Point, Ontario	1927	200	Snyder 1927	1983	0	100	A. Lambert[a]
Pennsylvania	1940	30	Todd 1940	1964	0	100	Poole 1964
Magdalen Is., Quebec	1973	100	McNeil et al. 1973	1983	40	60	B. Johnson[a]
Rhode Island	1945	80	C. Raithel[a]	1983	20	75	C. Raithel[a]
Big Quill Lake, Sask.	1978	126	Renaud 1979 (60% lake)	1984	149	11	Harris et al. (80%)
Middle Quill Lake, Sask.	1978	19	Renaud 1979	1984	4	79	Harris et al. 1985
Wisconsin	1900	140	Russell 1983	1983	6	96	Russell 1983

[a] Pers. comm.

TABLE 6. Reproductive success of Piping Plovers nesting throughout North America.

Location	Year	No. pairs monitored	No. chicks fledge per pair per year	Source
Maine	1981	10	0.9	J. Arbuckle[a]
	1982	10	1.8	
	1983	6	1.2	
Manitoba	1982	8	0.3	Haig 1985
	1983	8	1.1	
	1984	24	1.3	
Minnesota	1982	15	1.7	Wiens and Cuthbert 1984
	1983	21	2.1	
	1984	24	1.3	
New Brunswick	1982	11	2.4[b]	DuBois and Morgan 1982
New Jersey	1980	59	1.4	A. Galli[a]
	1983	32	1.2	
Nova Scotia	1976	30	1.3–2.1	Cairns 1982
	1983	62	1.2	Flemming 1984
Rhode Island	1981	14	0.6	C. Raithel[a]
	1982	10	0.6	
	1983	11	1.4	
Saskatchewan	1980	9	1.1	A. Whyte[a]
	1981	15	0.4	

[a] Pers. comm.
[b] Mean brood size.

TABLE 7. Causes of Piping Plover nest losses in Manitoba, Minnesota, and Nova Scotia (values show source of destruction/total nests destroyed).

	Lake Manitoba Manitoba[a]	Lake of the Woods Minnesota[b]	Cadden Beach Nova Scotia[c]
Predators	27%	48–70%	8.5%
Storms	45%	19–30%	0.0%
Humans	27%	8–14%	4.4%
Unknown	1%	4–19%	87.1%

[a] Haig 1985 (81.5% of all nests destroyed).
[b] Wiens and Cuthbert 1984.
[c] Cairns 1982.

F.10.10 Plissner and Haig 2000a

Observed Trend:

Table 3
Results of Great Lakes/Great Plains/Prairie baseline model

	Population					
	Metapopulation	Great Lakes	Manitoba and Lake of the Woods	N. Missouri River and Coteau	Nebraska rivers	Colorado
Baseline model						
Probability of survival for 100 years (SE)	0.002 (0.002)	0	0	0.002 (0.002)	0	0
Mean final population size [a] (SE)	18 (0)	0	0	18(0)	0	0
Mean years to first extinction [b] (SE)	55.39 (0.56)	30.64 (0.75)	32.27 (0.44)	51.15 (0.56)	46.71 (0.52)	22.05 (0.39
Population growth rate (r)	−0.136	−0.103	−0.115	−0.142	−0.113	−0.091

[a] For populations persisting 100 years.
[b] For populations going extinct.

Table 4
Results of Atlantic Coast piping plover metapopulation models

	Population				
	Atlantic metapopulation	Atlantic Canada	New England	Mid-Atlantic	Southern Region
Baseline model					
Probability of survival for 100 years	1.000	1.000	1.000	1.000	0.996
Mean final extant population size ±SE)	2172 ± 6.19	579 ± 8.28	1159 ± 7.81	367 ± 9.11	68 ± 2.25
Population growth rate (r)	0.073	0.032	0.126	−0.006	−0.015
Fecundity = 1.25 fledged/year					
Probability of survival for 100 years	1.000	1.000	1.000	1.000	1.000
Mean final extant population size (± SE)	2571 ± 29.07	547 ± 8.85	751 ± 14.68	741 ± 13.89	532 ± 9.55
Population growth rate (r)	0.023	0.022	0.010	0.014	0.021

Table 5
Persistence and size of Great Lakes/Great Plains populations at reproductive success (RS) = 1.7 and RS = 2.0

Population	RS = 1.7 fledged per pair		RS = 2.0 fledged per pair	
	Probability of survival for 100 years (SE)	Mean final population size[a] (SE)	Probability of survival for 100 years (SE)	Mean final population size[a] (SE)
Metapopulation	0.958 (0.009)	694.84 (40.35)	0.998 (0.002)	2398.77 (72.91)
Great Lakes	0.328 (0.021)	84.73 (7.02)	0.798 (0.018)	181.01 (5.00)
Manitoba and Lake of the Woods	0.870 (0.015)	104.08 (3.95)	0.996 (0.003)	185.10 (3.26)
Missouri River/Coteau	0.904 (0.013)	417.23 (34.69)	0.998 (0.002)	1671.19 (66.91)
Platte River, NE	0.884 (0.014)	171.88 (7.32)	0.998 (0.002)	366.43 (6.60)
Colorado	0.758 (0.019)	23.44 (0.64)	0.982 (0.006)	32.06 (0.44)

[a] For populations persisting 100 years.

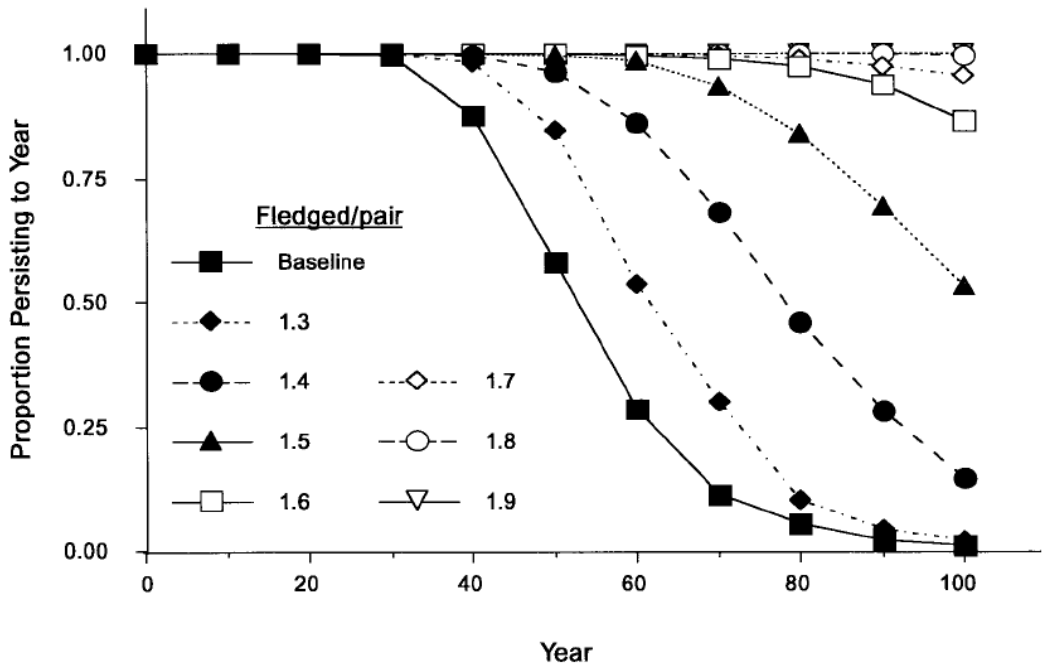


Fig. 2. Probability of Great Lakes/Great Plains piping plover metapopulation persistence relative to mean number of offspring fledged per pair.

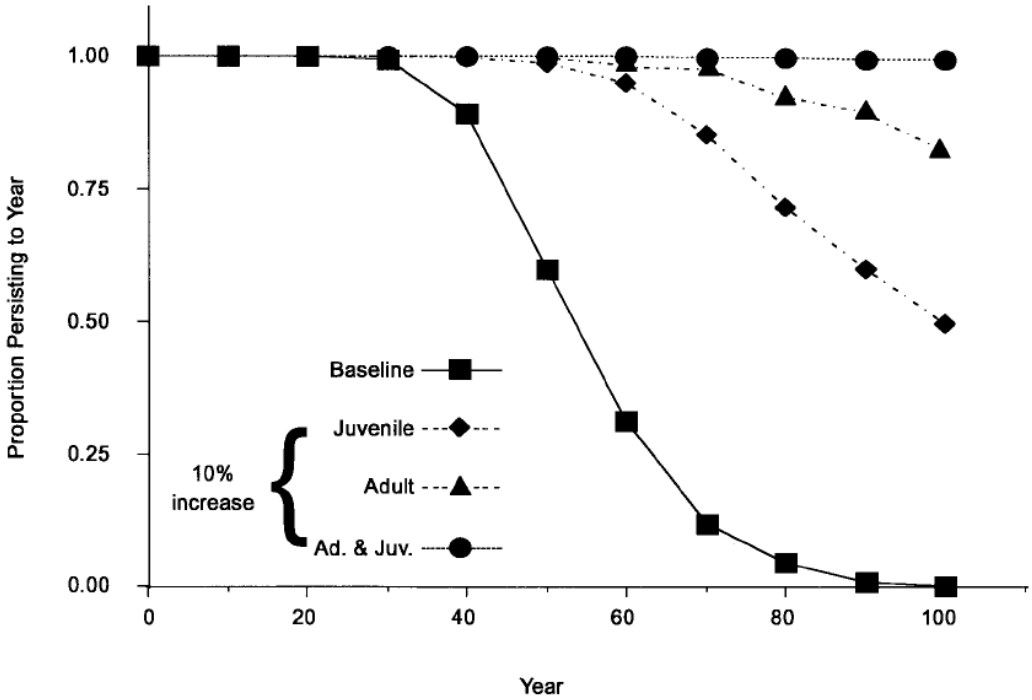


Fig. 3. Probability of Great Lakes/Great Plains piping plover metapopulation persistence as an effect of a 10% increase over baseline adults and/or juvenile survivorship.

Table 6
Connectivity and persistence of Atlantic and Great Lakes/Great Plains metapopulations

	Atlantic		Great Lakes/Great Plains		
Interpopulation dispersal rates	P_s[a]	N_{100}[b] (n)	P_s[a] (SE)	N_{100}[b] (n)	T_{ext1}[c] (SE)
No dispersal	1.000	1676 (16)	0.186 (0.017)	70.94 (93)	69.73 (0.82)
Baseline rates	1.000	2172 (16)	0.014 (0.005)	32 (7)	54.17 (0.58)
2×Baseline	1.000	2483 (17)	0.016 (0.006)	25.88 (8)	46.62 (0.54)
10×Baseline	1.000	3209 (22)	0.010 (0.004)	13.80 (5)	37.68 (0.57)

[a] Probability of survival for 100 years (SE).
[b] Mean final population size of n populations (out of 500 runs) persisting 100 years.
[c] Mean time to first extinction for populations going extinct at least once during runs.

F.10.11 Roche et al. 2010c

Observed Trend:

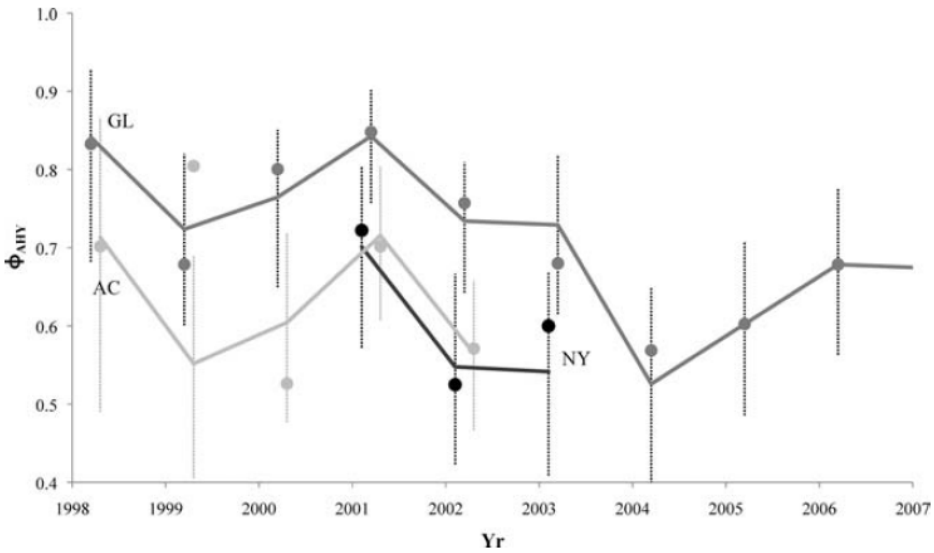


Figure 2. Annual adult apparent survival (Φ_{AHY}) of piping plover populations in North America (1998–2008) using Atlantic Coast nonbreeding habitat based on real parameter estimates generated by model {Φ [(BQL,PC,LD,MR)*t],[(AC,GL,NY)+t], *p*(reduced)} is represented by thick lines. Vertical dashed lines represent the associated 95% confidence intervals. Dots represent raw survival estimates generated from the fully temporal model [Φ (breeding*t), *p*(top)]. Populations using Atlantic Coast nonbreeding habitat were Atlantic Canada (AC), Great Lakes (GL), and Long Island, New York (NY). Populations using Gulf Coast nonbreeding habitat were Big Quill Lake, Prairie Coteau, Lake Diefenbaker, and Missouri River.

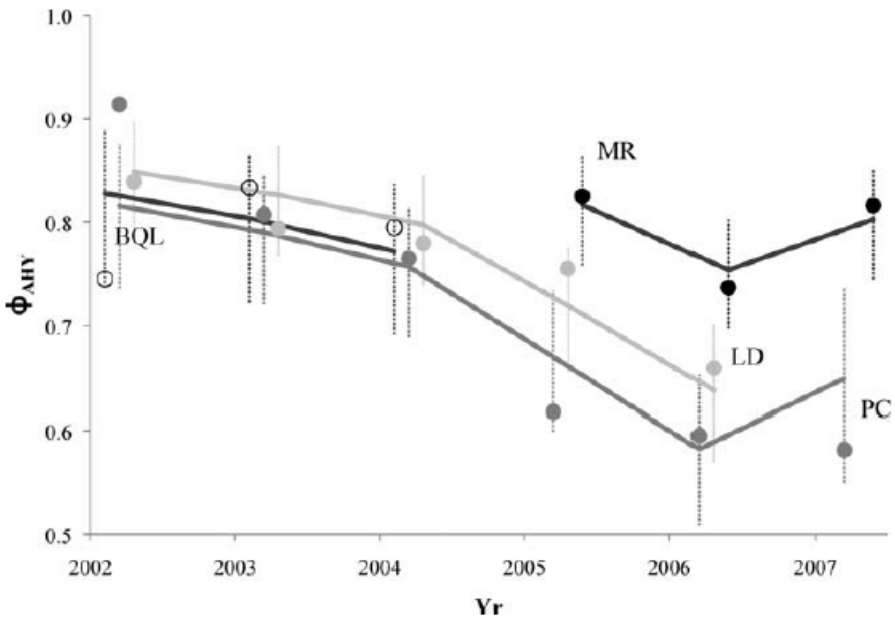


Figure 3. Annual adult apparent survival (Φ_{AHY}) of piping plover populations in North America (1998–2008) using Gulf Coast nonbreeding habitat based on real parameter estimates generated by model {Φ [(BQL,PC,LD,MR)+t],[(AC,GL,NY)*t],*p*(reduced)} is represented by thick lines. Vertical dashed lines represent the associated 95% confidence intervals. Dots represent raw survival estimates generated from the fully temporal model [Φ (breeding*t), *p*(reduced)]. Populations using Atlantic Coast nonbreeding habitat were Atlantic Canada, Great Lakes, and Long Island, New York. Populations using Gulf Coast nonbreeding habitat were Big Quill Lake (BQL), Prairie Coteau (PC), Lake Diefenbaker (LD), Missouri River (MR).

F.10.12 Ryan et al. 1993

Observed Trend:

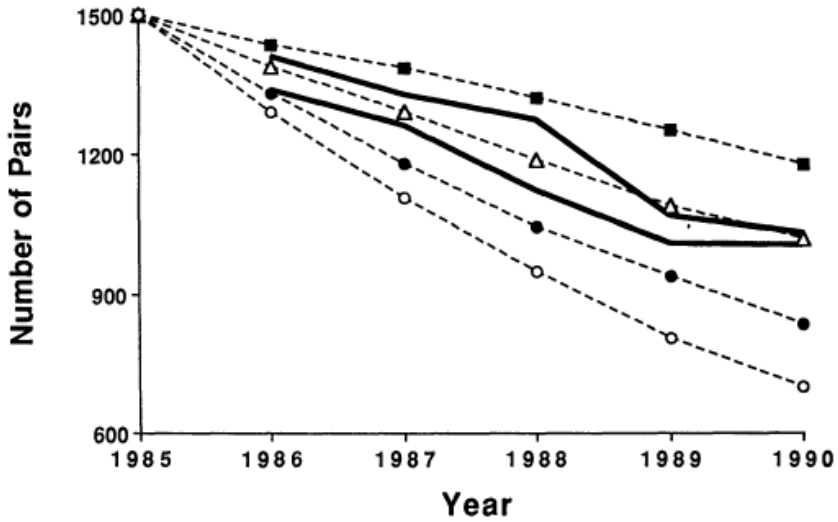


Figure 1. Comparison of count data and simulated population size of the Great Plains Piping Plover. Solid lines represent count data (corrected from Haig 1992); upper line = maximum estimate, lower line = minimum estimate. Population simulations are represented by open circles = [immature survival = 0.7 (adult survival)]; closed circles = [immature survival = 0.8 (adult survival)]; open triangles = [immature survival = 0.9 (adult survival)]; closed squares = [immature survival = adult survival]. See Table 1 for survival and reproductive rate data.

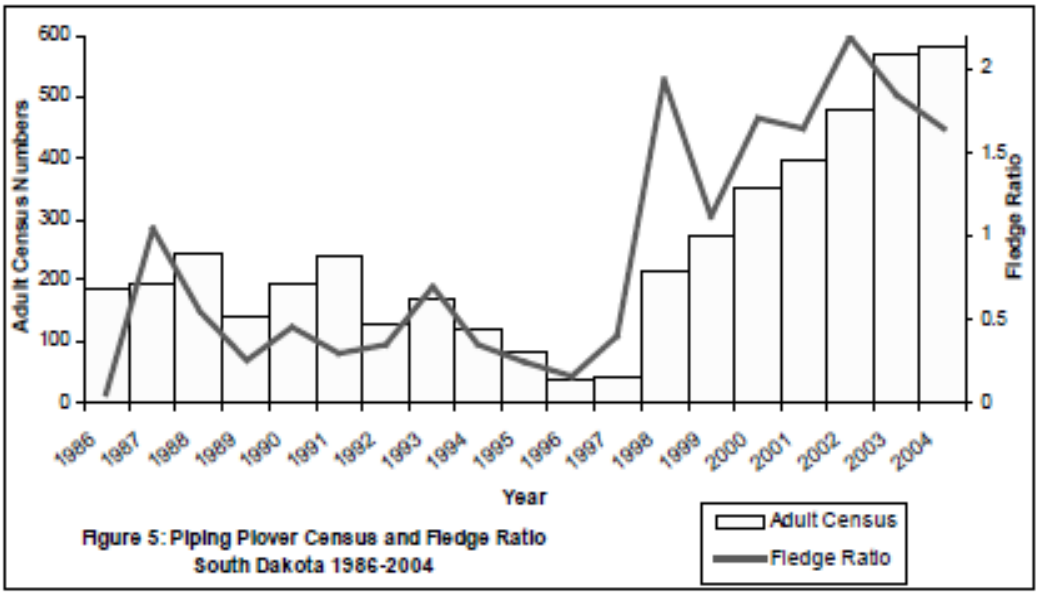
Table 3. The effect of population growth rate and management delays on Piping Plover population growth to recovery levels.

Management initiation year	*Projected population size*[a]	*Population growth rate*	*Years to recovery (95% confidence limit)*
Immediate	1500	2%	30 (26–34)
Immediate	1500	1%	51 (37–61)
Delay 1 year	1386	1%	64 (47–94)
Delay 5 years	1010	1%	118 (88–144)

[a] *Based on simulations under recent demographic conditions (with immature survival = 0.60), see Table 1 for data.*

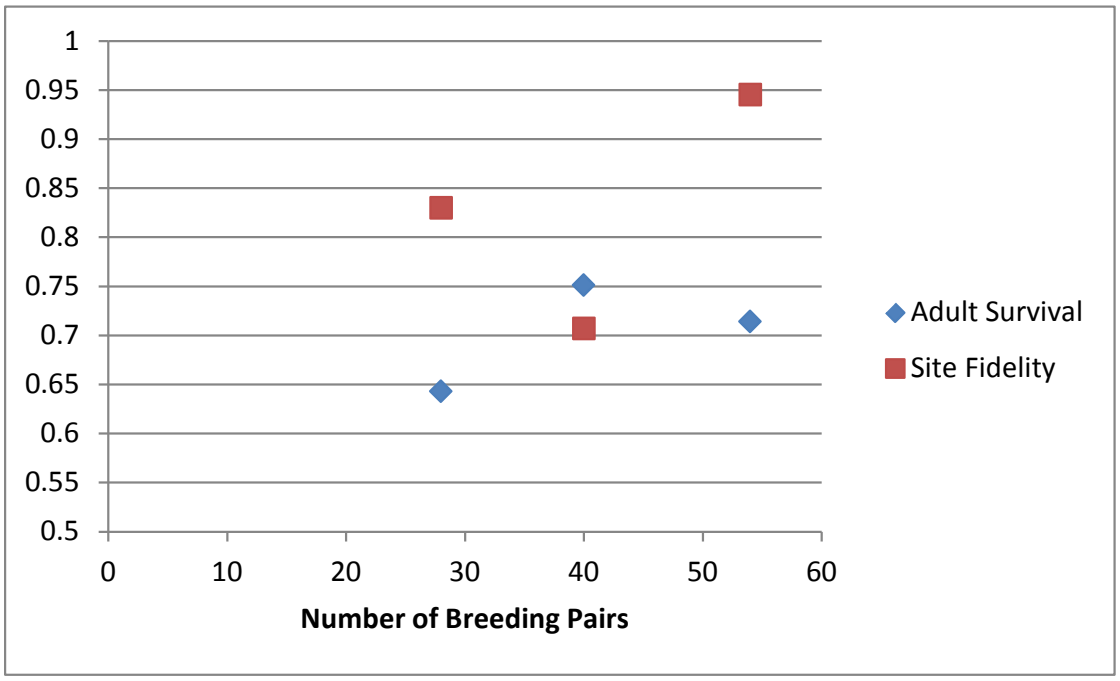
F.10.13 Aron 2005

Observed Trend:



F.10.14 Cohen et al. 2006

Observed Trend:



F.10.15 Cohen and Gratto-Trevor 2011

Study Design:

Table 5. Productivity (fledglings/nesting pair) of Piping Plovers in various study areas in Saskatchewan, 2002–2009.

Site	Year	No. birds	No. nests	Fledglings/ pair	Source
Lake Chaplin	2002	71	36	0.7	White 2005a
	2003	148	41	0.7	White 2005a
	2004	144	75	0.9	White 2005a
	2005	200	123	1.2	Martens and Goossen 2008
Lake Diefenbaker	2002	257	138	1.0[a]	J. P. Goossen and S. M. Westworth, unpubl. data
	2003	164	95	1.4	J. P. Goossen and S. M. Westworth, unpubl. data
	2004	185	97	1.3	J. P. Goossen and S. M. Westworth, unpubl. data
	2005	251	115	0.5[a]	White 2005b
	2006	162	84	0.9	White 2009
	2007	134	83	0.5	White 2009
	2008	154	89	0.2	White 2009
	2009	88	73	1.9	White 2009
Missouri Coteau[b]	2002	98	55	0.7	This study
	2003	148	66	0.8	This study
	2004	152	57	0.8	This study
	2005	150	54	0.9	This study
	2006	140	56	1.2	This study
	2007	-	-	-	
	2008	-	-	0.4	S. Westworth, pers. comm.
Big Quill Lake[b]	2002	64	30	1.1	This study
	2003	88	33	1.8	This study
	2004	104	43	0.8	This study
	2005	128	40	1.2	This study
	2006	118	59	0.2	This study
	2007	-	-	0.1	This study
	2008	-	-	0[c]	This study
	2009	-	-	0[c]	This study

[a]Includes captive-reared fledged chicks.
[b]Intensive study areas only.
[c]Fledging rate assumed to be zero because no breeding habitat remained around flooded lake.

F.10.16 Maxson and Haws 2000

Observed Trend:

Piping Plover populations at LOTW have gradually declined from peak levels during 1982-1985, despite rather intensive management efforts, particularly since 1990. There does not appear to be any single cause for this decline at the local level. Instead, Piping Plovers face a variety of problems (e.g., predators, fluctuating lake levels, weather, habitat degradation and loss, competition with gulls for nesting areas) and a unique mix of circumstances each year. For example, in 1988 water levels were very low, creating excellent habitat conditions, but the failure to trap one mink (four others were removed) and the subsequent predation on plover eggs and chicks (likely by this mink), led to lowered reproductive success. In contrast, 1989 had very high water levels, leading to poor habitat conditions and extensive erosion.

F.11 Detection Probabilities

F.11.1 Shaffer et al. 2013

Nest Detection:

Species	Study Area	Year	No. Opportunities	No. Detections	Daily Detection Probability	95% Confidence Interval	3 Visits	7 Visits
LETE	GRR	2006	241	117	0.49	0.42 – 0.55	0.86	0.99
LETE	GRR	2007	227	107	0.47	0.41 – 0.54	0.85	0.99
LETE	GVP	2008	436	194	0.44	0.40 – 0.49	0.83	0.98
LETE	GVP	2009	290	136	0.47	0.41 – 0.53	0.85	0.99
PIPL	GRR	2006	366	237	0.65	0.60 – 0.70	0.96	1.00
PIPL	GRR	2007	411	205	0.50	0.45 – 0.55	0.87	0.99
PIPL	GVP	2008	572	237	0.41	0.37 – 0.45	0.80	0.98
PIPL	GVP	2009	314	194	0.62	0.56 – 0.67	0.94	1.00
PIPL	SAK	2007	202	107	0.53	0.46 – 0.60	0.90	0.99
PIPL	SAK	2008	167	86	0.51	0.44 – 0.59	0.89	0.99

Chick Detection:

Table 26. Average number of attempts to relocate a fledging-age chick (14–18 days for terns, 18–25 days for plovers), and the estimated detection probability from simulation by species and study area.

[Detection probability values and confidence intervals interpolated from table 25. LETE, least tern; PIPL, piping plover; GRR, Garrison River Reach; GVP, Gavins Point River Reach; SAK, Lake Sakakawea; NA, not applicable]

Species	Study area	Year	Attempts	Detection probability	95-percent confidence interval
LETE	GRR	2006	2.33	0.54	0.47–0.61
LETE	GRR	2007	1.97	0.50	0.43–0.57
LETE	GVP	2008	2.46	0.49	0.46–0.53
LETE	GVP	2009	1.98	0.42	0.39–0.46
PIPL	GRR	2006	NA	NA	NA
PIPL	GRR	2007	2.54	0.86	0.84–0.89
PIPL	GVP	2008	2.45	0.70	0.67–0.73
PIPL	GVP	2009	4.54	0.80	0.78–0.82
PIPL	SAK	2007	3.38	0.99	0.99–0.99
PIPL	SAK	2008	3.41	0.99	0.99–0.99

Appendix G

Images Enlarged from Tern Literature Review Table

Appendix G

Images Enlarged from Tern Literature Review Table

G.1 Reproductive Success

G.1.1 Akcakaya et al. 2003

Fledglings per Nest or Number of Adult Pairs:

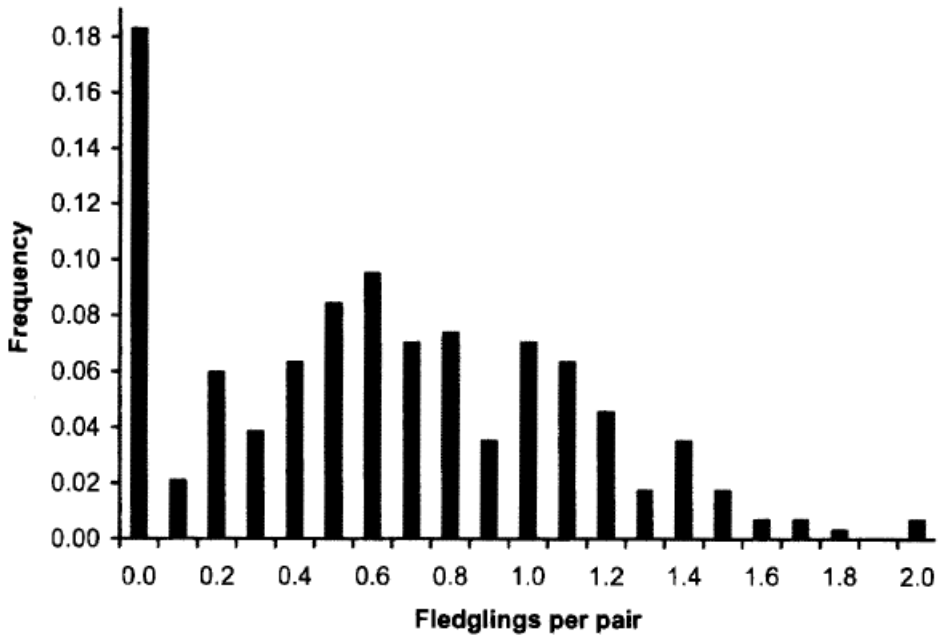


Fig. 3. Frequency distribution of fledglings per pair for all populations of California least tern in California, USA, and Baja California, Mexico, from 1971 to 1998 combined. Each x-value is the upper limit of the class represented by the vertical bar for that value; thus, all data in the "0.0" class is exactly zero.

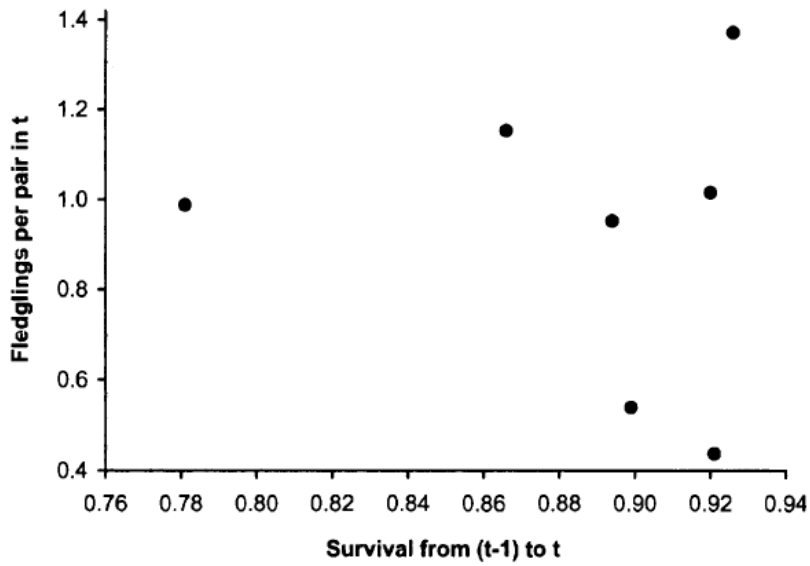


Fig. 4. Number of fledglings per pair of California least tern in year *t*, in the North San Diego population (California, USA), as a function of the adult survival rate from year $t-1$ to t in Camp Pendleton (which is part of this population) for years 1988–1994. The coefficient of correlation is –0.12 (SE = 0.44).

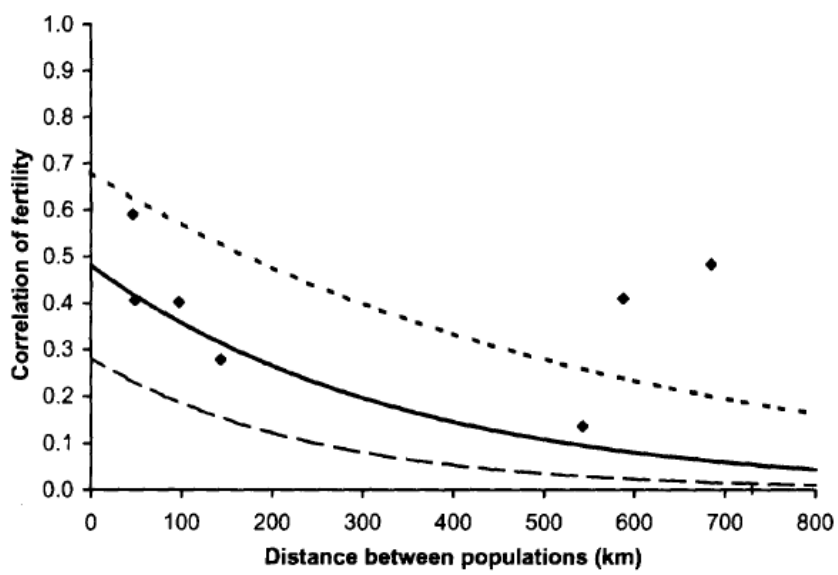


Fig. 7. Correlation in the number of fledglings per pair between populations of California least tern. The points show the observed correlation among several populations with complete data sets. The curves show the low (dashed), medium (solid), and high (dotted) correlation-distance functions used in our model.

G.1.2 Bailey and Servello 2008

Fledglings per Nest or Number of Adult Pairs:

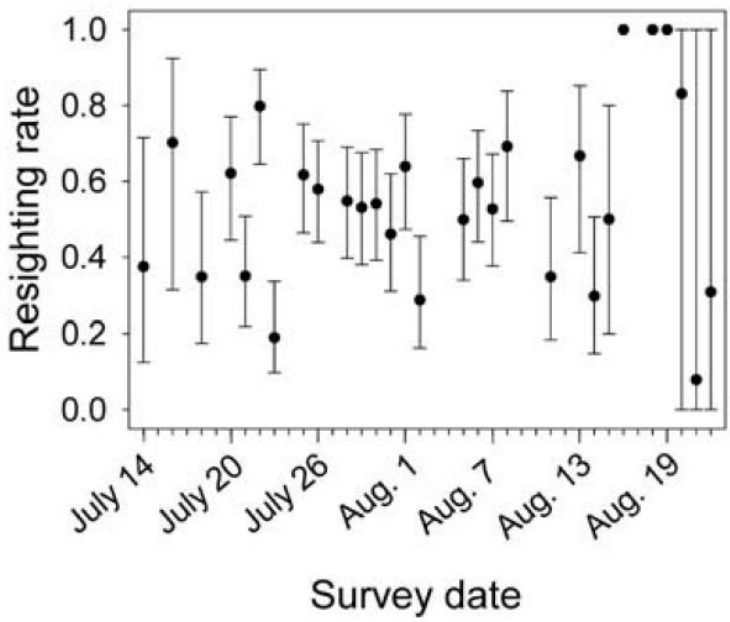


Figure 1. Resighting probabilities (p and 95% CI) for banded Least Tern fledglings for 30 survey dates (29 resighting intervals) at Crescent Surf-Laudholm Beach in 2002.

Sample Size – Years – Design – Strata:

Table 1. Chick survival estimates for three Least Tern breeding colonies in Maine, 2002-2003.

Colony	Number of chicks banded	Proportion of banded chicks resighted as fledglings[1]	Chick survival (Φ) based on mark-resighting analyses: Daily (Φ ± SE)	Chick survival (Φ) based on mark-resighting analyses: 21-d chick period
Crescent Surf-Laudholm Beach	96	0.74	0.9864 ± 0.0069	0.73
Higgins Beach	62	0.71	0.9855 ± 0.0042	0.74
Crescent Surf Beach	53	0.15	0.9098 ± 0.0148	0.14

[1]Chicks were classified as fledglings when age was ≥21 d.

G.1.3 Brooks et al. 2013

Nest Fates:

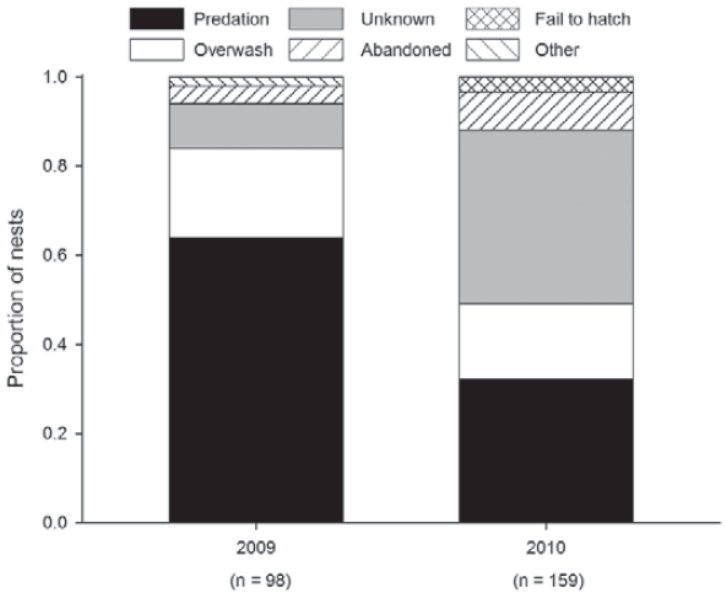


Figure 2. Cause of nest loss for Least Terns (*Sternula antillarum*) in Cape Romain National Wildlife Refuge, South Carolina, 2009-2010.

G.1.4 Brunton 1997

Nest Success – Nest Fate – Fledglings per Nest or Number of Adult Pairs:

TABLE 1. Hatching and fledging success, and causes of nest failure within regions of the colony during 1987 and 1988.

	1987			1988		
	Edge	Center	Total	Edge	Center	Total
Hatching success						
# Eggs hatched/nest						
$\bar{x}$	1.39	1.28	1.31	0.77	0.80	0.79
SD	0.08	0.07	0.08	0.08	0.07	0.07
Fledging success						
# Chicks hatched/nest						
$\bar{x}$	0.94	0.41	0.57	0.12	0.01	0.06
SD	0.08	0.09	0.08	0.07	0.09	0.08
95% CI*	0.85–0.95	0.37–0.45	0.55–0.57	0.09–0.17	0.00–0.02	0.04–0.08
(Total # chicks)	(136–152)	(142–174)	(298–310)	(21–42)	(1–9)	(24–49)
Causes of failure						
Herons	1	46	47	75	204	279
(%)			(32.9)			(73.8)
Crows	7	0	7	39	2	41
(%)			(4.9)			(10.8)
Human	2	10	12	0	0	0
(%)			(8.4)			(0.0)
Tides	7	0	7	1	0	1
(%)			(4.9)			(0.3)
Abandoned	24	22	46	11	16	27
(%)			(32.2)			(7.1)
Unknown	5	19	24	13	17	30
(%)			(16.8)			(7.9)
Total failed nests	46	97	143	139	239	378
(% of each region)	(28.6)	(25.3)	(26.2)	(57.2)	(65.8)	(62.4)
Number of nests	161	384	545	243	363	606

* Confidence limits for proportions (Fisher and Yates 1963).

G.1.5 Conway et al. 2003

Nest Initiation:

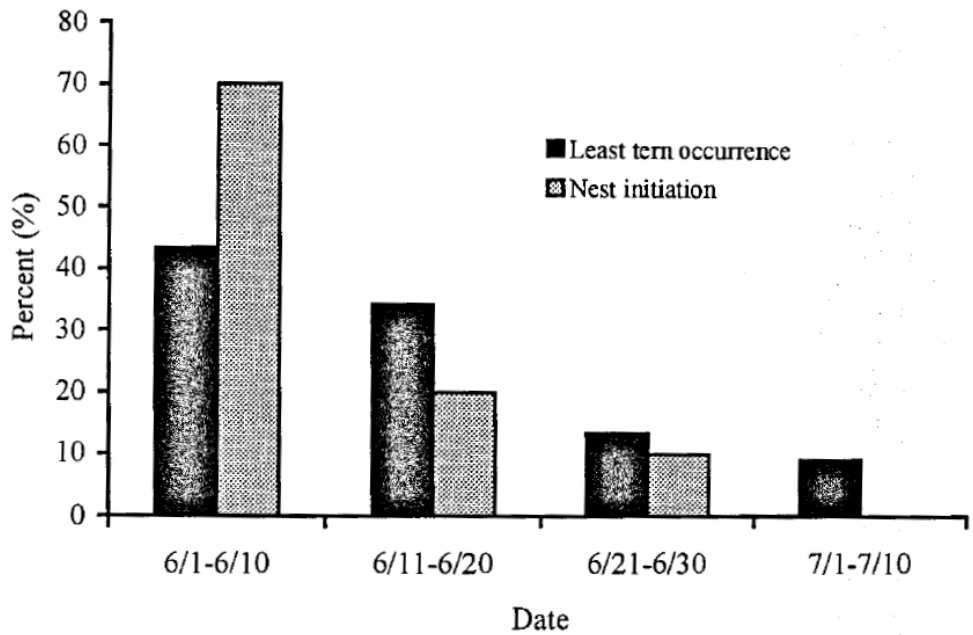


Figure 1. Chronology of occurrence and nest initiation (% of total number for each category) for interior least terns nesting in the Prairie Dog Town Fork of the Red River in Childress County, Texas, 1998.

Nest Success – Nest Fate:

Table 1. Nesting parameters of interior least terns nesting in the Prairie Dog Town Fork of the Red River in Childress County, Texas, 1998.

Nesting parameters	Least tern nests
Total number of nests (*n*)	20
Successful nests (*n*)	13
Abandoned nests (*n*)	2
Nests with unknown status (*n*)	5
Raw nest success (%)	65
Mayfield estimate (%)	70.7
Clutch ($\bar{x}$)	2.25
Range	1—3
Nests located on gravel substrate (*n*)	5
Nests located on sand substrate (*n*)	15
Nest dimensions (north to south) (cm) ($\bar{x}$)	9.7
Nest dimensions (east to west) (cm) ($\bar{x}$)	10.2
Nest depth (from nest rim to nest bowl bottom) (cm) ($\bar{x}$)	1.9

G.1.6 Dugger et al. 2002

Fledglings per Nest or Number of Adult Pairs:

Table I. Estimates of least tern reproduction on the LMR from colonies adjacent to Missouri during 1986–1993 (1986–1989; data from Smith and Renken, 1993)

Year	No. pairs	No. fledglings	Chicks/pair	Fledglings/pair
1986	275	141	1.01	0.5
1987	298	210	1.70	0.7
1988	277	392	1.68	1.4
1989	499	96	0.73	0.2
1990	706	334	0.75	0.5
1991	1135	836	1.07	0.7
1992	587	342	0.61	0.6
1993	**	**	**	0.0
Mean (S.E.)	540 (117.9)	336 (93.2)	1.09 (0.17)	0.58 (0.15)

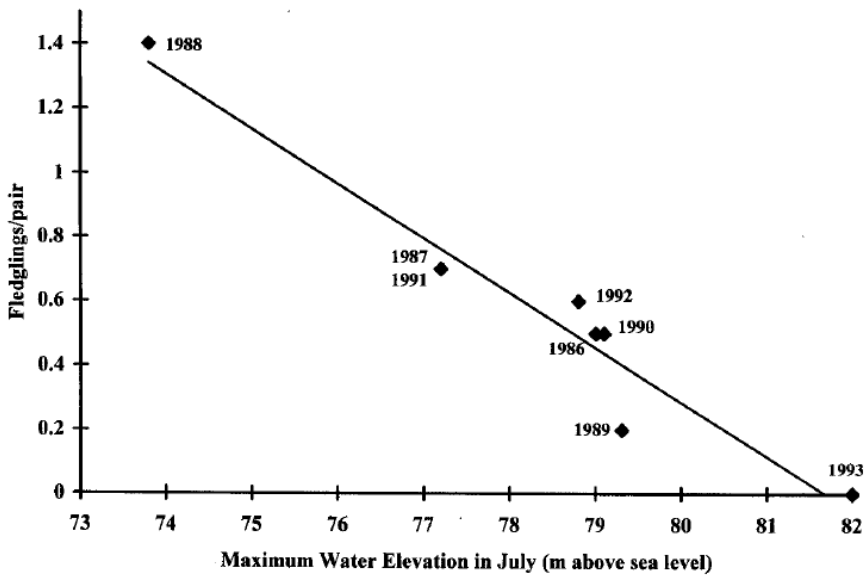


Figure 3. The relationship between July maximum river elevation from the LMR at Caruthersville, MO and estimates of the number of fledglings/pair for tern colonies between Rkm 1521 and 1334 during 1986–1993

Sample Size – Years – Design – Strata:

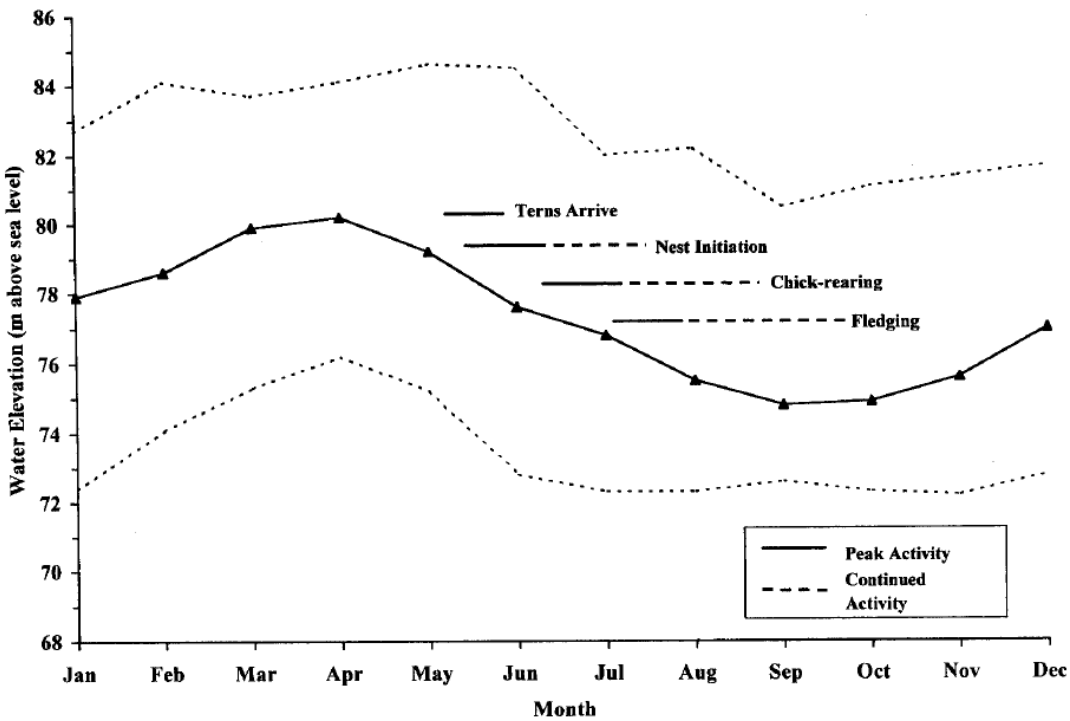


Figure 1. The relationship between tern reproductive chronology and the long-term annual hydrograph (1943–1995) (see Tibbs and Galat, 1998) with monthly mean, minimum and maximum river elevations for the LMR at Caruthersville, MO; Rkm 1362

G.1.7 Dugger et al. 2000

Fledglings per Nest or Number of Adult Pairs:

TABLE 3. Fledging success of Interior Least Terns reported as mean number of fledglings per breeding pair (FL/PR).

Site	FL/PR	Range (n)[a]	Source
Missouri River (Montana)	0.7	—	Rabenburg et al. (1993)
Fort Peck Reservoir	1.3	—	Rabenburg et al. (1993)
Yellowstone Reservoir	0.2	—	Rabenburg et al. (1993)
North Dakota	0.6	0.4–0.9 (4)	Mayer (1993)
Missouri River[b] (South Dakota)	0.4	0.2–0.7 (6)	Schwalbach et al. (1993)
Missouri River[c] (South Dakota)	0.4	0.1–0.6 (6)	Schwalbach et al. (1993)
Cimarron River (Kansas)	0.5	0.0–1.4 (23)	Boyd (1993)
Lower Mississippi River (Missouri)	0.7	0.2–1.4 (7)	Dugger (1997)
Lower Mississippi River (Missouri)	0.85	0.7–1.0 (2)	This Study

[a] Refers to number of years or sites in average.
[b] Below Gavins Point Dam.
[c] Below Fort Randall.

G.1.8 Elliott et al. 2007

Nest Initiation:

Table 1. Mean nest initiation dates for the Alameda Point Least Tern colony from 2000-2004.

Year	Mean nest initiation date ± SD	N
2000	5 June ± 13	312
2001	22 May ± 10	275
2002	24 May ± 10	326
2003	28 May ± 15	368
2004	26 May ± 15	440

Nest Success – Nest Fate:

Table 2. Clutch sizes and egg fates for the Alameda Point Least Tern colony from 2000-2004.

	Year				
	2000	2001	2002	2003	2004
(A) Clutch size (% nests)					
1-egg	55 (17.6)	17 (6.2)	40 (12)	68 (18.5)	148 (33.6)
2-eggs	251 (80.4)	247 (89.8)	267 (82)	287 (78.0)	281 (63.9)
3-eggs	6 (2.0)	11 (4.0)	18 (5)	13 (3.5)	10 (2.3)
4-eggs	0	0	0	0	1 (0.2)
5-eggs	0	0	1 (1)	0	0
Mean ± SD (N)	**1.84** ± 0.41 (312)	**1.98** ± 0.32 (275)	**1.93** ± 0.42 (325)[a]	**1.85** ± 0.45 (368)	**1.69** ± 0.52 (440)
(B) Egg fates (% eggs)					
hatched	360 (62.6)	489 (89.9)	435 (68.7)	516 (75.8)	554 (74.6)
failed to hatch	105 (18.3)	28 (5.1)	56 (8.8)	75 (11.0)	63 (8.5)
abandoned	19 (3.3)	7 (1.3)	30 (4.7)	63 (9.2)	78 (10.5)
depredated	11 (1.9)	1 (0.2)	44 (7.0)	0	1 (0.1)
died while hatching	1 (0.2)	0	0	0	1 (0.1)
unknown	79 (13.7)	19 (3.5)	64 (10.1)	25 (3.7)	46 (6.2)
unknown if attended[b]	0	0	4 (0.6)	0	0
incidental take	0	0	0	2 (0.3)	0
TOTAL	575	544	633	681	743
Mean hatching success ± SD (N)	**0.67** ± 0.39 (284)	**0.89** ± 0.26 (275)	**0.67** ± 0.42 (326)	**0.70** ± 0.43 (368)	**0.70** ± 0.43 (440)

[a]Results do not include the 5-egg clutch.
[b]Undetermined if nest was attended during 21 day incubation period; egg did not hatch.

Fledglings per Nest or Number of Adult Pairs:

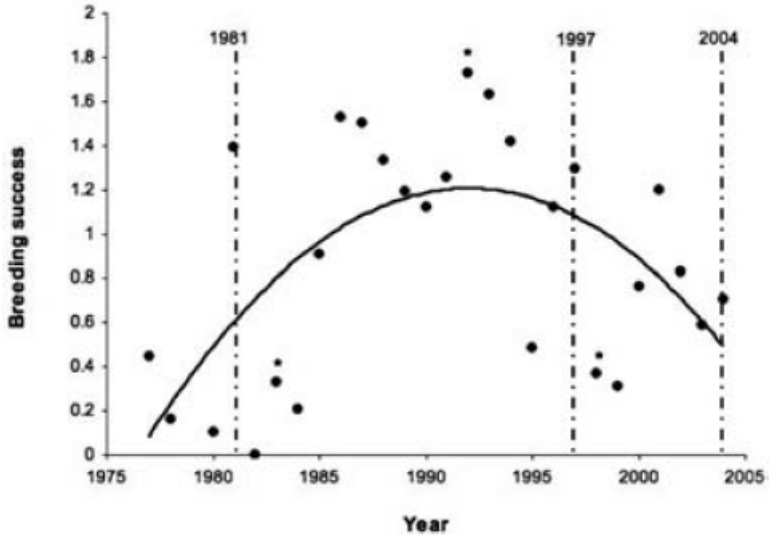


Figure 3. Breeding success (fledglings / breeding pair) of the Alameda Point Least Tern colony from 1976-2004. (* = El Niño years. Years of important site changes are noted: 1981 = electric fence was erected; 1997 = Naval Air Station closure; 2004 = chain link fence erected.)

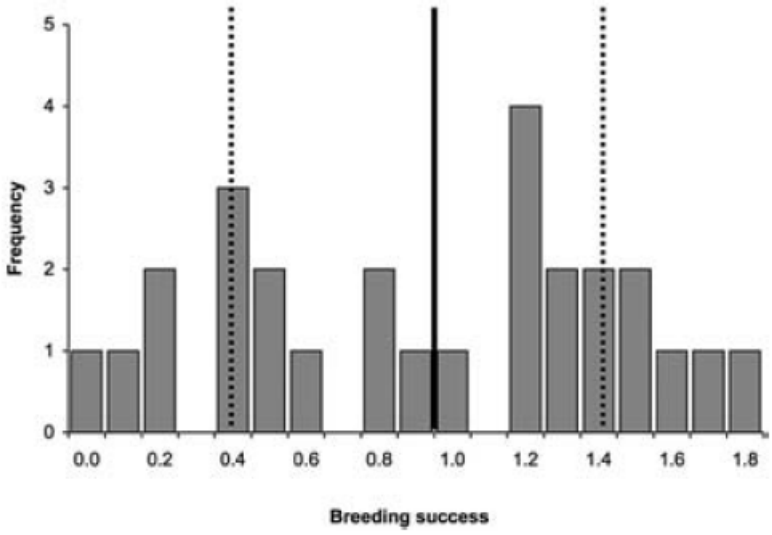


Figure 4. Frequency distribution of breeding success (fledglings/breeding pair) at the Alameda Point Least Tern colony from 1976-2004. (Note: the solid line represents the mean, and the dashed lines represent ± 1 SD of the mean.)

G.1.9 Fancher 1992

Fledglings per Nest or Number of Adult Pairs – Sample Size – Years – Design:

Table 1. California least tern breeding colonies: Numbers of breeding pairs and in parentseses (number of fledlings produced). When ranges were estimated, numbers here represent averages.

Colony	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990
San Francisco Bay Area													
PGE, Pittsburgh	- -	- -	- -	- -	- -	- -	8 (8)	4 (4)	1 (2)	4 (5)	3 (1)	4 (2)	3 (0)
Port Chicago	2 (1)	3 (2)	12 (5)	- -	- -	6 (1)	- -	- -	- -	3 (2)	1 (0)	- -	- -
NAS Alameda	80 (13)	40 (0)	60 (5)	74(103)	73 (0)	3 (1)	44 (9)	53 (48)	43 (75)	58 (87)	63 (84)	74 (84)	99(105)
Oakland Airport	- -	- -	- -	- -	12 (11)	61 (8)	15 (2)	12 (9)	32 (11)	9 (4)	7 (0)	7 (2)	8 (0)
Bair Island	- -	4 (0)	38 (14)	23 (28)	53 (58)	22 (0)	3 (2)	- -	- -	- -	- -	- -	- -
San Luis Obispo County													
Oso Flaco Lake	- -	7 (0)	7 (3)	- -	2 (0)	1 (0)	- -	- -	- -	- -	- -	- -	4 (0)
Santa Barbara County													
Guadalupe Dunes	19 (15)	21(10)	17 (15)	25 (8)	12 (3)	7 (3)	10 (2)	11 (10)	13 (0)	23 (36)	11 (0)	18 (13)	32 (7)
San Antonio Creek	9 (6)	4 (4)	2 (0)	4 (4)	6 (2)	18 (12)	17 (0)	14 (5)	3 (0)	2 (0)	7 (3)	3 (0)	1 (0)
Purisma Point	5 (7)	27(25)	28 (20)	30 (12)	18 (1)	14 (9)	20 (2)	18 (3)	- -	14 (0)	3 (1)	17 (2)	9 (3)
Santa Ynez River	- -	- -	- -	- -	- -	8 (4)	- -	- -	- -	4 (6)	- -	3 (0)	- -
Ventura County													
Santa Clara River	13 (12)	18(25)	13 (14)	23 (25)	19 (16)	3 (2)	8 (6)	12 (7)	14 (15)	8 (10)	3 (4)	6 (8)	27 (34)
Ormond Beach[a]	- -	7 (3)	- -	- -	7 (0)	4 (2)	- -	5 (0)	- -	0 (0)	3 (0)	3 (1)	42 (60)
Point Mugu	11 (0)	- -	12 (1)	- -	13 (0)	22 (15)	18 (5)	60 (70)	51 (40)	20 (3)	100 (25)	86(118)	12 (0)
Los Angeles County													
Venice Beach	68 (75)	88(140)	158(240)	150(195)	170 (60)	145(140)	83 (94)	96(113)	104(113)	109 (82)	165(192)	137(134)	206(279)
Playa del Rey	28 (30)	22(25)	2 (0)	16 (0)	- -	- -	- -	- -	- -	- -	- -	- -	- -
Terminal Island	- -	- -	- -	38 (7)	65 (15)	78 (90)	117 (70)	60 (63)	79 (83)	40 (5)	5 (0)	19 (6)	32 (12)
Costa del Sol	- -	- -	- -	18 (12)	21 (2)	23 (14)	5 (7)	33 (0)	4 (0)	- -	- -	- -	- -
San Gabriel River	63 (70)	53(60)	2 (0)	16 (0)	- -	- -	- -	- -	- -	- -	- -	- -	- -
Orange County													
Anaheim Bay	- -	6 (0)	41 (24)	43 (20)	19 (2)	4 (2)	22 (33)	20 (3)	41 (81)	69(103)	83 (66)	97(104)	102(147)
Bolsa Chica	- -	39 (6)	23 (15)	63 (20)	90 (75)	141 (45)	103 (50)	111 (65)	68 (81)	80 (63)	92 (65)	115 (45)	217(190)
Huntington Beach St Pk	83(100)	88(90)	80 (85)	113(168)	98 (50)	88 (60)	70 (20)	45 (42)	69 (34)	58 (9)	86 (43)	70 (5)	46 (18)
Upper Newport Bay	9 (0)	7 (0)	4 (0)	- -	- -	9 (2)	6 (0)	- -	22 (26)	43 (33)	73 (55)	74 (55)	70 (85)

[a] Multiple nesting areas were combined.

Table 1. California least tern breeding colonies (cont.)

Colony	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990
San Diego County													
White Beach	- -	15 (8)	70 (22)	23 (10)	1 (0)	12 (9)	12 (4)	7 (11)	4 (0)	- -	- -	12 (14)	20 (21)
Sta Margarita River[a]	35 (8)	36(22)	56 (4)	80 (50)	131 (50)	237 (91)	234(113)	197(108)	163(220)	192 (60)	246(387)	151 (67)	293(306)
Buena Vista Lagoon	- -	- -	1 (2)	3 (1)	- -	- -	- -	- -	4 (0)	- -	- -	16 (6)	- -
Agua Hedionda	13 (4)	26 (9)	12 (4)	4 (0)	- -	- -	- -	- -	- -	- -	- -	- -	- -
Batiquitos Lagoon	25 (0)	39(33)	27 (17)	39 (26)	25 (6)	1 (2)	3 (6)	18 (24)	17 (0)	8 (18)	48 (28)	3 (0)	27 (26)
San Elijo Lagoon	9 (0)	12 (7)	17 (8)	12 (8)	28 (12)	28 (23)	20 (10)	13 (0)	9 (2)	13 (2)	11 (3)	17 (0)	11 (3)
Los Penasquitos	22 (10)	16 (0)	15 (0)	- -	- -	- -	- -	- -	- -	- -	- -	- -	- -
N. Fiesta Island	9 (8)	15 (4)	8 (4)	8 (1)	55 (75)	68 (0)	- -	- -	4 (2)	- -	- -	- -	- -
FAA Island	145 (5)	96(48)	150(190)	75 (80)	- -	12 (18)	60 (0)	- -	55 (22)	25 (3)	37 (50)	125 (30)	177(135)
Mariner's Point	- -	- -	- -	- -	- -	- -	- -	- -	- -	- -	- -	2 (0)	25 (37)
Other Mission Bay[a]	- -	- -	- -	- -	42 (5)	- -	- -	- -	- -	- -	- -	- -	- -
Lindbergh Field	43 (10)	108(53)	71 (31)	- -	8 (3)	27 (14)	12 (2)	- -	11 (10)	50 (60)	80 (30)	9 (0)	- -
Naval Training Center	10 (5)	- -	- -	- -	- -	- -	- -	- -	6 (6)	11 (0)	1 (1)	- -	- -
North Island NAS	36 (0)	78(70)	100 (9)	60 (5)	66 (28)	75 (90)	45 (40)	83 (25)	35 (0)	6 (4)	20 (4)	24 (14)	38 (23)
Delta Beach	4 (4)	11 (3)	- -	- -	- -	- -	- -	13 (17)	43 (25)	28 (10)	7 (10)	33 (20)	45 (54)
Chula Vista Wildlife Res	- -	- -	57 (31)	97 (35)	73 (14)	75 (9)	19 (4)	- -	- -	- -	24 (35)	28 (7)	70 (32)
D Street Fill	47 (15)	26(18)	14 (0)	- -	1 (2)	1 (0)	16 (15)	44 (0)	6 (7)	28 (10)	19 (0)	2 (0)	- -
Coronado Cays	9 (10)	39 (7)	- -	- -	- -	- -	- -	- -	- -	- -	- -	- -	- -
Saltworks	29 (2)	29 (9)	21 (4)	1 (0)	- -	- -	15 (4)	30 (6)	12 (2)	21 (4)	17 (15)	28 (3)	25 (10)
Tijuana River Mouth[a]	10 (8)	28(19)	38 (25)	12 (15)	26 (17)	63 (50)	66 (16)	37 (24)	39 (33)	21 (16)	44 (30)	49 (20)	72 (28)
Total breeding pairs	832	1000	1160	1067	1130	1264	1046	1003	962	944	1253	1240	1708
Total fledglings produced	(418)	(696)	(769)	(833)	(511)	(894)	(518)	(654)	(904)	(633)	(1130)	(764)	(1612)

[a] Multiple nesting areas were combined.

G.1.10 Jenniges and Plettner 2008

Nest Initiation:

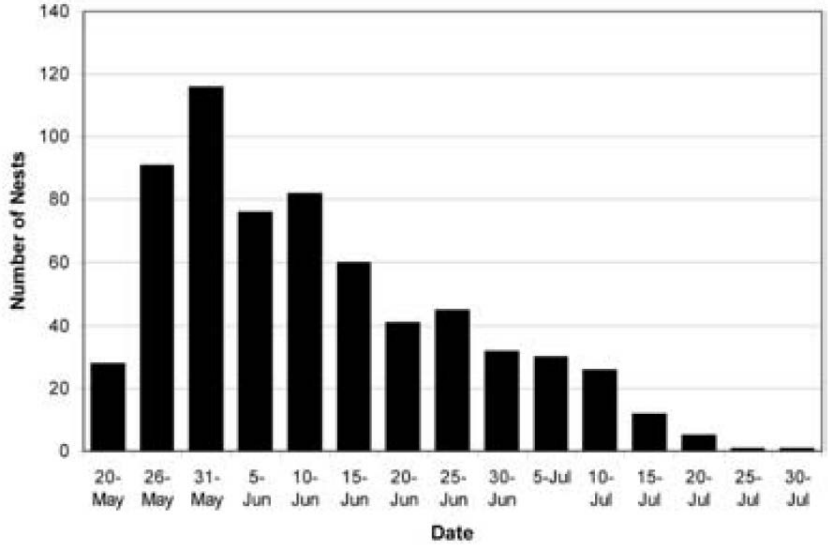


Figure 2. Nest initiation dates for Least Terns at sites monitored along the central Platte River, Nebraska 1991 to 2005.

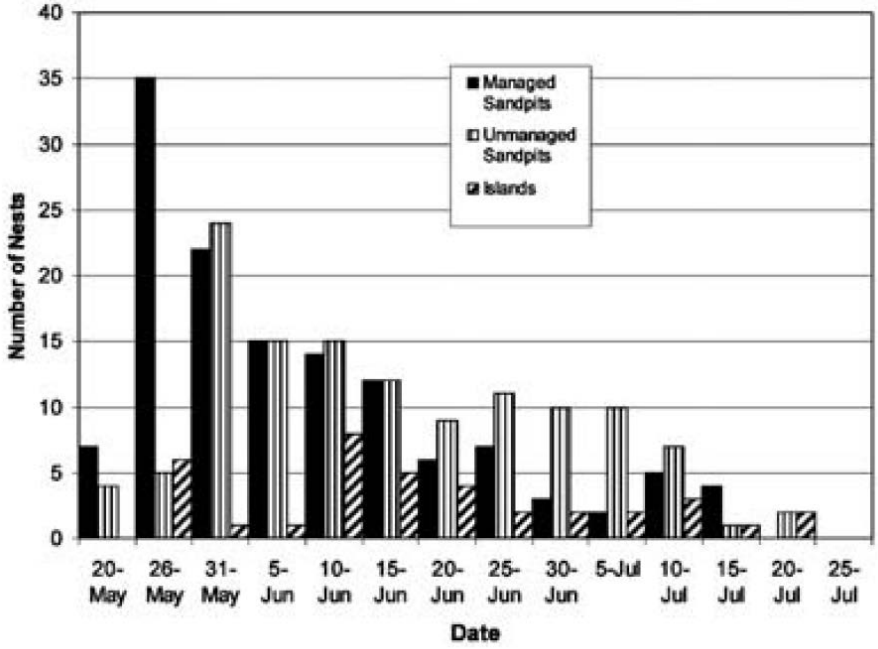


Figure 5. Nest initiation dates for Least Tern nests at managed sandpits, unmanaged sandpits and created islands 1994-1997.

Nest Success – Nest Fate:

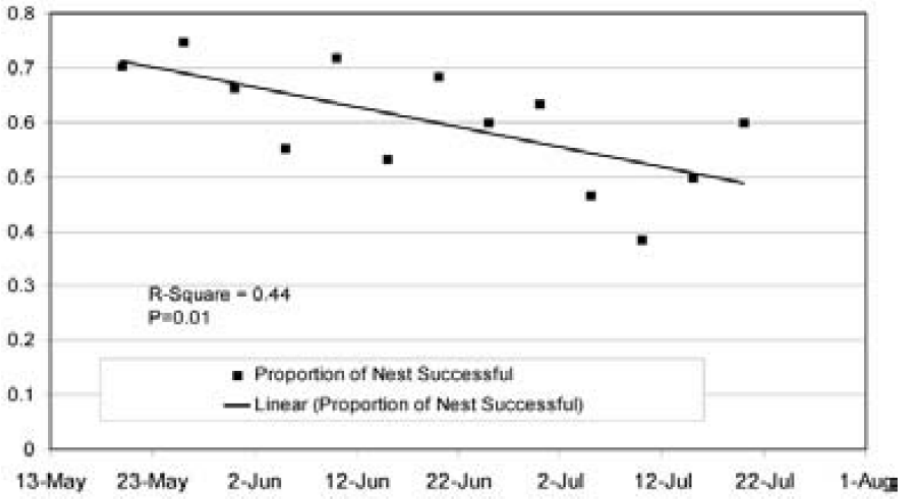


Figure 4. Regression analysis of the proportion of observed Least Tern nests which were successful through the nesting at human created habitat along the central Platte River, Nebraska.

Table 3. Nest success for Least Terns nesting at sites monitored 1991-2005. Success shown as apparent as well as the Mayfield estimate with 95% confidence intervals calculated.

Management type	Number of nests	Apparent nest success	Mayfield nest success	Mayfield lower 95% C.I.	Mayfield upper 95% C.I.
Unmanaged sandpits	125	38%	36%	29%	45%
Managed sandpits	473	65%	64%	59%	68%
Managed riverine islands	49	71%	69%	56%	84%
Total	647	63%	61%	57%	65%

Fledglings per Nest or Number of Adult Pairs:

Table 4. Least Tern reproductive output at nesting sites monitored. Successful nests are those that hatched at least one egg, successful broods are those broods where at least one chick fledged.

Management type	Total nests	Successful nests	Estimated successful broods	Number of initial chicks observed	Number of chicks fledged	Percent of chicks fledged
Unmanaged sandpits	125	48	32	101	54	53%
Managed sandpits	473	309	254	679	534	79%
Managed river islands	49	35	28	74	51	70%
Total	647	392	314	854	639	75%

Table 5. Fledge ratios for Least Terns at sites monitored along the central Platte River. Results present as fledglings per nest and fledglings per pair. A pair is defined as the maximum number of nest and broods present during a single visit.

Management type	Number nests	Number of pairs	Fledglings per nest	Fledglings per pair
Unmanaged sandpits	125	97	0.43	0.56
Managed sandpits	473	409	1.13	1.31
Managed river islands	49	47	1.04	1.09
Total	647	553	0.99	1.16

Sample Size – Years – Design:

Table 1. Areas monitored for Least Tern nesting along the central Platte River. *Indicates the area is known to have had nesting outside the dates indicated but was not part of this monitoring effort.

Site	Location	Years monitored	Years with nesting	Years managed
Johnson Sandpit	Elm Creek	1991-2005	1991-2005*	1991-2005
Lexington Sandpit	Lexington	1992-2005	1992-2005*	1992-2005
Bluehole Sandpit	Elm Creek	1994-2005	1994-2005	1997-2005
Overland Sandpit	Central City	1994-1997	1994-1997	None
T&F Odessa	Odessa	1994-2005	1994, 1995, 1997	None
T&F Elm Creek	Elm Creek	1994-2002	1994, 1995	None
Whitetail	Columbus	1997	1997*	None
Elm Creek Island	Elm Creek	1991-2005	1991, 1995-1998	1991-2005
Overton Island	Overton	1993-2005	1993, 1995-1999	1993-2005
Lexington Island	Lexington	1992-2005	None	1992-2005

G.1.11 Kirsch 1996

Nest Initiation:

Table 11. Distribution of dates of nest initiation for least terns on river and sandpit colonies along the lower Platte River, Nebraska, 1987–90.

Year	Range of nest initiation dates		Median nest initiation date		No. of nests in analysis		Test statistic[a]	*P*
	River	Sandpits	River	Sandpits	River	Sandpits		
1987	21 May–30 Jun	28 May–3 Jul	4 Jun	5 Jun	36	41	0.100	>0.98
1988	21 May–25 Jun	23 May–18 Jun	5 Jun	31 May	61	44	0.250	>0.18
1989	20 May–11 Jul	21 May–10 Jul	7 Jun	1 Jun	74	46	0.250	>0.08
1990	31 May–7 Jul	24 May–14 Jul	11 Jun	6 Jun	104	95	0.135	>0.78

[a] Kolmogorov–Smirnov test statistic of maximum distance.

Nest Success – Nest Fate – Fledglings per Nest or Number of Adult Pairs:

Table 13. Nest and fledging success, and fledglings per pair for least terns on sandbars and sandpits along the lower Platte River, Nebraska, 1987–90.

	No. of study colonies		Nest success				Fledging success				Fledglings per pair			
			Sandbars		Sandpits		Sandbars		Sandpits		Sandbars		Sandpits	
Year	Sandbars	Sandpits	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
1987	3	5	0.49	0.25	0.46	0.18	0.14	0.04	0.37	0.21	0.21	0.12	0.28	0.08
1988	4	3	0.72	0.14	0.72	0.12	0.27	0.02	0.24	0.13	0.73	0.15	0.55	0.23
1989	6	6	0.41	0.08	0.54	0.06	0.35	0.10	0.38	0.11	0.57	0.17	0.50	0.06
1990	6	7	0.44	0.05	0.64	0.12	0.12	0.08	0.33	0.05	0.25	0.16	0.64	0.14

Table 16. Causes of egg loss for least tern colonies along the lower Platte River, Nebraska, 1987–90.

	Year and habitat										
	1987			1988			1989			1990		
Egg variables	Sandbar		Sandpit	Sandbar		Sandpit	Sandbar		Sandpit	Sandbar		Sandpit
Number of eggs	171		188	197		143	185		145	247		298
Porportion of eggs												
Abandoned	0.09		0.11	0.09		0.04	0.10	***	0.15	0.02	***	0.10
Depredated	0.19	*	0.24	0.12	***	0.17	0.15		0.10	0.02	***	0.14
Human destroyed	0.01		0.00	0.02		0.02	0.02		0.04	0.00	***	0.02
Flooded	0.05	***	0.00	0.04	*	0.00	0.28	***	0.00	0.53[a]	***	0.00
Other losses[b]	0.00	**	0.04	0.03		0.01	0.02	*	0.10	0.00		0.00
Unknown outcome	0.06		0.05	0.04		0.08	0.00		0.02	0.01		0.04

[a] In 1990, all early eggs were lost (approx 122 eggs as calculated from no. of pairs renesting and the average clutch size before the flood: (53 nests) × (2.3 eggs/nest). Only 7 eggs from renests were flooded.
[b] Includes severe weather and disturbance by nonpredatory animals.
* Chi-square differences between sandbars and sandpits each year, $P < 0.05$.
** Chi-square differences between sandbars and sandpits each year, $P < 0.01$.
*** Chi-square differences between sandbars and sandpits each year, $P < 0.001$.

Table 23. Summary of productivity estimates for least terns in the Great Plains.

Location	Years	Nest success	Fledglings per pair	Frequency of visits to sites	No. pairs monitored	Source
Missouri River, N.D.	1988	0.62	0.42[a]	7–10 days	42	Mayer and Dryer 1989
	1989	0.56	0.21	7–19 days	52	
Missouri River, S.D.	1986		0.20	3–5 times/season	102	Schwalbach 1988
	1987		0.64		146	
	1988	0.36	0.44[b]	7–10 days	148	Dirks 1990
	1989	0.51	0.55	7–10 days	121	
Lower Platte River, Nebr.	1987–90	0.60	0.50	2–3 days	291	This study
Central Platte River, Nebr.	1986	0.62	0.26[c,d,e]	2–3 days	68[f]	Lingle 1993*a*
	1987	0.72	0.11[c,d,e]		75[f]	
	1988	0.52	0.33–0.46[c,d]		94[f]	
	1989	0.57	0.44–1.44[c,d,e]		84[f]	
	1990	0.60	0.19[c,d,e]		68[f]	
Mississippi River, Mo.	1986	0.65[g]	0.40	variable	381[g]	Smith and Renken 1993
	1987	0.69	0.6		397	
	1988	0.59	1.2		455	
	1989	0.45	0.2		912	
Cimmeron River, Kans.	1980		1.19	>3 times/season	46	Schulenberg and Ptacek 1984
	1981		1.08		41	
	1982		0.56		36	
Salt Plains NWR, Okla.	1977	0.64		1–3 days	28	Grover and Knopf 1982
	1978	0.38			42	
	1987	0.12			100–105	
	1987	0.44–0.33	0.44–0.15	1–3 days		Hill 1985
	1990	0.46	0.64	variable	120	Boyd 1990
Quivera NWR, Okla.	1980–90	0.10–0.57[h]	0.00–1.00[h]	variable	15–34	Boyd 1990
Optima Reservoir, Okla.	1986	0.60	1.00	variable	26	Boyd 1987
	1987	0.59	1.07		30	

[a] Chicks seen at 12 days of age or older were assumed to fledge.
[b] Chicks seen at 15 days of age or older were assumed to fledge.
[c] Estimates for number of nests, not pairs.
[d] Cursory estimate.
[e] Fledglings per nest.
[f] Number of nests monitored.
[g] Mayfield weighted average.
[h] Values are a range for the years of study.

G.1.12 Kirsch and Sidle 1999

Fledglings per Nest or Number of Adult Pairs:

Table 4. Estimated fledging success for interior least terns in selected local areas, 1980–96.

Local area	Years	Fledglings/pair $\bar{x}$[a]	SD	Range	Source(s)
Yellowstone River, Montana	1994–96	0.73[b]	0.651	0.10–1.40	Bacon and Rotella 1998
Missouri River, Fort Peck Reservoir, Montana	1991–95	0.24	0.272	0.00–0.50	USFWS[c], Bismark, North Dakota, unpublished data; USACE[d], Omaha, Nebraska, annual reports 1993–95
Fort Peck Dam, Montana, to Lake Sakakawea, North Dakota	1988–95	0.94	0.586	0.20–1.66	Annual reports, Montana Piping Plover Recovery Committee, 1988–95; USFWS, unpublished data
Garrison Dam to Lake Oahe, North Dakota	1986–95	0.59	0.187	0.36–0.93	P. M. Mayer and P. J. Dryer. 1989. unpublished report, USFWS, Bismark, North Dakota; USACE, Omaha, Nebraska, annual reports 1993–95
Lake Oahe, North and South Dakota	1988–95	0.48	0.634	0.00–1.62	USFWS, Pierre, South Dakota, unpublished data; USACE, Omaha, Nebraska, annual reports 1993–95
Fort Randall Dam, South Dakota to Niobrara, Nebraska	1986–95	0.19	0.190	0.00–0.48	Schwalbach 1988; Dirks 1990; USFWS, Pierre, South Dakota, unpublished data; USACE, Omaha, Nebraska, annual reports 1993–95
Lewis and Clark Lake, South Dakota and Nebraska	1989–95	0.58	0.764	0.00–2.09	USFWS, Pierre, South Dakota, unpublished data; USACE, Omaha, Nebraska, annual reports 1993–95
Gavins Point Dam, South Dakota and Nebraska to Ponca, Nebraska	1986–91	0.45	0.206	0.16–0.55	Schwalbach 1988; Dirks 1990; USFWS, Pierre, South Dakota, unpublished data; USACE, Omaha, Nebraska, annual reports 1993–95
Council Bluffs, Iowa	1984–91	0.70	0.514	0.00–1.40	Dinsmore et al. 1993
Sioux City, Iowa	1995	1.60			Huser 1996
Lower Platte River, Nebraska	1986–90	0.49	0.158	0.27–0.61	Kirsch 1992
Central Platte River, Nebraska	1986–90	0.49[b]	0.546	0.11–0.46	Lingle 1993*b*
Arkansas River watershed reservoirs Colorado	1990–92	1.22	0.720	0.47–1.90	D. L. Nelson and M. F. Carter, Colorado Bird Observatory, unpublished data
Quivira NWR, Kansas	1980–87, 1989–91	0.47	0.460	0.00–1.41	Boyd 1993
Cimarron River, Kansas and Oklahoma	1980–82	0.75	0.185	0.56–0.93	Schulenberg and Ptacek 1984
	1990	0.44[e]	0.180	0.27–0.75	Boyd 1993
	1992	0.06[e]	0.076	0.00–0.21	R. L. Boyd, Baker University, unpublished data
Salt Plains NWR, Oklahoma	1982–84	0.22	0.081	0.15–0.44[f]	Hill 1985
	1989–90	0.52	0.170	0.40–0.64	R. L. Boyd, Baker University, unpublished data
Arkansas River, Oklahoma	1986–88, 1990–91, 1995–96	0.67	0.182	0.46–0.90	F. Pianalto and J. Hoffman, unpublished data, cited in Hill 1993; USACE, Tulsa, Oklahoma, unpublished data
	1992–93	0.59	0.018	0.48–0.70[f]	Wood 1994
Canadian River, Oklahoma	1987, 1991–96	1.23	0.637	0.50–2.03	A. M. Archibeque, unpublished data; V. J. Byre, unpublished data cited in Hill 1993; V. J. Byre, Canadian River Preserve, unpublished data; N. Kuhnert, Canadian River Preserve, unpublished data

Table 4. Continued.

Local area	Years	Fledglings/pair $\bar{x}$[a]	SD	Range	Source(s)
Lower Mississippi River[g]	1986–89	0.70	0.510	0.20–1.40	Smith and Renken 1993
	1995–96	1.00	0.460	0.68–1.33	M. S. Woodry and C. Szell, Mississippi Museum of Natural Science, unpublished report
Wabash River, Gibson Lake, Indiana	1986–87	1.31	0.900	0.20–2.33	Johnson and Castrale 1993
Bitter Lake NWR area, New Mexico	1986–92	0.54	0.660	0.00–2.00	S. O. Williams, N.M. Game and Fish Dept., unpublished data
Optima NWR, Oklahoma	1986–92	0.52	0.608	0.00–1.10	R. L. Boyd, Baker University, unpublished data; M. Eddings, personal communication, cited in Hill 1993

[a] Sum of fledging success estimates for each year and area (not by colony) divided by the number of years estimates were made.
[b] Fledglings/nest.
[c] U.S. Fish and Wildlife Service.
[d] U.S. Army Corps of Engineers.
[e] Fledglings/pair averaged over all areas in 1 year.
[f] Data from Hill (1985, 1993) and Wood (1994) represent "best" and "worst" cases based on a range of pair and fledgling estimates for the area, and did not represent an estimated range of fledging success from different sites within the area or among years. Means and standard deviations were calculated from midpoints of the range for each year's estimates of fledging success.
[g] Smith and Renken (1993) studied least terns along the Mississippi River south of Cape Girardeau, Missouri, in Missouri, Illinois, Kentucky, and Tennessee. M. S. Woodry and C. Szell studied least terns along the Mississippi River between Memphis, Tennessee and Rosedale, Mississippi, in Mississippi and Arkansas.

G.1.13 Koenen et al. 1996

Nest Success – Nest Fate:

TABLE 1. Least Tern and Snowy Plover nest success (Mayfield Method), 95% confidence intervals, and losses for all nests on and off of nest ridges (includes nests on mounds), Salt Plains National Wildlife Refuge, Oklahoma.

	1990[a]	1991		1992		1993		1994		All years	
	On	On	Off	On	Off	On	Off	On	Off	On	Off
Least Terns											
n	2	4	0	7	7	13	15	8	6	32	28
Nest Success	0	1.00	—	0.61	0.60	0.23	0.47	1.00	0.62	0.53	0.53
Confidence Interval (95%)	—	—	—	0.30–1.00	0.29–1.00	0.08–0.64	0.23–0.91	—	0.23–1.00	0.36–0.79	0.34–0.83
Predation	2	0	0	0	0	0	3	2	1	2	4
Flood	0	0	0	2	2	8	5	0	1	10	8
Other nest losses[b]	0	0	0	1	0	0	0	1	1	2	1
Unknown nest outcome	0	0	0	0	0	2	1	1	1	3	2
Snowy Plover											
n	1	2	—	7	13	8	6	5	7	22	26
Nest Success	0	1.00	—	0.71	0.62	0.79	0.12	1.00	1.00	0.79	0.62
Confidence Interval (95%)	—	—	—	0.45–1.00	0.35–1.00	0.49–1.00	0.01–0.92	—	—	0.58–1.00	0.42–0.48
Predation	1	0	—	1	0	0	0	0	1	1	1
Flood	0	0	—	2	3	1	4	0	0	3	7
Other nest losses[b]	0	0	—	0	0	2	0	0	0	2	0
Unknown nest outcome	0	0	—	0	1	2	0	1	1	3	2

[a] From Boyd (1990).
[b] Includes nests with cracked eggs, abandoned nests, and chicks that died during hatching process.

TABLE 2. Least Tern and Snowy Plover nest success (Mayfield Method), 95% confidence intervals, and losses for all nests in and out of electric fences, Salt Plains National Wildlife Refuge, Oklahoma.

	1991		1992		1993		1994		All years	
	In	Out	In	Out	In	Out	In	Out	In	Out
Least Tern										
n	4	11	14	9	28	50	14	59	60	129
Nest Success	1.00	0.17	1.00	0.51	0.78	0.75	0.61	0.52	0.81	0.56
Confidence Interval (95%)	—	0.03–0.78	—	0.23–1.00	0.59–1.00	0.58–0.97	0.34–1.00	0.36–0.74	0.68–0.96	0.44–0.70
Predation	0	5	0	3	3[b]	5	3[c]	13	6	26
Flood	0	0	4	3	13	23	1	1	18	27
Other nest losses[a]	0	3	1	1	0	3	2	5	3	12
Unknown nest outcome	0	0	0	2	3	9	2	9	5	20
Snowy Plover										
n	2	9	20	23	18	67	12	24	52	123
Nest Success	1.00	0.82	0.91	0.55	0.87	0.77	0.89	1.00	0.90	0.76
Confidence Interval (95%)	—	0.55–1.00	0.75–1.00	0.34–0.86	0.66–1.00	0.61–0.97	0.70–1.00	—	0.79–1.00	0.64–0.88
Predation	0	1	1[d]	7	1	5	1[b]	0	3	13
Flood	0	1	5	7	5	30	0	1	10	39
Other nest losses[a]	0	1	0	0	0	7	0	2	0	10
Unknown nest outcome	0	0	1	3	6	15	2	5	9	23

[a] Includes nests with cracked eggs, abandoned nests, and chicks that died during hatching process.
[b] 1 nest predated by a Ring-billed gull.
[c] Predated by a coyote on 14 Jun. 1994.
[d] Predated by a Cattle Egret (Utych 1993)

G.1.14 Kruse et al. 2001

Nest Success—Nest Fate – Fledglings per Nest or Number of Adult Pairs:

TABLE 1. Productivity summary for Piping Plovers and Least Terns nesting on the Gavins Point and Fort Randall river reaches of the Missouri River, South Dakota, 1991 and 1992.

	Piping Plover	Least Tern
Total Nests	238	354
Total Destroyed	128 (53.8%)	179 (50.6%)
Predation	61 (25.6%)	80 (22.6%)
Chicks Hatched	368	330
Chicks Fledged	57 (15.5%)	81 (24.5%)

Apparent nesting success of caged nests (n = 86, 62%) was higher ($P < 0.001$) than for control nests (n = 122, 34%). Of 33 caged nests that were unsuccessful, 6 (18%) were inundated, 16 (48%) were lost to predation, 2 (6%) were abandoned for unknown reasons, and 9 (27%) were lost due to human disturbance, sandbar erosion, weather, or unknown causes.

G.1.15 Leslie et al. 2000

Nest Fate:

TABLE 2—Losses of least tern nests and chicks (percentages in parentheses) on the Arkansas River, Oklahoma.

Year	Category	Total observed[a]	Total lost[a]	Losses		
				Flood-related	Predation	Cause unknown[b]
1992	Nests	192	77 (40)	35 (46)	18 (23)	24 (31)
	Chicks	118	38 (32)	9 (24)	10 (26)	19 (50)
1993	Nests	345	133[c] (39)	12 (9)	20 (15)	100 (75)
	Chicks	285	142 (50)	0 (0)	1 (<1)	141 (>99)

[a] Tallied from regular monitoring of colonies from river bank; June through August, 1992–1993.

[b] Nests and chick that disappeared between observation days and were assumed lost; causes could have included flooding and predation.

[c] One nest destroyed by an all-terrain vehicle in 1993.

Fledglings per Nest or Number of Adult Pairs:

TABLE 1—Numbers of adult breeding pairs and fledglings per breeding pair of least terns using number of successful nests, median-date method, probable renests subtracted from the total number of nests, and airboat surveys (see methods for description) on the Arkansas River, Oklahoma.

		Census methods			
Year	Category	Successful nests	Median date	Probable renests	Airboat survey
1992	Adult pairs	140[a]	166	159	125
	Fledglings/breeding pair	0.59	0.50	0.52	0.66
1993	Adult pairs	212[a]	299	296	203
	Fledglings/breeding pair	0.67	0.48	0.48	0.70

[a] Considered a minimal estimate of adult pairs because it did not account for unsuccessful pairs.

G.1.16 Lombard et al. 2010

Nest Success:

TABLE 1. Daily and period nest survival for Least Terns at St. Croix, U.S. Virgin Islands, 2003–2006. Period nest survival is the daily nest survival extrapolated to 21 days. Estimates summarized by habitat (substrate), year, and colony size. Managed colonies = exclosure or platform. Small = 10–30, medium = 40–90, large ≥110 nests. Standard errors were adjusted for design effect.

Category	Daily nest survival	Period nest survival
Habitat		
Salt flats	0.910 (0.017)	0.138
Sandy beach	0.936 (0.036)	0.252
Offshore cay	0.929 (0.005)	0.215
Industrial	0.929 (0.016)	0.195
Managed	0.968 (0.023)	0.511
Year		
2003	0.902 (0.025)	0.115
2004	0.934 (0.025)	0.242
2005	0.923 (0.012)	0.188
2006	0.906 (0.023)	0.127
Colony size		
Small	0.876 (0.041)	0.062
Medium	0.883 (0.023)	0.074
Large	0.927 (0.001)	0.204

G.1.17 Massey and Atwood 1981

Nest Initiation:

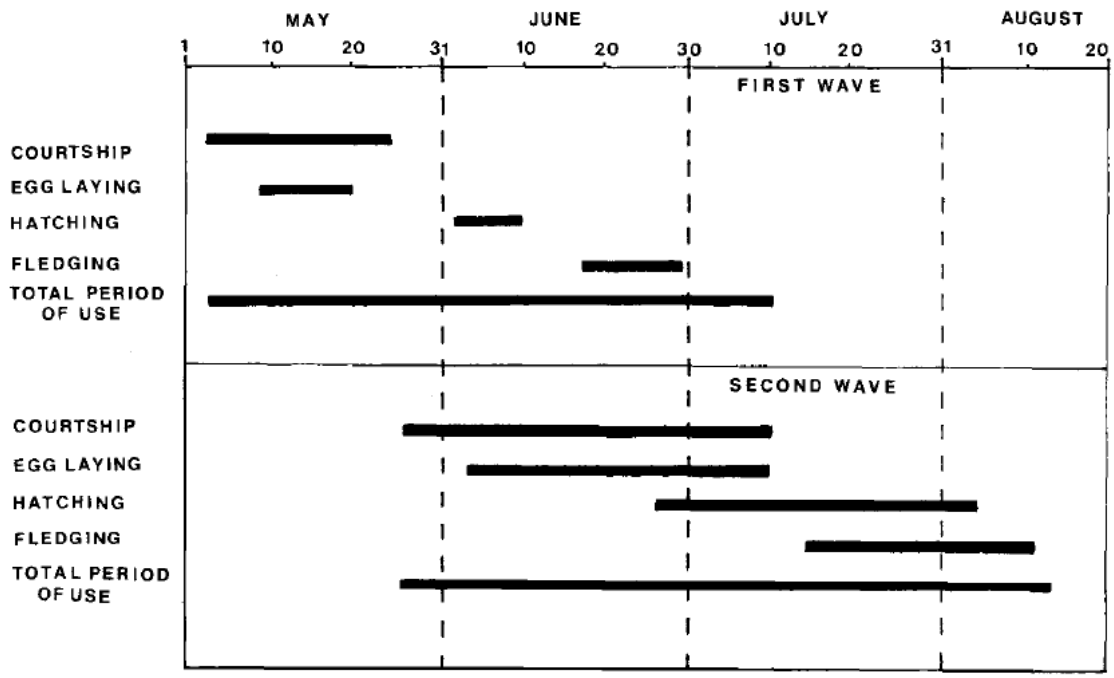


Fig. 1. Chronology of the California Least Tern nesting colony in Anaheim Bay, 1980.

TABLE 2. Egg-laying period—second nesting wave, Anaheim Bay 1980.

	Date of laying of first egg											
	June								July			
	1	4	8	12	16	20	24	28	1	4	8	12
Total number of nests (n = 29)[a]		1	2	1 2	2	1 2	2	2 1	4	2 3	1	3
Nests of 2-yr-old terns (n = 10)[b]				1	1	1	1	1	1	1 2		1
Nests of terns ≥3 yr of age (n = 2)[c]				1								1

[a] The total number of nests in the second wave was 33; laying dates of four abandoned nests could not be determined.
[b] Twelve 2-yr-old terns nested in the second wave at 10 nests; in two instances both pair members were banded 2-yr-olds.
[c] Three birds ≥3 yr of age nested in the second wave at two nests; both pair members were banded and were ≥3 yr old at one nest.

Nest Success – Nest Fate:

TABLE 3. Clutch sizes of first- and second-wave nests at several California Least Tern colonies in 1980.

Colony	Number of eggs			
	1	2	3	$\bar{x} \pm$ SE
First wave				
Venice Beach (36/150)[a]	3 (8%)	31 (86%)	2 (6%)	1.97 ± 0.063
Huntington Beach (29/65)	3 (10%)	25 (86%)	1 (3%)	1.93 ± 0.069
Anaheim Bay (6/10)	1 (17%)	2 (33%)	3 (50%)	2.33 ± 0.333
Second wave				
Anaheim Bay (33/33)	11 (33%)	21 (64%)	1 (3%)	1.70 ± 0.092
Anaheim Bay—nests of 2-yr-old birds (10/10)	3 (30%)	7 (70%)	0	1.70 ± 0.153

[a] Sample size = 36 nests, colony size = 150 nests. The 2 × 3 table of clutch size/colony was subjected to a χ^2 test and revealed a significant difference in clutch size between the first and second waves of nesting ($P < 0.01$).

Fledglings per Nest or Number of Adult Pairs:

TABLE 4. Fledging success at several California Least Tern colonies of varying sizes in 1980.

Colony	Number of nesting pairs	Minimum number of fledglings	Fledglings/ nesting pairs[a]	Fledging success
Venice Beach	150–165	240	1.45	Good
Huntington Beach	70–90	85	0.94	Moderate
Los Cerritos	7	6	0.86	Moderate
Bolsa Chica	20–26	15	0.58	Moderate
Anaheim Bay (total)	38–43	24	0.56	Moderate
First wave	8–10	5	0.50	Moderate
Second wave	33	19	0.58	Moderate
Second wave (nests of 2-yr-old birds)	10	8	0.80	Moderate

[a] Minimum number of fledglings/maximum number of nesting pairs.

Sample Size – Years – Design:

TABLE 1. Ages of banded, breeding California Least Terns.

	Age (yr)				
	1	2	≥3	Unknown	Total
Birds nesting at Anaheim Bay in 1980 (n = 23)					
First nesting wave	0	0	2	2	4
Second nesting wave	0	12 (63%)	3[a] (16%)	4 (21%)	19
Birds nesting in Los Angeles and Orange Counties, 1976–1980 (n = 62)					
First nesting wave	0	2 (5%)	39 (95%)	0	41
Second nesting wave	0	16 (76%)	5[a] (24%)	0	21

[a] All renesting after initial failures.

G.1.18 Schuetz 2011

Fledglings per Nest or Number of Adult Pairs:

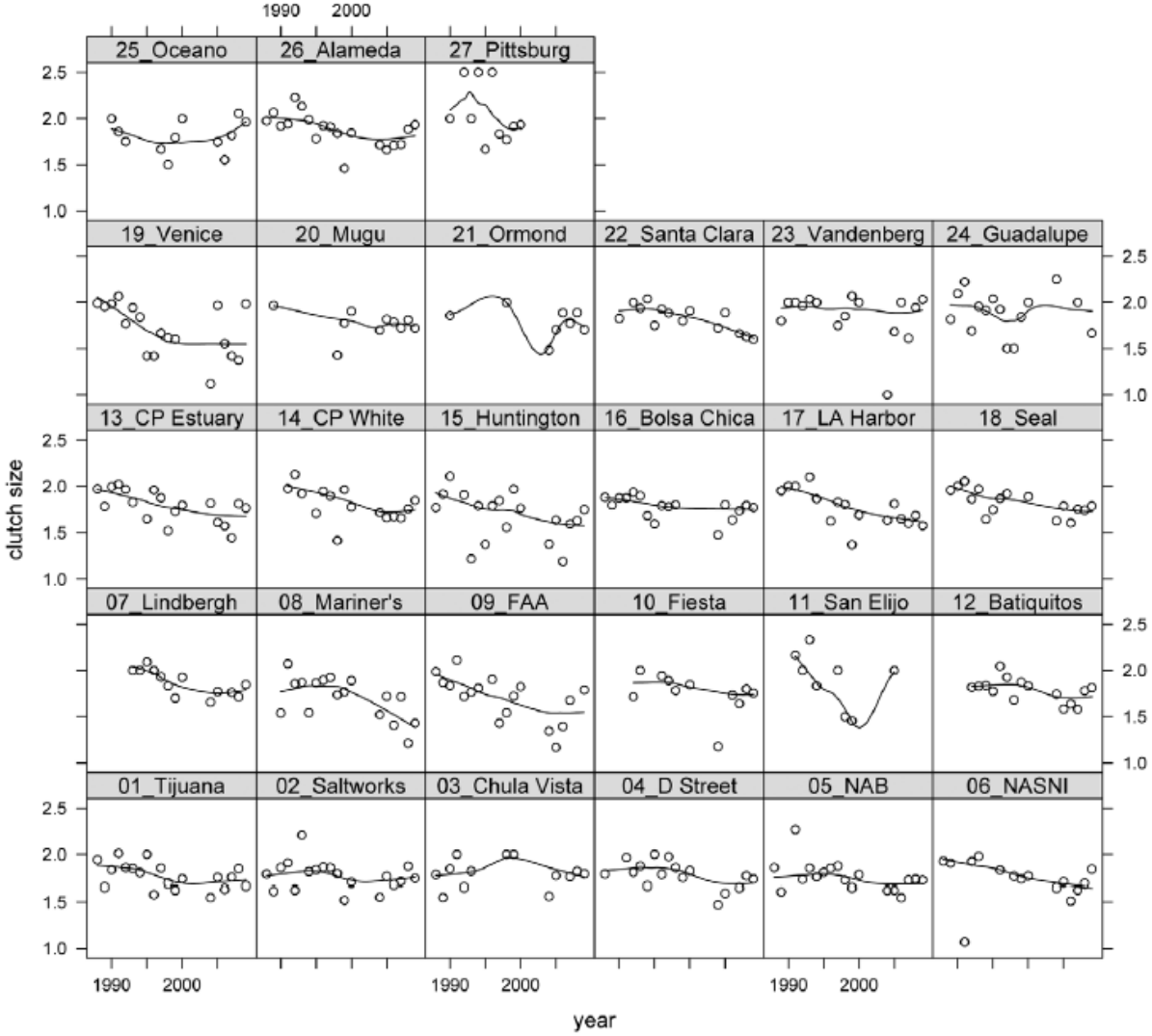


Figure 2. Plots of clutch size by year conditional on site. Sites are ordered by latitude with Tijuana Estuary being the southernmost site and Pittsburg Power Plant being the northernmost. Dots indicate clutch size estimates in a single season and lines are smoothers used to reveal patterns in the data.
doi:10.1371/journal.pone.0019489.g002

G.1.19 Shaffer et al. 2013

Nest Initiation:

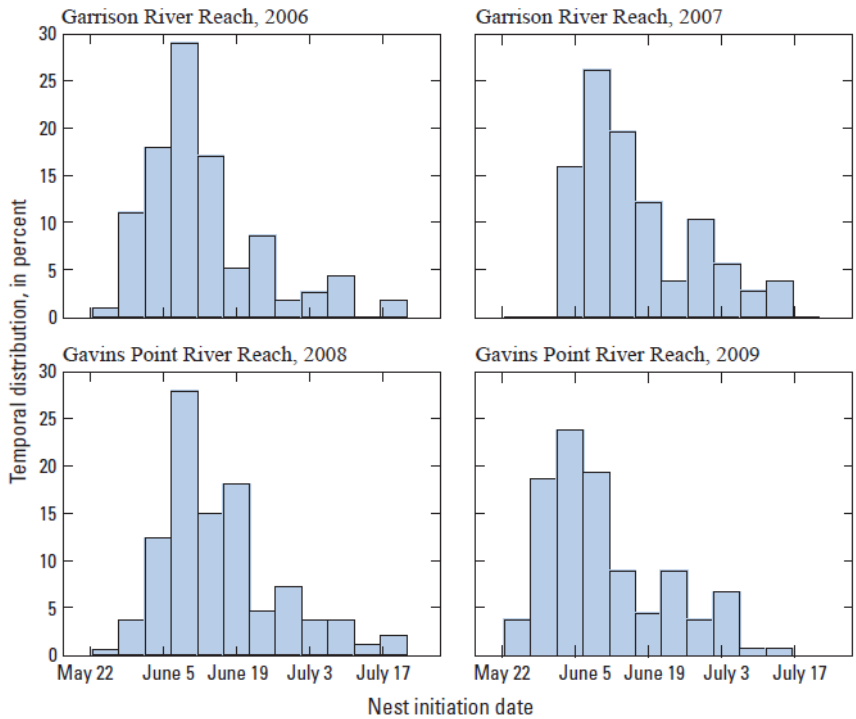


Figure 7. Temporal distribution of observed least tern nest initiations for Garrison River Reach and Gavins Point River Reach in 2006–09.

Nest Success:

Table 8. Estimates of nest success for least terns and piping plovers on the Missouri River in 2006–09, as determined from data collected by U.S. Geological Survey research crews and by U.S. Army Corps of Engineers (apparent and Mayfield nest success) monitoring crews.

[LETE, least tern; PIPL, piping plover; GRR, Garrison River Reach; GVP, Gavins Point River Reach; SAK, Lake Sakakawea; USGS, U.S. Geological Survey; Corps, U.S. Army Corps of Engineers; U.S. Geological Survey nest success estimates are from a logistic-exposure analysis (Shaffer, 2004) and are comparable to Mayfield estimates.]

Species	Study area	Year	Percent nest success				Relative bias (percent)	
			USGS estimate	USGS 95-percent confidence interval	Corps reported apparent	Corps reported Mayfield	Corps reported apparent	Corps reported Mayfield
LETE	GRR	2006	65	54–73	77	76	18	17
LETE	GRR	2007	78	68–85	89	87	14	12
LETE	GVP	2008	66	59–73	75	69	14	5
LETE	GVP	2009	73	64–80	79	76	8	4
PIPL	GRR	2006	21	16–27	54	46	157	119
PIPL	GRR	2007	47	39–54	73	65	55	38
PIPL	GVP	2008	42	35–48	67	53	60	26
PIPL	GVP	2009	41	34–49	54	46	32	12
PIPL	SAK	2007	25	17–33	41	12	64	-52
PIPL	SAK	2008	28	19–37	44	16	57	-43

Fledglings per Nest or Number of Adult Pairs:

Table 30. Estimates of U.S. Geological Survey minimum fledge ratios in relation to numbers reported by the U.S. Army Corps of Engineers 2006–09.

[BPOP, breeding population; Fledge ratio is defined as number of fledging-age chicks divided by number of breeding pairs (BPOP/2). U.S. Geological Survey minimum fledge ratios are based on resightings of banded fledging-age chicks corrected for imperfect detection. U.S. Geological Survey breeding population estimates depended on renesting probability. Because of uncertainty over renesting probability, results are shown for two scenarios, high and low renesting. Relative bias (percent) is the difference between the number reported by the U.S. Army Corps of Engineers and the U.S. Geological Survey estimate as a percentage of the U.S. Geological Survey estimate. LETE, least tern; PIPL, piping plover; GRR, Garrison River Reach; GVP, Gavins Point River Reach; SAK, Lake Sakakawea; Corps, U.S. Army Corps of Engineers; USGS, U.S. Geological Survey]

Species	Study area	Year	Reported by Corps	Low renesting			High renesting		
				USGS expanded estimate	USGS 95-percent confidence interval	Relative bias (percent)	USGS expanded estimate	USGS 95-percent confidence interval	Relative bias (percent)
LETE	GRR	2006	0.81	0.78	0.51–1.17	4	0.90	0.59–1.35	-10
LETE	GRR	2007	1.06	0.80	0.46–1.32	33	0.89	0.52–1.48	18
LETE	GVP	2008	1.14	1.23	0.98–1.50	-7	1.37	1.09–1.68	-17
LETE	GVP	2009	1.00	1.43	1.16–1.75	-30	1.61	1.30–1.97	-38
PIPL	GRR	2006	NA	NA	NA	NA	NA	NA	NA
PIPL	GRR	2007	0.97	0.96	0.46–1.77	1	1.12	0.54–2.06	-14
PIPL	GVP	2008	1.37	0.89	0.76–1.05	54	1.03	0.88–1.21	33
PIPL	GVP	2009	1.09	0.60	0.57–0.63	83	0.68	0.65–0.72	60
PIPL	SAK	2007	0.70	0.34	0.04–1.11	107	0.42	0.06–1.37	68
PIPL	SAK	2008	0.68	0.27	0.05–0.83	155	0.32	0.06–0.98	116

G.1.20 Sherfy et al. 2008

Nest Initiation – Nest Success:

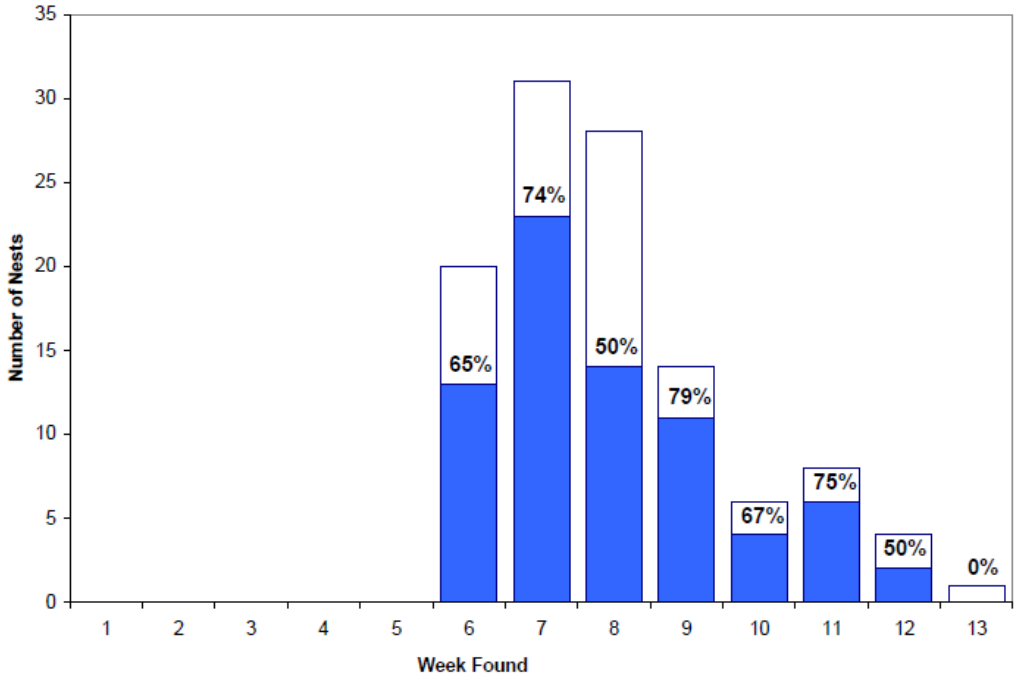


Figure 10. Number of least tern nests found on the Garrison Reach by week and the ultimate fate of those nests. Dark bars indicate number of successful (known or probable) nests and open (white) bars indicate number of unsuccessful (known or probable) nests. Percentages indicate percent of nests found during that week that ultimately produced at least 1 hatchling. Week 1 = 29 April - 5 May; Week 13 = 22 July - 28 July, 2007.

Nest Fate:

Table 19. Fate of least tern nests on the Gavins Point Reach and Lewis and Clark Lake with and without radiomarked incubating adults. Only nests surviving >7 days (the earliest incubation stage at which radios could be deployed) are included.

	Nests With One Radiomarked Adult		Nests With No Radiomarked Adult	
	# Nests	%	# Nests	%
Successful	37	52	82	50
Probable successful	23	32	40	24
Probable failure	1	1	1	1
Failure	10	14	36	22
Unknown	1	1	5	3
Total	72		164	

G.1.21 Sherfy et al. 2009a

Nest Initiation:

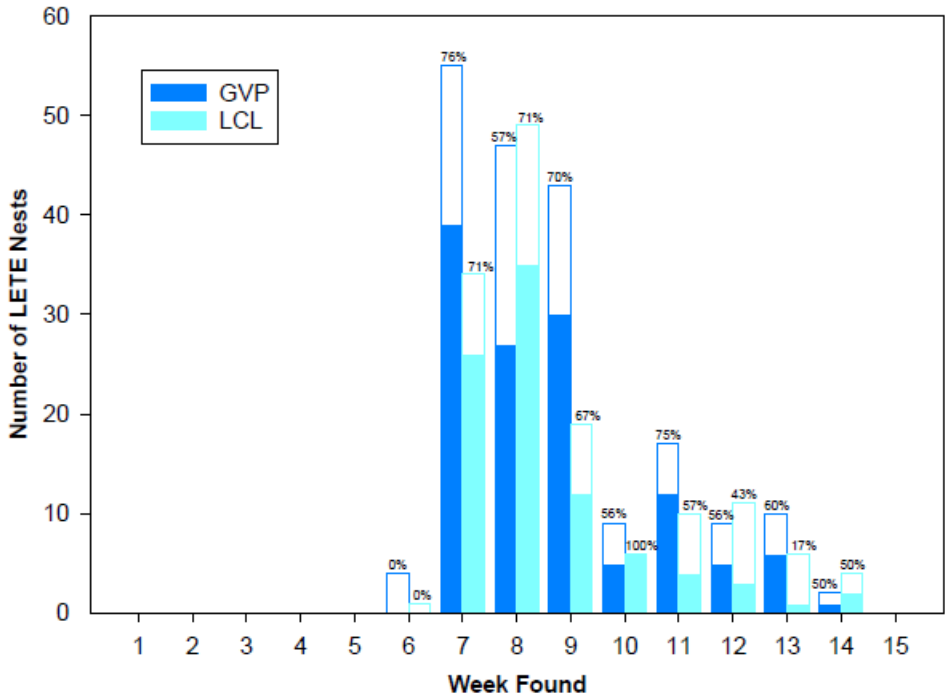


Figure 10. Number of least tern nests found on the Gavins Point Reach and Lewis and Clark Lake by week and the ultimate fate of those nests. Dark bars indicate number of successful (known or probable) nests and open (white) bars indicate number of unsuccessful (known or probable) nests. Percentages indicate percent of nests found during that week that ultimately produced at least one hatchling. Week 1 = 20 April to 26 April; Week 15 = 27 July to 2 August, 2008.

Nest Success – Nest Fate:

Table 12. Numbers and fates of least tern and piping plover nests on created and natural sandbars on the Gavins Point Reach (GVP) and Lewis and Clark Lake (LCL) in 2008.

				Nest Fate				
Species	Study Area	Sandbar Type	Total Nests	Successful	Probable Successful	Probable Failure	Failure	Unknown
Least Tern	GVP	Mechanically Created	155	80	25	0	38	12
		Natural	41	15	5	0	19	2
	LCL	Mechanically Created	140	58	38	0	31	13
Piping Plover	GVP	Both	237	94	13	0	108	22
Total			537	247	81	0	196	49

G.1.22 Sherfy et al. 2009b

Nest Initiation:

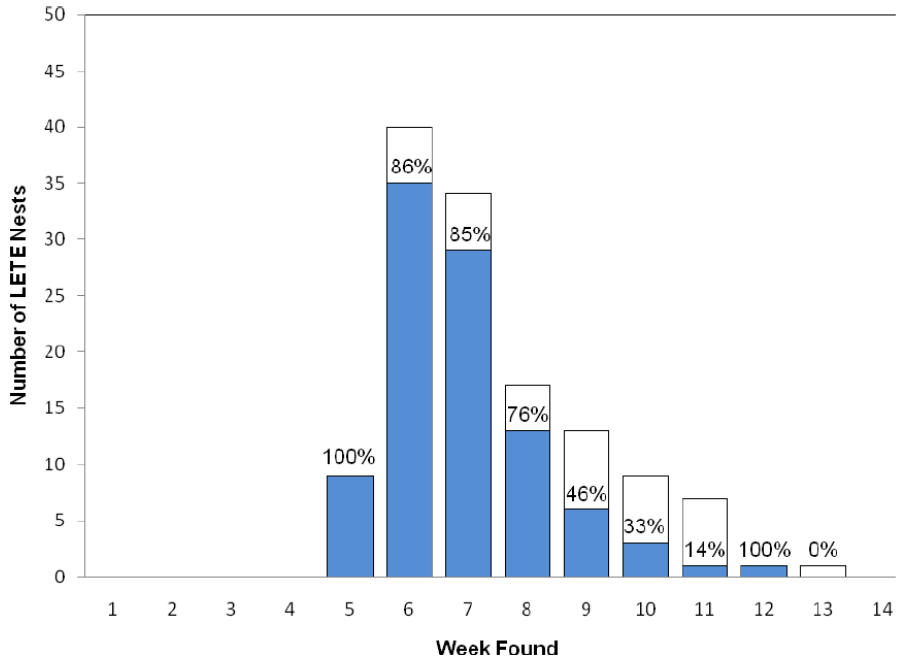


Figure 6. Number of least tern nests found on the Gavins Point Reach by week and the ultimate fate of those nests. Dark bars indicate number of successful (known or probable) nests and open (white) bars indicate number of unsuccessful (known or probable) nests. Percentages indicate percent of nests found during the week that ultimately produced at least one hatchling. Week 1 = 26 April to 2 May; Week 14 = 26 July to 1 August, 2009.

Nest Success – Nest Fate:

Table 7. Numbers and fates of least tern and piping plover nests on the Gavins Point Reach in 2009.

		Nest Fate				
	Total Nests	Successful	Probable Successful	Probable Failure	Failure	Unknown
Least Tern	136	91	6	1	33	5
Piping Plover	194	72	16	0	98	8
Total	330	163	22	1	131	13

G.1.23 Sidle and Harrison 1990

Nest Success – Nest Fate – Fledglings per Nest or Number of Adult Pairs:

Table 6. Some examples of the productivity of interior least terns.

Locations	Year	Nest Success	Fledgings per Pair	Frequency of Visits	% Population Monitored	Source
Missouri River North Dakota	1988 1989	0.62 0.56	0.42 0.21	7-10 days "	100% "	Mayer and Dryer 1989
Missouri River South Dakota	1986 1987		0.20 0.64	7-10 days "	100% "	Schwalbach 1988
Missouri River South Dakota	1988 1989	0.36 0.51	0.44 0.55	7-10 days "	100% "	Dirks 1990
Lower Platte River River Nebraska	1987 1988 1989	0.57 0.67 0.43	0.29 0.71 0.47	2-3 days " "	39% 44% 42%	Kirsch 1987-89
Cimarron River Kansas	1982-83	0.18	1.09-0.56	--	--	Schulenberg and Ptacek 1984
Salt Plains NWR, Oklahoma	1987	0.44-0.33	0.44-0.15	1-3 days		Hill 1987

1921, Hardy 1957, Stiles 1939). Thompson (1982) presented the following longevity data for coastal least terns revealed by band recoveries:

Age (years)	Percentage of Recoveries Known and Assumed Dead (N)
0-5	74 percent (58)
5-10	9 percent (7)
10-15	10 percent (8)
15-20	4 percent (3)
>20	3 percent

G.1.24 Smith and Renken 1993

Fledglings per Nest or Number of Adult Pairs:

Table 1. Estimates of reproductive parameters for interior Least Terns nesting in the Mississippi Rive valley during 1986 - 1989.

Reproductive parameter	Year			
	1986	1987	1988	1989
Nest survival rate	0.63 A[1]	0.68 A	0.64 A	0.51 B
(s.e.)	(0.029)	(0.033)	(0.033)	(0.023)
Egg survival rate	0.94 AB	0.98 A	0.88 B	0.74 C
(s.e.)	(0.023)	(0.011)	(0.024)	(0.038)
Clutch size	2.39 A	2.37 A	2.48 B	2.09 C
(s.e.)	(0.036)	(0.038)	(0.036)	(0.026)
Chick production[2]	1.01 A	1.70 B	1.68 B	0.73 C
(s.e.)	(0.102)	(0.095)	(0.079)	(0.056)
N[3]	165	189	257	375
Fledging success[4]	0.5	0.7	1.4	0.2
Total number of nests per year[5]	381	397	455	912

[1]Estimates with the same letter are not significantly different at P=0.05, using LSD tests (SAS1988).
[2]No. of chicks hatched per pair. Yearly comparisons made using Chi-square tests.
[3]No. of known-fate nests.
[4]No. of fledglings per pair.
[5]Includes all nesting attempts.

G.1.25 Stucker et al. 2013

Nest Success – Nest Fate – Fledglings per Nest or Number of Adult Pairs:

TABLE 2. Counts of Least Tern nests and surrounding-area habitat assessments on natural and constructed sandbars on the Missouri River, by year and point type (nest or surrounding area).

	Natural		Constructed		
Year	Nest	Surrounding area	Nest	Surrounding area	Nest total
2006	64	64	208	208	272
2007	52	52	212	212	264
2008	40	40	293	286	333
Total	156	156	713	706	869

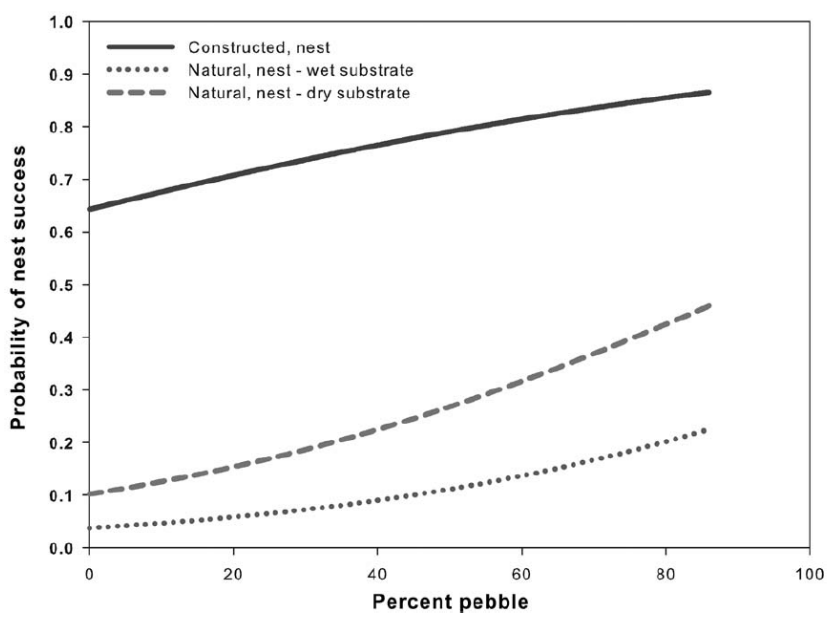


FIG. 2. Increasing percentage of pebble at Least Tern nesting sites was associated with increased nest success on constructed and natural sandbars on the Missouri River (Gavins Point Reach and Lewis and Clark Lake), 2006–2008. Substrate moisture differences in surrounding area (3 m from nest) for natural sandbar sites are indicated by dashed (dry) and dotted (wet) lines.

G.1.26 Szell and Woodrey 2003

Nest Initiation:

Table 2. Clutch initiation dates at four Least Tern colonies along the Lower Mississippi River. Data presented in Julian date.

	1995			1996			1997		
Colony	Range (N)[1]	Median	Peak[2]	Range (N)[1]	Median	Peak[2]	Range (N)[1]	Median	Peak[2]
Cow Island	176 - 201 (546)	181[a]	176 - 185	181 - 198 (272)	184[a]	181 - 186	192 - 210 (182)	199[a]	192 - 203
Porter Lake	178 - 202 (195)	183[b]	178 - 189	182 - 199 (371)	187[b]	182 - 191	149 - 208 (372[3])	187[ab]	184 - 191
St. Francis	177 - 205 (255)	187[c]	177 - 193	183 - 205 (196)	192[c]	183 - 192	150 - 206 (382[4])	187[b]	185 - 200
Kangaroo Pt.	175 - 206 (240)	179[d]	175 - 184	177 - 199 (161)	182[d]	177 - 185	143 - 187 (418)	161[c]	148 - 172
Totals	175 - 206 (1236)	182[e]	175 - 188	177 - 205 (1000)	185[e]	181 - 192	143 - 210 (1354)	185[f]	161 - 204

[1]Number of nests for which clutch initiation could be determined includes nests of known and unknown outcome.

[2]Peak clutch initiation period represents the first 75% of active clutches laid (see text).

[3]N includes 95 clutches laid 29 May 1997 which became inundated 2 June 1997. These clutches are not included in the calculation of peak initiation.

[4]N includes 136 clutches laid 30 May 1997 which became inundated 2 Jun 1997. These clutches are not included in the calculation of peak initiation.

[abcd]Median clutch initiation date varied significantly among colonies within a given year (see text).

[ef]Median clutch initiation date varied significantly among years (see text).

Nest Success – Nest Fate:

Table 3. Nest success and the percentage of nests affected by flooding (F) or predation (P) at four Least Tern colonies along the Lower Mississippi River.

	1995				1996				1997			
Colony	Number of nest	Nest Success[1]	%F	%P	Number of nest	Nest Success[1]	%F	%P	Number of nest	Nest Success[1]	%F	%P
Cow Island	550	0.99	0	0.2	265	0.03	0	95.5	172	0.94	0	1.7
Porter Lake	201	0.97	0	1.5	392	0.30	25.3	44.4	361	0.07	26.3	59.3
St. Francis	245	0.98	0	0	183	0.93	5.5	0	374	0.55	36.4	3.7
Kangaroo Pt.	219	0.89	0	5.0	159	0.64	0	25.8	346	0.38	15.0	4.6
Totals	1215	0.97[a]	0	1.2	999	0.40[b]	10.9	46.8	1253	0.42[b]	22.6	19.7

[1]Calculated as the number of successful nests divided by the total number of nests of known outcome.
[a,b]Nest success varied significantly among years (see text).

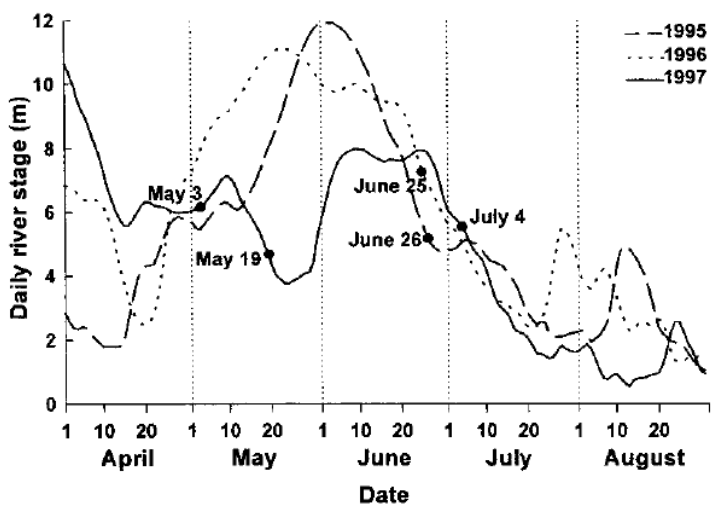


Figure 1. Daily river stages for the Mississippi River at Memphis, Tennessee for 1995-1997. Data provided by the U.S. Army Corps of Engineers, Memphis District.

Fledglings per Nest or Number of Adult Pairs:

Table 5. Fledgling estimates and reproductive success at four Least Tern colonies along the Lower Mississippi River based on mark-recapture methodology.

	1995			1996			1997		
Colony	N[1]	Bailey-modified ± SE	Reproductive success[2] (N)[3]	N	Bailey-modified ± SE	Reproductive success[2] (N)[3]	N	Bailey-modified ± SE	Reproductive success[2] (N)[3]
Cow Island	154	824 ± 162	1.50 (550)	0	0	0.00 (265)	96	150 ± 11	0.87 (172)
Porter Lake	15	155 ± 74	0.77 (201)	12	50 ± 18	0.13 (392)	0	0	0.00 (266)[4]
St. Francis	81	438 ± 69	1.79 (245)	101	226 ± 22	1.23 (183)	150	198 ± 10	0.83 (238)[4]
Kangaroo Pt.	10	65 ± 35	0.30 (219)	1	0	0.00 (159)	79	134 ± 13	0.45 (299)[4]
Totals	260	1546 ± 190	1.27 (1215)	114	276 ± 27	0.28 (999)	325	483 ± 20	0.50 (975)

[1]Number of fledglings banded during initial capture (see text).
[2]Reproductive success is estimated as the number of fledglings produced per nest based on the Bailey modified estimate of the number of fledglings.
[3]Number of nests of known outcome used to estimate reproductive success.
[4]Reproductive success does not include the nests that were inundated early in the nesting season (see text).

G.1.27 Zuria and Mellink 2002

Nest Success – Nest Fate:

TABLE 1—Characterisitcs of the 4 nesting waves of least terns, percentage of nests hatched, and causes of egg destruction at La Purinera during 1995. Values for clutch size and egg measurements are means ± standard deviations.

	1st nesting wave (mid-May)	*n*	2nd nesting wave (early June)	*n*	3rd nesting wave (early July)	*n*	4th nesting wave (late July)	*n*
Marked nests	38		244		178		37	
Clutch size	1.88 ± 0.33	25	1.52 ± 0.50	99	1.70 ± 0.46	67	1.50 ± 0.53	10
(Range)	(1–2)	25	(1–2)	99	(1–2)	67	(1–2)	10
Egg volume (cm^3)	7.52 ± 0.57	65	7.64 ± 0.55	376	7.60 ± 0.58	300	7.27 ± 0.69	52
Egg length (mm)	30.50 ± 1.22	65	30.58 ± 1.15	376	30.71 ± 1.15	300	30.32 ± 0.04	52
Egg width (mm)	22.50 ± 0.64	65	22.64 ± 0.65	376	22.54 ± 0.69	300	22.19 ± 0.58	52
% Nests hatched	2.6		0.4		2.8		—	
% Nests predated by coyotes	18.4		35.2		43.3		10.8	
% Nests predated by birds	—		—		0.6		—	
% Nests flooded	76.4		63.6		53.3		89.2	
% Nests abandoned	2.6		—		—		—	
% Nests destroyed by humans	—		0.8		—		—	

TABLE 2—Characteristics of the 2 nesting waves of least terns, percentage of nests hatched, and causes of egg destruction at Punta Banda during 1995. Values for clutch size and egg measurements are means ± standard deviations.

	1st nesting wave (20 May to 15 June)	*n*	2nd nesting wave (16 June to 16 July)	*n*
Marked nests	33		24	
Clutch size	1.82 ± 0.47	33	1.38 ± 0.50	24
(Range)	(1–3)	33	(1–2)	24
Egg volume (cm^3)	7.60 ± 0.50	60	7.38 ± 0.60	33
Egg length (mm)	30.83 ± 1.19	60	31.10 ± 1.35	33
Egg width (mm)	22.50 ± 0.64	60	22.07 ± 0.76	33
% Nests hatched	63.6		12.5	
% Nests predated	—		4.2	
% Nests flooded	—		—	
% Nests abandoned	18.2		16.7	
% Nests destroyed by humans	24.3		66.6	

G.1.28 Dirks 1990

Nest Success – Nest Fate:

Table 3. Least tern nest fates along the Missouri River in South Dakota during 1988 and 1989.

	Gavins Pt.		Ft. Randall		Total 1988		Total 1989	
	1988	1989	1988	1989	N	(%)	N	(%)
Nests with Known Fate	199	170	13	27	212	(100)	197	(100)
No. Hatched	74	91	2	10	76	(36)	101	(51)
No. Destroyed or Abandoned	125	79	11	17	136	(64)	96	(49)
Cause of Destruction/Abandonment								
Predator	29	42	0	1	29	(21)	43	(45)
Flooding	0	1	11	14	11	(8)	15	(16)
Human Disturbance	7	1	0	0	7	(5)	1	(1)
Weather	11	7	0	0	11	(8)	7	(7)
Sandbar Erosion	3	3	0	0	3	(2)	3	(3)
Unknown Agent	75	25	0	2	75	(55)	27	(28)
Fate Unknown	49	11	7	4	56	(21)	15	(7)
Total No. Nests Found	248	181	20	31	268	(100)	212	(100)

G.2 Prey

G.2.1 Schweitzer and Leslie 1996

Findings:

TABLE 1—Acceptance[1] of fish in different size classes by different-aged least tern chicks. A dash means that no fish of the size class were offered to the indicated chick(s).

	Size class of fish (cm)[2]				
Age of chicks (days)	≤1.3 (n = 9)	1.4–2.0 (n = 6)	2.1–2.6 (n = 17)	2.7–3.0 (n = 1)	>3.1[3] (n = 5)
1 (n = 7)[4]	100%	100%	50%	0%	—
2 (n = 7)	100%	100%	71%	—	—
3 (n = 4)	100%	100%	100%	—	—
10 (n = 2)	—	—	100%	—	—
12 (n = 1)	—	—	100%	—	—

[1] Expressed as a percentage: number of fish consumed ÷ number of fish offered.
[2] Size (length) of fish estimated relative to bill length of adults.
[3] All fish in this size class brought to nests were consumed by an adult.
[4] Number of chicks.

TABLE 2—Number and success (%) of least terns foraging off the salt flat during different periods of the breeding season.

Period	Location[2]	Number of birds	Number of dives	Number of captures	Success (no. captures ÷ no. dives)	Overall success[1] $\bar{x}$	Overall success[1] SE
Arrival	A	0	—	—	—		
	B	18	44	5	11.4	10.4	1.0
	C	10	85	8	9.4		
Early	A	15	50	2	4.0		
	B	8	8	4	50.0	21.1	14.5
	C	15	32	3	9.4		
Middle	A	0	—	—	—		
	B	0	—	—	—	—	
	C	0	—	—	—		
Late	A	54	70	19	27.1		
	B	1	2	0	0.0	13.6	13.6
	C	0	—	—	—		

[1] Overall means were not different among periods of the breeding season (Kruskal-Wallis Test, P = 0.8984).
[2] Fixed-point observation stations were, A: Great Salt Plains Reservoir shoreline east of the salt flat (7 stations); B: Byron State Fish Hatchery (4 stations); and C: Salt Plains National Wildlife Refuge ponds (4 stations).

TABLE 4—Length and body depth[1] (cm) of fish collected in colony sites and waterways upstream of the salt flat. Common names are given in Table 3.

	Colony sites on the salt flat					Upstream of the salt flat				
		Length		Depth			Length		Depth	
Scientific name	n	$\bar{x}$	SE	$\bar{x}$	SE	n	$\bar{x}$	SE	$\bar{x}$	SE
Cyprinella lutrensis	3	5.9	0.6	1.6	0.2	10	4.3	0.6	1.1	0.2
Cyprinus carpio	5	5.9	0.6	1.8	0.2	4	5.6	1.2	1.5	0.4
Dorosoma cepedianum	10	6.4	0.2	1.9	0.1	9	6.4	0.1	1.8	0.0
Fundulus zebrinus	2	6.9	0.1	1.2	0.2	10	3.1	0.3	0.5	0.1
Gambusia affinis	0	—	—	—	—	10	2.5	0.1	0.5	0.0
Hybognathus placitus	1	9.0	—	1.1	—	10	2.4	0.1	0.5	0.0
Ictalurus punctatus	0	—	—	—	—	10	2.8	0.2	0.4	0.0
Menidia beryllina	5	7.7	0.6	1.2	0.1	10	4.4	0.5	0.6	0.1
Notropis atherinoides	0	—	—	—	—	10	4.5	0.6	1.0	0.2
Notropis stramineus	0	—	—	—	—	11	3.1	0.3	0.6	0.1
Pimephales promelas	0	—	—	—	—	11	3.1	0.2	0.6	0.0

[1] Measured at widest point.

G.2.2 Tibbs and Galat 1998

Findings:

Table II. Mean fish catch (number per 100 $m^3 \pm 1$ S.D.) and number of replicates for all season–habitat combinations, years combined, within six aquatic habitats in the lower Mississippi River, Missouri, river km 1535–1334, during 1993 and 1994

Habitat category	Aquatic habitat	Season			
		Incubation		Rearing	
		Fish catch	Replicates	Fish catch	Replicates
Deep-water	Main channel	17.8 ± 14.9 A A	7	5.9 ± 5.3 A A	6
	Secondary channel	42.5 ± 47.3 AB A	5	19.9 ± 43.4 A B	6
	Side channel	11.0 ± 11.2 A A	8	5.9 ± 6.4 A A	6
	Connected slough	121.7 ± 118.7 BC A	5	35.0 ± 79.6 A B	6
Shallow-water	Main-channel interface	192.4 ± 246.9 C A	10	906.4 ± 1518.1 B A	7
	Side-channel interface	830.3 ± 952.3 D A	9	800.1 ± 678.4 B A	7

Means within a column with the same letter on the right are not different ($p > 0.05$) when ranks are compared. Means within a row with the same letter beneath are not different ($p > 0.05$) when ranks are compared

Table III. Mean taxa richness of collected fish (± 1 S.D.) and number of replicates within six aquatic habitats, years, and seasons combined, in the lower Mississippi River, Missouri, river km 1535–1334, during 1993 and 1994

Habitat category	Aquatic habitat	Total richness	Replicates
Deep-water	Main channel	4.6 ± 2.0 A	13
	Secondary channel	4.4 ± 3.1 AB	11
	Side channel	3.9 ± 1.9 A	14
	Connected slough	2.6 ± 1.4 B	11
Shallow-water	Main-channel interface	6.8 ± 2.9 C	17
	Side-channel interface	8.3 ± 3.2 C	16

Means within a column with the same letter on the right are not different ($p < 0.05$) when ranks are compared.

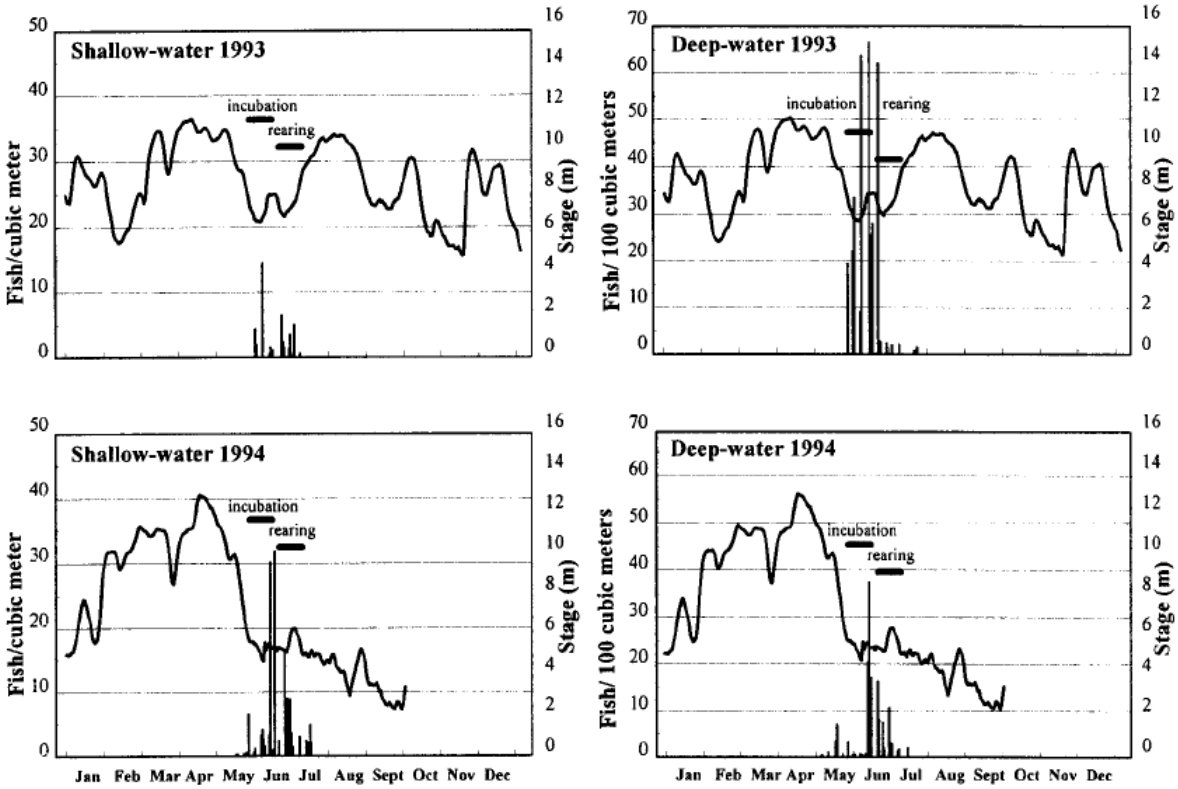


Figure 2. Temporal associations between forage-fish availability (vertical bars), least tern reproductive seasons (horizontal bars), and river stage (solid line) in the lower Mississippi River, Missouri, river km 1535–1334, during 1993 and 1994. Catch is reported as fish m^{-3} in shallow-water habitats and as fish 100 m^{-3} in deep-water habitats

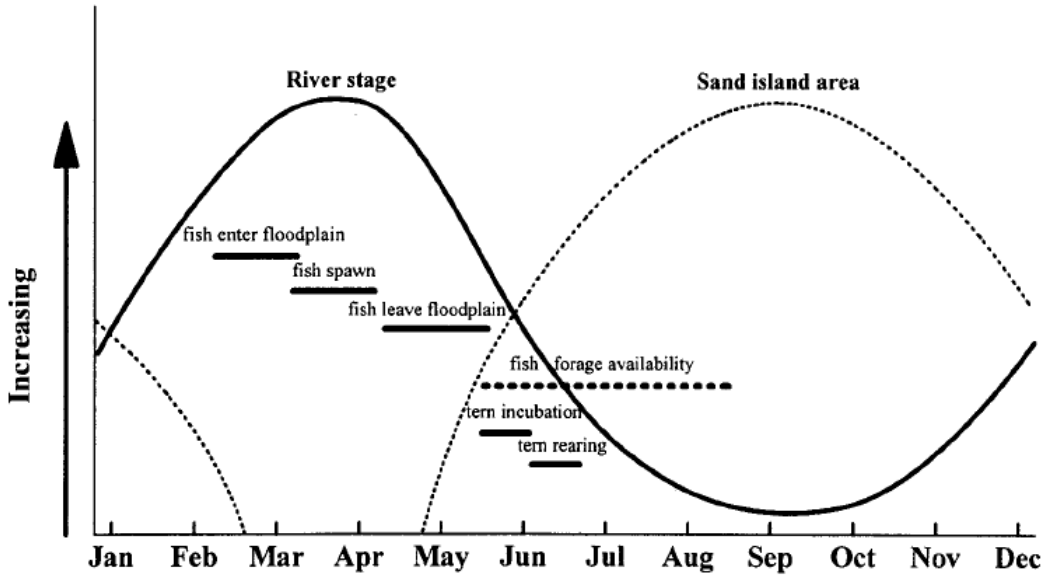


Figure 3. Conceptual model of temporal linkages among river stage, least tern reproduction, sand island area, and forage-fish availability in the lower Mississippi River, Missouri

G.2.3 Elliott et al. 2007

Findings:

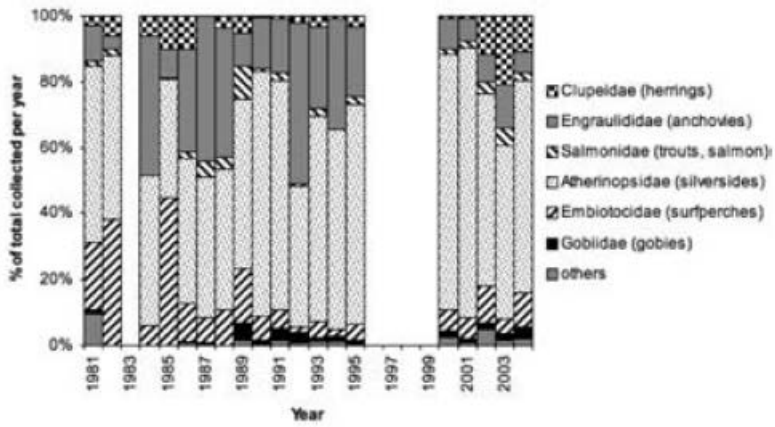


Figure 5. Least Tern dropped prey collected at Alameda Point from 1981-1982, 1984-1995, and 2000-2004. (Years 1981-1995 from Collins (1995).)

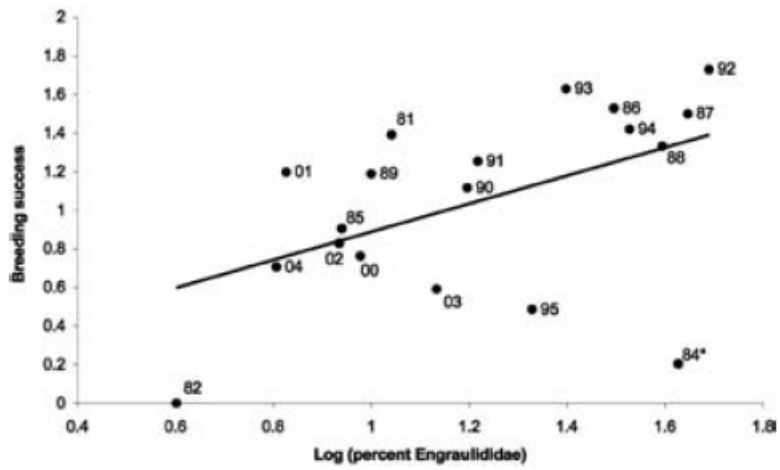


Figure 6. Least Tern breeding success and the percentage of Engraulididae in the dropped prey. (Each point is labeled with the two-digit year. * = possible outlier.)

G.2.4 Stucker et al. 2012

Findings:

Table V. Measures of fish species and relative abundance during fish sampling in areas of shallow-water habitat near emergent sandbars of the Missouri River (RKM 1213–RKM 1331), 2006–2008

	Mechanically created	River created	$F_{1,16}$	p
Species	3.3 (0.5)	3.4 (0.3)	0.03	0.86
Count of fish[a]	2.1 (0.3) [7.0]	2.2 (0.2) [8.5]	0.31	0.59
Count of shiners[a]	1.5 (0.3) [3.7]	1.7 (0.2) [4.6]	0.32	0.58
Count of size class 0–1	0.8 (0.2) [1.2]	0.9 (0.1) [1.6]	0.67	0.43
Count of size class 2[a]	1.1 (0.2) [1.9]	1.1 (0.1) [1.9]	0	0.95
Count of size class 3[a]	0.7 (0.2) [1.0]	0.8 (0.1) [1.2]	0.19	0.67
Count of size class 4[a]	1.2 (0.3) [2.3]	1.4 (0.2) [3.2]	0.59	0.45

Results from least squares means (standard errors) [back-transformed means] are presented by emergent sandbar type, with associated F-test and level of significance (p).
[a]Least square means presented are the means at the average of the included covariate, depth, 0.61 m.

Table II. Fish size class, fish body length (TL) and relationship with least tern bill length

Size Class	Fish length (TL, mm)	Relationship to least tern bill length
0	<20	<1 bill
1	21–33	~1 bill
2	34–46	>1 bill
3	47–52	2 bills
4	>52	>2 bills

Least tern bill length is estimated as 26 mm (Thompson *et al.*, 1997).

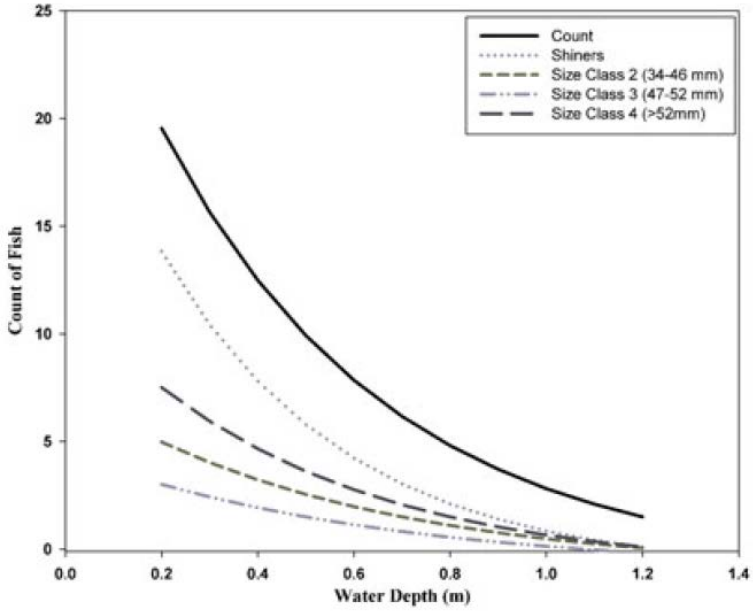


Figure 3. Increasing depths were associated with reduced counts of captured fish. Significant relationships between average water depth of sample sites and measures of fish relative abundance for sampled areas of the Missouri River (RKM 1213–RKM 1331) in 2006–2008.

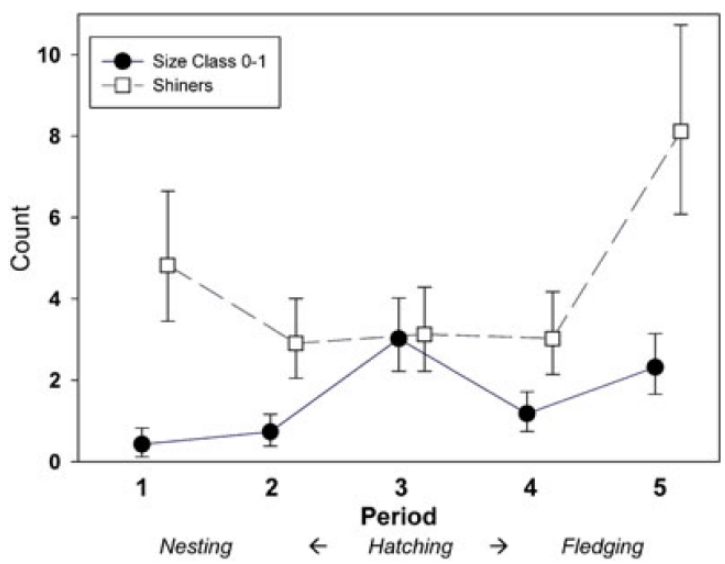


Figure 5. Significant intra-seasonal variability in fish relative abundance was observed for fish size class 0–1 and shiners ($p < 0.05$). Values of fish relative abundance were obtained from back-transformed least squares means and standard errors within sampled areas of the Missouri River (RKM 1213–RKM 1331) in 2006–2008. *X*-axis indicates periods sampled and the relationship to least tern nesting chronology.

G.3 Nest Site Selection

G.3.1 Leslie et al. 1997

Findings:

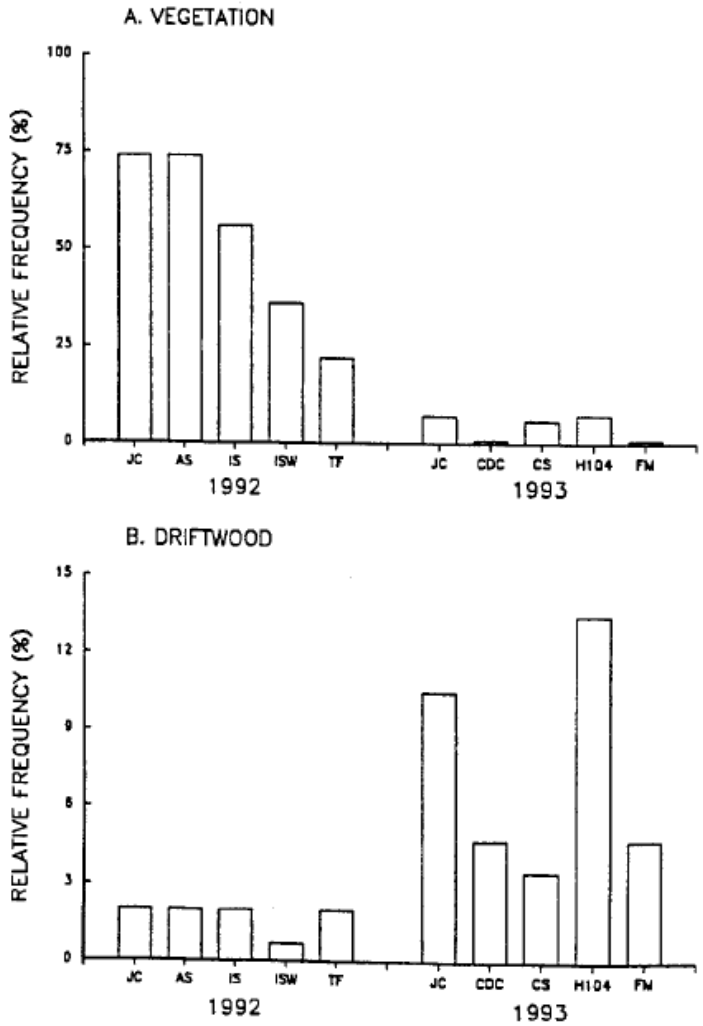


Figure 1. Frequency of occurrence of vegetation and driftwood in 75–200 0.1-m^2 plots at 5 colony sites of least terns on the Arkansas River from Tulsa to Muskogee, Oklahoma, prior to (summer 1992) and after (summer 1993) scouring floods. Colony abbreviations: JC, Joe Creek; AS, Anchor Stone; IS, Indian Springs; ISW, Indian Springs West; TF, Taft; CDC, Cedar Creek; CS, Coweta; H104, Highway 104; FM, Fern Mountain.

Table 2. Mean distance (m) from nests (*N*) to the narrowest and widest water barriers from the shores of sandbars to the river bank at 5 least tern colonies on the Arkansas River, 1992–1993.

Year	Colony	*N*	Distance to narrowest water barrier		Distance to widest water barrier		P[a]
			$\bar{x}$	SE	$\bar{x}$	SE	
1992	Anchor Stone	6	184.2	5.42	40.3	5.16	<0.001
	Joe Creek	8	104.3	3.72	35.4	5.18	<0.001
	Taft	10	31.7	4.61	38.5	6.00	0.457
1993	Joe Creek	28	70.3	2.61	57.3	4.78	<0.021
	Cedar Creek	9	142.4	6.39	30.5	5.68	<0.001
	Coweta	8	103.7	3.82	35.8	6.70	<0.001
	Hwy. 104	21	102.4	3.81	33.0	3.34	<0.001
	Fern Mtn.	14	162.3	13.00	48.7	8.83	<0.001

a. Intracolony ranked *t*-tests.

G.3.2 Marcus et al. 2007

Findings:

Table 3. Substrate composition and vegetation cover at Least Tern (LT) or Piping Plover (PP) nests (representing what the birds selected) and at random points within used plots (representing what was available to them in the area around their nest) at gravel mines along the Platte and Elkhorn rivers, Nebraska, 2000 and 2001. A used plot is defined as a plot with at least one nest initiated in it. Values with the same letter within a row are not significantly different (P > 0.05).

		Attractant plots		Control plots	
		Nests	Random	Nests	Random
Fine Sand	LT	16.8 A	13.4 AB	16.1 A	8.5 B
	PP	18.7 A	17.2 A	19.8 A	12.5 B
Coarse Sand	LT	56.5 A	53.5 A	64.9 A	85.4 B
	PP	43.2 A	54.7 A	43.3 A	79.2 B
Small Gravel	LT	18.9 A	23.3 A	9.0 AB	3.0 B
	PP	28.4 A	20.4 A	29.6 A	17.6 A
Large Gravel	LT	7.9 A	10.0 A	9.5 A	3.0 B
	PP	10.0 A	6.2 A	11.6 A	1.6 A
Vegetation	LT	2.2 A	3.1 A	1.5 A	2.0 A
	PP	4.6 A	1.7 A	3.2 A	3.0 A

G.3.3 Schweitzer and Leslie 1999

Findings:

TABLE 1.—Textures and colors of soils associated with active (≥3 nests) and inactive (≤2 nests) colony sites of least terns and distances from nests or paired random points to nearest habitat features at Salt Plains National Wildlife Refuge, Oklahoma

Characteristic	Active colony sites			Inactive colony sites (soil) or random points (distances)		
	n[a]	Median	Range	n[a]	Median	Range
Colony sites (1992–1993 combined)						
Soil texture[b]	12	SL	SC–S	4	SCL	SC–S
Soil color[c]						
Hue (5y–7.5y)	12	5y	5y–7.5y	4	5y	5y–7.5y
Value (scale 2–7)	12	4.0	3.5–4.5	4	3.7	3.0–4.0
Chroma (scale 0–6)	12	3.5	2.7–3.8	4	3.3	2.4–3.9
Nests and random points						
Distance to water (m)						
1992	69	>100[d]	84.9–>100	69	>100	84.2–>100
1993	77	277.2	90.8–599.1	77	277.2	91.5–599.5
Distance to vegetation (m)						
1992	69	89.2	21.4–>100	69	89.8	21.2–>100
1993	77	130.5	18.2–543.8	77	130.4	18.0–544.9
Distance to driftwood/debris (m)[e]						
1992	69	0.45	0.0–9.2	69	3.95	1.8–15.8
1993	77	2.35	0.0–7.0	77	3.25	1.5–4.9
Internest distance (m)[f]						
1992	69	80.9	66.4–>100			
1993	77	59.2	21.4–90.8			

[a] Number of colony sites for colony site characteristics and number of nests and random points for nest and random point characteristics

[b] Soil texture class abbreviations: SL = sandy loam; SC = sandy clay; SCL = sandy clay loam; S = sandy (Brady, 1974:47)

[c] Estimates from Munsell Soil Color Chart; hue indicates relation to red, yellow, green, blue and purple; value indicates lightness; chroma indicates strength or departure from a neutral of the same lightness

[d] During 1992, distances were not measured beyond 100 m and were grouped as >100 m for ranked tests

[e] Nests were closer to driftwood/debris than were random points in 1992 and 1993; t-tests of ranked data

[f] Mean internest distance was greater ($P < 0.01$) in 1992 than 1993; t-tests of ranked data; no internest distances from inactive colony sites

G.3.4 Sherfy et al. 2012

Findings:

Table 2. Arithmetic means and 95% confidence limits for habitat variables at least tern nests and random points on natural and constructed sandbars on the Missouri River, 2006–2008.

	Constructed sandbars				Natural sandbars			
	Nest sites		Random points		Nest sites		Random points	
	Mean	95% CL	Mean	95% CL	Mean	95% CL	Mean	95% CL
WOODSTEM	1.3	0.9–1.7	1.1	0.8–1.2	4.9	3.1–6.7	3.4	2.6–4.1
WOODCOV	0.5	0.4–0.5	1.5	1.1–1.8	1.6	1.0–2.1	5.1	3.8–6.3
TERR_HERB	0.4	0.3–0.5	1.7	1.2–2.0	1.9	0.8–3.0	5.4	4.4–6.4
WET_HERB	0.3	0.1–0.3	1.1	0.7–1.4	0.4	0.1–0.5	2.6	1.9–3.2
MAX_VEG_HT	0.2	0.1–0.2	0.3	0.2–0.3	0.4	0.2–0.4	0.6	0.5–0.6
MEAN_VEG_HT	0.2	0.1–0.2	0.2	0.2–0.2	0.3	0.2–0.3	0.5	0.4–0.5
SAND	72.1	70.4–73.7	64.2	62.1–66.1	74.6	70.8–78.3	59.6	55.7–63.4
SILT	6.1	4.7–7.4	16.8	14.7–18.7	1.6	−0.5–3.6	24.9	20.7–29.0
PEBBLE	11.9	10.7–12.9	11.4	10.0–12.6	14.1	10.2–17.9	9.7	7.0–12.2
GRAVEL	2.2	1.9–2.4	1.2	0.9–1.4	0.4	0.2–0.6	0.5	0.1–0.7
LEAF	0.2	0.1–0.3	1.3	0.9–1.7	0.1	−0.1–0.1	4.1	2.8–5.3
SM_DEBRIS	2.8	2.4–3.2	2.2	1.8–2.6	2.7	2.0–3.2	5.1	3.8–6.2
LG_DEBRIS	2.6	2.1–2.9	1.4	0.9–1.7	2.9	1.9–3.9	1.6	0.5–2.5

G.3.5 Sidle and Kirsch 1993

Findings:

Table 2. Numbers of adult Least Terns (LT) and Piping Plovers (PP), percentages of all individual birds using sand pits, and the number of sandbar and sand pit nesting sites on the central and lower reaches of the Platte River, 1988-1991. Data were derived from Lingle (1993a), and Nebraska Game and Parks Commission (1988, 1989, and unpublished data), and Sidle *et al.* 1991.

	Central Platte						Lower Platte					
	Number of birds		% birds on pits		No. nesting sites		Number of birds		% birds on pits		No. nesting sites	
Year	LT	PP	LT	PP	Bar	Pit	LT	PP	LT	PP	Bar	Pit
1988	124	96	61	62	6	16	497	161	26	19	24	11
1989	194	94	94	64	6	16	409	188	14	13	38	10
1990	176	74	91	78	3	17	361	149	40	35	23	13
1991	158	58	87	76	3	14	295	73	30	37	35	11
Mean	163.0	80.5	83.2	70.0	4	15.8	390.5	142.8	27.5	26.0	30.0	11.2
SD	29.9	18.0	15.1	8.2	1.7	1.3	85.0	42.3	10.8	11.8	7.6	1.3

G.3.6 Carreker 1985

Findings:

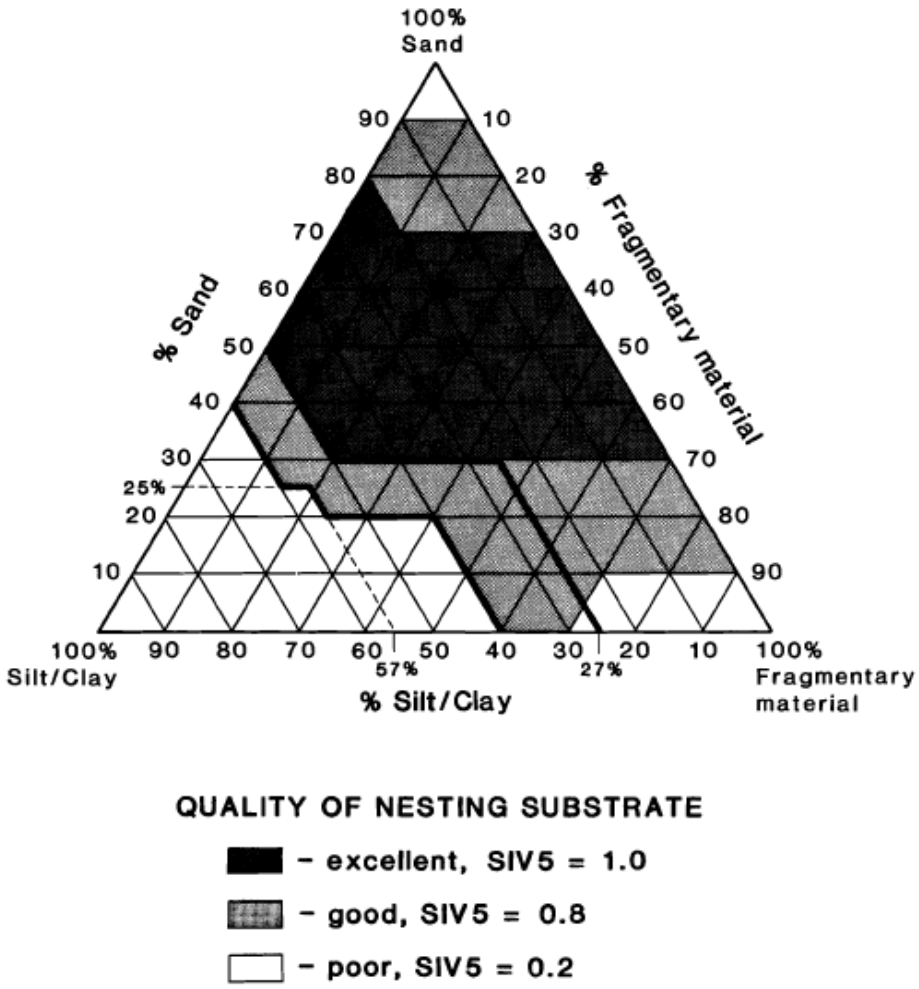


Figure 6. The relationship between substrate composition and the suitability index value for least tern reproduction.

G.3.7 Conway et al. 2003

Findings:

Table 2. Habitat variables between interior least tern nests and random sites in the Prairie Dog Town Fork of the Red River, Childress County, Texas, 1998.

Habitat variable [1]	Nests (n = 20) ($\bar{x}$)	Nests (n = 20) *SE*	Random (n = 20) ($\bar{x}$)	Random (n = 20) *SE*	*F*	*P*
Distance to upland (m)	285.7	28.9	257.1	24.5	0.64	0.43
Distance to vegetation (m)	10.9	2.1	10.5	2.7	0.09	0.77
Distance to mudflat (m)	106.1	25.6	97.2	26.8	0.01	0.97
Distance to water (m)	198.3	35.2	195.5	34.7	0.01	0.95
Vertical cover [2]	1.0	0	1.0	0	0.00	1.0
Relative elevation [3]	14.2	2.9	14.9	2.8	0.01	0.94

1. Means followed by the same letter in the same row did not vary ($P > 0.05$).
2. Vertical cover measure using Robel pole (Robel et al. 1970) from each of the cardinal directions, 4 m from the nest and/or random point.
3. Relative elevation measured as the difference between the nest/random sites and the mean elevation of the river bottom generated from random transects.

Table 3. Habitat variables between successful and non-successful interior least tern nests in the Prairie Dog Town Fork of the Red River, Childress County, Texas, 1998.

Habitat variable [1]	Success (n = 13) ($\bar{x}$)	Success (n = 13) *SE*	Non-success (n = 7) ($\bar{x}$)	Non-success (n = 7) *SE*	*F*	*P*
Distance to upland (m)	263.2	34.3	341.7	50.7	1.03	0.322
Distance to vegetation (m)	11.9	2.76	8.23	2.46	0.52	0.48
Distance to mudflat (m)	113.1	27.4	88.8	61.8	0.03	0.87
Distance to water (m)	196.4	45.1	203.0	55.2	0.05	0.83
Vertical cover [2]	1.0	0	1.0	0	0.00	1.0
Relative elevation [3]	13.7	3.97	15.5	2.12	0.34	0.57

1. Means followed by the same letter in the same row did not vary ($P > 0.05$).
2. Vertical cover measure using Robel pole (Robel et al. 1970) from each of the cardinal directions, 4 m from the nest and/or random point.
3. Relative elevation measured as the difference between the nest/random sites and the mean elevation of the river bottom generated from random transects.

G.3.8 Jenniges and Plettner 2008

Findings:

Table 2. Number of Least Tern nests at sites monitored along the central Platte River 1991 to 2005.

Year	Managed Sandpits	Unmanaged Sandpits	Islands	Total
1991	7	NA	1	8
1992	31	NA	0	31
1993	29	NA	4	33
1994	34	46	0	80
1995	36	30	14	80
1996	30	19	13	62
1997	34	30	10	77'
1998	31	NA	4	35
1999	24	NA	3	27
2000	21	NA	0	21
2001	25	NA	0	25
2002	37	NA	0	37
2003	47	NA	0	47
2004	43	NA	0	43
2005	44	NA	0	44
Total	473	125	49	647

G.3.9 Kirsch 1996

Findings:

Table 2. Summary statistics for habitat variables derived from aerial videos of river habitat along the lower Platte River, Nebraska, 1989–90. Pairs of values from used and unused sites that differ significantly (ANOVA) are indicated with asterisks.

Nesting period and year	Date of video	Date of site-use survey	River flow m^3/sec[a]	Type of site	n	Habitat variables: Channel width (m) $\bar{x}$	Channel width (m) SE	Mean size sand areas (m^2) $\bar{x}$	Mean size sand areas (m^2) SE	Total area of sand (m^2) $\bar{x}$	Total area of sand (m^2) SE
Initial 1989	3 Jun	9–13 Jun	69.3	Used	35	424	21	71*	8	12,348**	152
				Unused	30	394	168	45	6	7,070	839
Renesting 1989	29 Jun	20–22 Jun	328.3	Used	16	452	19	48	10	1,553	553
				Unused	33	394	14	32	9	785	250
Initial 1990	11 Jun	12–14 Jun	145.5	Used	16	474	17	538*	274	3,484**	599
				Unused	33	445	15	33	9	1,363	457
Renesting 1990	4 Jul	2–3 Jul	57.2	Used	22	489**	12	510*	91	25,477	3,468
				Unused	29	428	16	308	54	1,100	1,759

[a] Mean daily flow in cubic meters per second at the North Bend gauge river kilometer 116.8 (river mile 72.4) (Boohar et al. 1989, 1990).
* Differences between used and unused within nesting periods, $P < 0.05$.
** Differences between used and unused within nesting periods, $P < 0.01$.

Table 3. Summary statistics for habitat variables (1,000 m^2) derived from aerial videos of sandpit sites along the lower Platte River, Nebraska, 1989–90. Pairs of values from used and unused sites that differ significantly (ANOVA) are indicated with asterisks.

Nesting period and year	Date of video[a]	Date of site-use survey	Type of site	n	Habitat variables: Area of usable sand[b] $\bar{x}$	Area of usable sand[b] SE	Area of unusable sand[c] $\bar{x}$	Area of unusable sand[c] SE	Area of water $\bar{x}$	Area of water SE	Area of sandpit site $\bar{x}$	Area of sandpit site SE
Initial 1989	4 Jul	9–13 Jun	Used	6	23.4**	6.1	14.7	6.4	87.6	35.2	158.3	57.2
			Unused	33	11.9	1.4	6.7	1.5	51.1	11.2	89.2	14.2
Renesting 1989	4 Jul	20–22 Jun	Used	7	17.9	4.7	7.3	1.9	104.9*	44.0	145.4	48.9
			Unused	32	12.8	1.7	8.1	2.0	46.2	8.9	89.9	14.8
Initial 1990	17 Jun	12–14 Jun	Used	12	31.4	4.5	3.6	12.4	135.8*	25.5	243.3**	26.9
			Unused	23	22.1	7.7	22.7	4.8	66.5	13.8	131.2	19.7
Renesting 1990	17 Jun	2–3 Jul	Used	12	31.5	4.6	28.9	9.0	125.7	26.1	229.9*	24.8
			Unused	23	22.1	7.7	26.7	6.6	71.8	14.5	138.2	21.9

[a] A single video was actually taken of sandpits in each year because sandpit habitat changes little during the breeding season.
[b] Sand that was not disturbed by equipment or humans or covered with slurry run-off.
[c] Sand that was frequently disturbed by equipment and humans or covered by slurry run-off.
* Differences between used and unused within nesting periods, $P < 0.05$.
** Differences between used and unused within nesting periods, $P < 0.01$.

Table 10. Numbers of nesting sites and numbers of least terns counted on river and sandpit habitats between river kilometers 164.5 and 0 (river miles 102 and 0) of the lower Platte River, Nebraska, 1984–90. Data for 1984–86 were obtained from Nebraska Game and Parks Commission (unpubl. data[a]). Data for 1987–90 include Game and Parks Commission data as well as those collected in the present study.

	River		Sandpits	
Year	No. sites	No. pairs terns	No. sites	No. pairs terns
1984	0	0	10	137–144
1985	12	77	7	35
1986	6	150	8	77
1987	27	214	15	105
1988	24	256	11	118
1989	38	240	10	64
1990 early[b]	17	155	9	122
1990 renest[b]	23	144	11	74

[a] J. J. Dinan, Platte River interior least tern and piping plover nesting survey, Unpubl. reports to Nebr. Game and Parks Comm., Lincoln, 1984, 1985, 1986.

[b] Flooding inundated all sandbars during 14–19 June (Boohar et al. 1990). These 2 rows indicate the numbers counted before the flood and about 10 days after the flood had receded (20 Jun) when terns had begun to renest.

G.3.10 Stucker 2012

Findings:

Table 4. Count of Least Tern nests and surrounding area habitat assessments included in study by point type (nest, surrounding area) on natural and constructed sandbars, for year of the study.

	Natural		Constructed		
Year	Nest	Surrounding Area	Nest	Surrounding Area	Nest Total
2006	64	64	208	208	272
2007	52	52	212	212	264
2008	40	40	293	286	333
Total	156	156	713	706	869

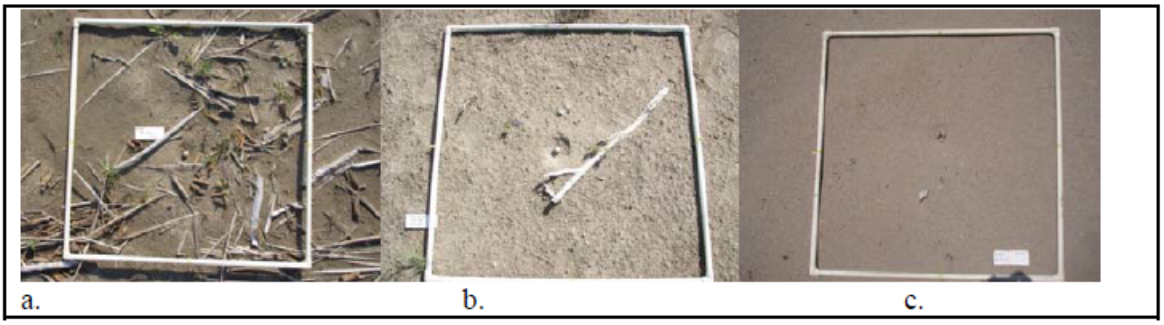


Figure. 2. Photographs demonstrating range of habitat variability at Least Tern nests on constructed sandbars at time of discovery. Each nest is surrounded by a 1-m2 quadrat frame. a). Nest in wrack line, with abundant small debris, nest furniture, and limited vegetation on wet sand and silt substrate (RM 827); b) Nest with nest furniture, small debris, in contact with the nest bowl, consolidated moist sand substrate with surficial pebbles (RM 770); c) Dry, unconsolidated sand substrate (no vegetation, no debris) (RM 755).

G.4 Nest Density

G.4.1 Akcakaya et al. 2003

Findings:

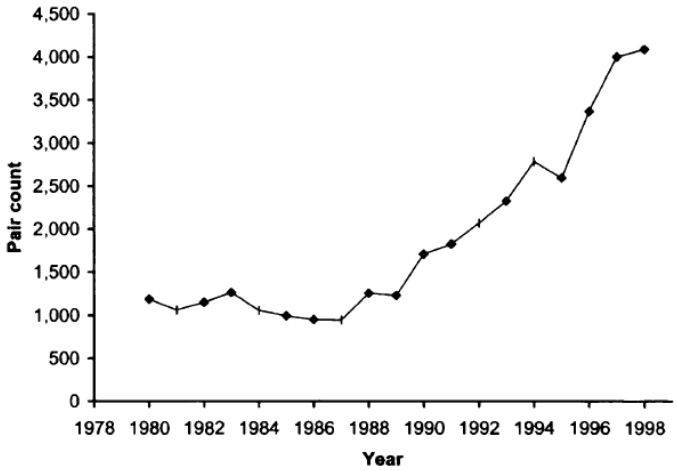


Fig. 2. Total count of California least tern pairs from 1980 to 1998 (from Keane 2001) in California, USA, and Baja California, Mexico.

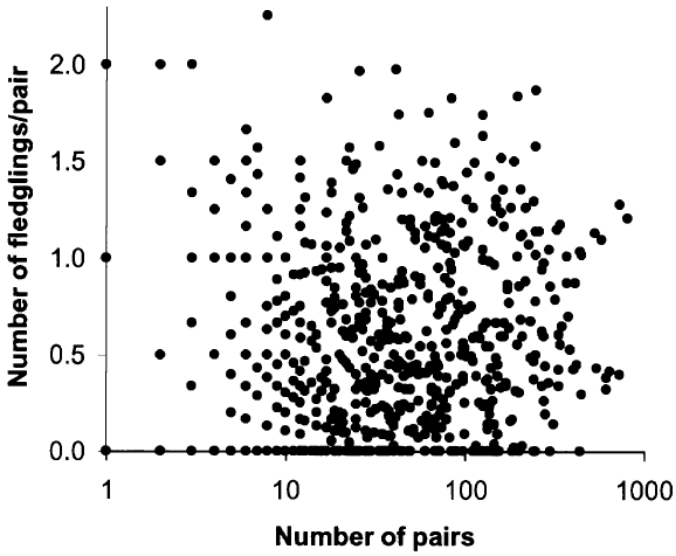


Fig. 5. The relationship between the size of a California least tern colony (in number of pairs) and the number of fledglings per pair in the colony, based on pooled data from all colonies in California, USA, and Baja California, Mexico, 1971–1998. The colony size explains <2% of the variation in the number of fledglings per pair, suggesting a lack of density-dependent effects on fecundity.

G.4.2 Schweitzer and Leslie 1999

Findings:

TABLE 2.—Spatial distribution pattern of least tern nests from nearest-neighbor analyses (Krebs, 1989) in active colony sites

Colony site	Number of nests	Area (ha)	Density (nests/ha)	Mean distance between nests (m)	Expected distance between nests (m)	Index of aggregation (r)	Standard deviation	Test statistic (z)	Spatial pattern
1992									
F	11	40.0	0.26	94.0	95.4	1.0	15.0	−0.1	random
G	12	16.0	0.75	71.2	57.7	1.2	8.7	1.5	random
K	20	37.5	0.53	66.4	68.5	1.0	8.0	−0.3	random
L	15	37.5	0.40	90.6	79.1	1.2	10.7	1.1	random
M	5	7.5	0.67	70.5	61.2	1.2	14.3	0.6	random
N	6	8.8	0.69	101.7	60.4	1.7	12.9	3.2	uniform
1993									
DC	10	27.5	0.36	90.8	82.9	1.1	13.7	0.6	random
F	25	10.9	0.23	56.0	33.0	1.7	3.4	6.7	uniform
FC	14	27.0	0.52	89.5	69.4	1.3	9.7	2.1	uniform
G	7	2.0	0.35	44.7	26.7	1.7	5.3	3.4	uniform
KL	10	29.5	0.34	62.4	85.9	0.7	14.2	−1.7	random
CH	11	1.1	0.10	21.4	15.8	1.4	2.5	2.2	uniform
$\bar{x}_{overall}$	12	20.4	0.60	71.6	64.8	1.1	9.7	0.7	random

G.5 Habitat Quality and Quantity

G.5.1 Sidle et al. 1992

Findings:

Table 1. The number of monitored Least Tern (LT) and Piping Plover (PP) nests inundated along the Platte River between the Loup River confluence and the Missouri River, 1986-90.

Yr	No. Nests Before First Flood	No. Flooded	% Flooded	No. Renests[1]	No. Renests Flooded	Total % Flooded	Maximum Daily Flow (cms) NB/LV[2]
	LT/PP	LT/PP	LT/PP	LT/PP	LT/PP	LT/PP	
1986[3]	136/28	18/6	13/21	0/0	0/0	13/21	421.7/793.0
1987[4]	69/17	5/0	7/0	0/0	0/0	7/0	302.8/484.3
1988[5]	77/32	3/1	4/3	0/0	0/0	4/3	143.2/162.6
1989[5]	61/13	21/7	34/54	24/5	0/0	28/39	331.1/320.0
1990[5]	94/28	94/28	100/100	93/13	4/6	52/85	911.3/1713.3

[1]Least terns and piping plovers renested after losing eggs or chicks from other causes in 1986 and 1988 but initiated no new nests on the river after flooding.
[2]Flows measured at U.S. Geological Survey gauging stations at North Bend and Louisville, Nebraska.
[3]Flooding occurred in late June after most nests had hatched.
[4]Flooding occurred in late May-early June before birds had initiated many nests.
[5]The first flood inundated sandbars along the entire river and occurred during peak nesting. The second flood inundated sandbars below Salt Creek (U.S. Army Corps of Engineers river mile 26) and occurred in late July after most tern but no plover renests had hatched. The second flood killed all chicks on sandbars below river mile 26.

G.5.2 Smith and Renken 1991

Findings:

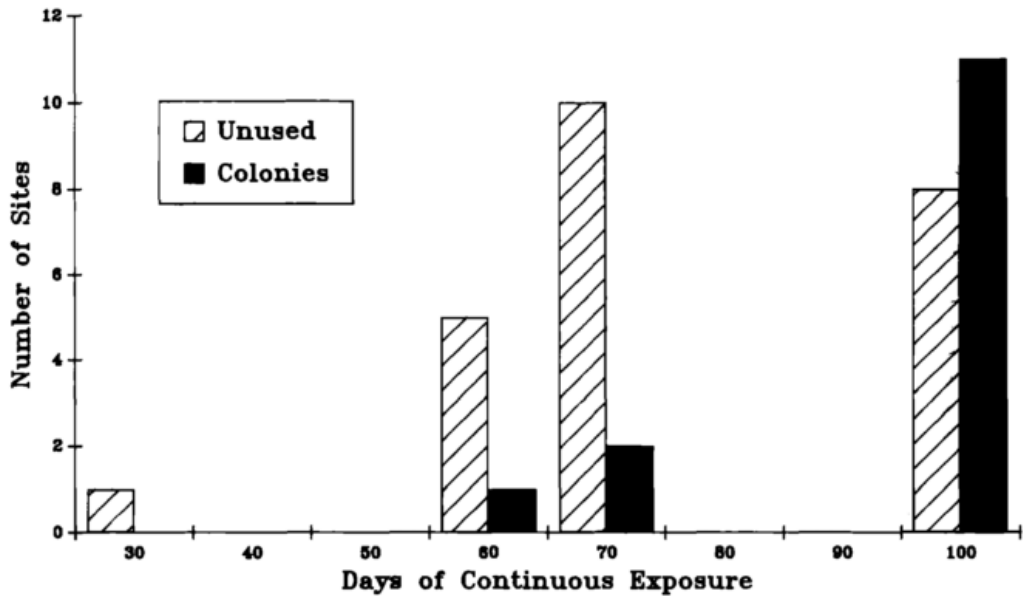


FIGURE 1. Distribution of interior Least Tern colonies and unused sites by category of days of continuous exposure, lower Mississippi River, 1985.

TABLE 1. Mean ± SE of habitat characteristics of interior Least Tern colonies and unused islands or sandbars in the Mississippi River within the region where Least Terns nest (Lower River Mile 827 to Upper River Mile 30), and sites outside the breeding range (Upper River Mile 31 to Upper River Mile 361).

	Colonies (n = 14)	Unused islands (n = 24)	Outside breeding region (n = 48)
Mean days of continuous exposure during 15 May–31 August	101.6 ± 4.0	83.7 ± 4.1	85.4 ± 3.8
Mean % vegetation cover	8.9 ± 4.2	4.5 ± 2.2	16.8 ± 12.3

G.5.3 Leslie et al. 1997

Findings:

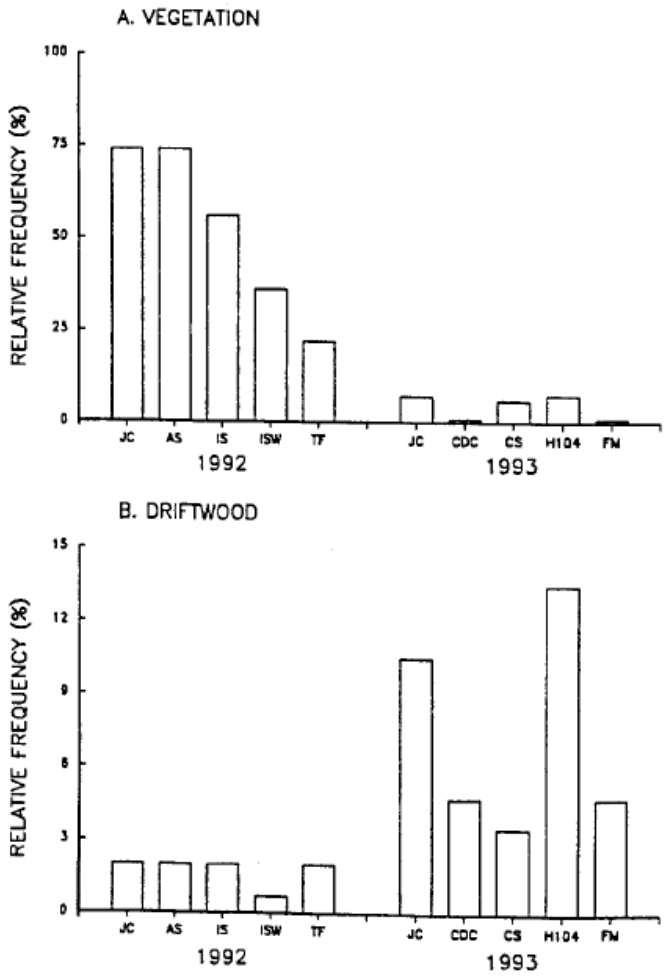


Figure 1. Frequency of occurrence of vegetation and driftwood in 75–200 0.1-m^2 plots at 5 colony sites of least terns on the Arkansas River from Tulsa to Muskogee, Oklahoma, prior to (summer 1992) and after (summer 1993) scouring floods. Colony abbreviations: JC, Joe Creek; AS, Anchor Stone; IS, Indian Springs; ISW, Indian Springs West; TF, Taft; CDC, Cedar Creek; CS, Coweta; H104, Highway 104; FM, Fern Mountain.

G.5.4 Sherfy et al. 2009a

Findings:

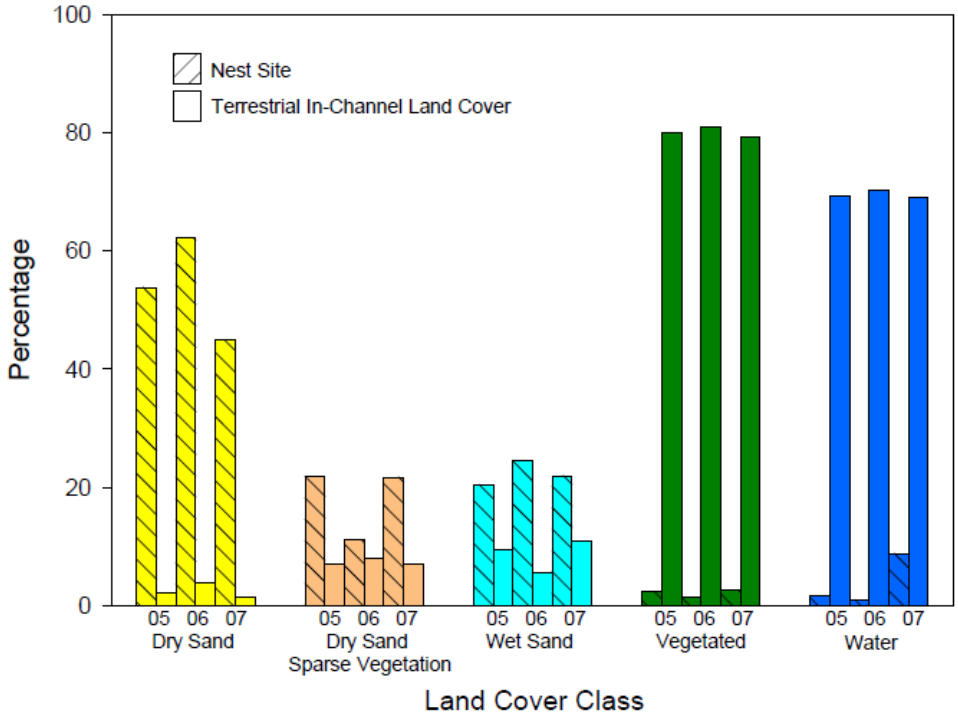


Figure 2. Distribution of least tern and piping plover nests among land cover classes and the percentage of the land cover class for the Gavins Point Reach in 2005, 2006 and 2007.

Table 9. Numbers (n) of least tern and piping plover nests where habitat characteristics were evaluated on the Missouri River in summer 2008.

Study Area	River Miles	Species	n	Measurements
Gavins Point	754–795	Least Tern	192	n, 3m
		Piping Plover	223	n
Lewis & Clark Lake	826–832	Least tern	140	n, 3m
		Piping Plover	61	n
Lake Sakakawea	1390–1508	Piping Plover	93	n, 3m, 10m, 2rand, s
		Least Tern	29	n, 3m, 10m, 2rand, s

n = Evaluation of 1-m-square plot centered on nest at time of discovery.

3m = Evaluations on 4 1-m-square plots positioned 3 m from nest on cardinal directions.

10m = Evaluations on 4 1-m-square plots positioned 10 m from nest on cardinal directions.

2rand = 2 random points where n, 3m, and 10m were evaluated.

s = Segment of shoreline, 1km centered on nest, evaluated with transects.

G.5.5 Carreker 1985

Findings:

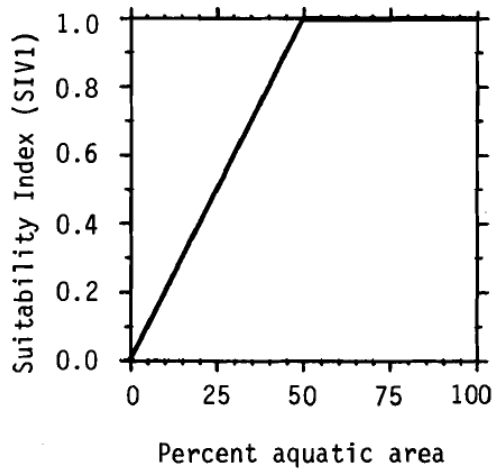


Figure 3. The relationship between the percent of the area within the average maximum flight distance from the potential nesting habitat that is aquatic habitat and the suitability index value for least tern food.

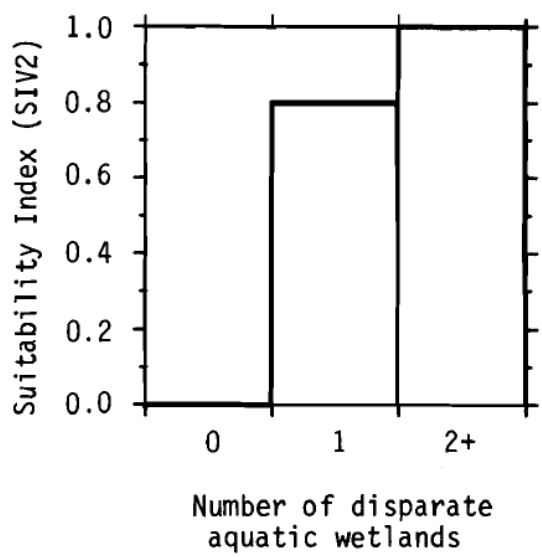


Figure 4. The relationship between the number of disparate aquatic wetlands within the average maximum flight distance from the potential nesting habitat and the suitability index value for least tern food.

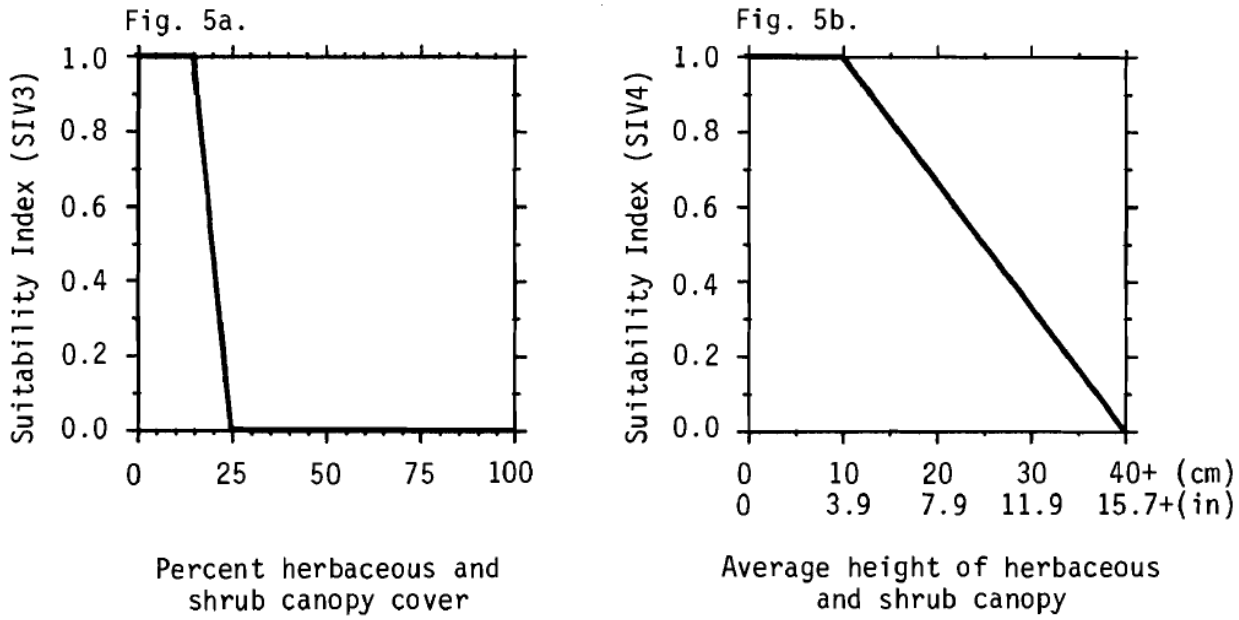


Figure 5. The relationships between vegetation cover and the suitability index values for least tern reproduction.

G.5.6 Kirsch 1996

Findings:

Table 20. Summary of habitat variables at least tern colony sites on sandbars along the lower Platte River, Nebraska, 1987–90.

Year	No. sites	Distance to bank (m)	% of colony area[a] flooded
1987	3	5–25	0–5
1988	4	25–120	0–10
1989	6	85–165	80–95
1990	6	0–190	0–100

[a] That part of a nesting site that terns actually used for nesting, brooding and feeding chicks, and loafing.

Table 21. Summary of habitat variables at least tern colony sites on sandpits along the lower Platte River, Nebraska, 1987–90.

Year	No. sites	% of colony area[a] perimeter bordered by water	Distance to abundant vegetation[b] (m)
1987	5	25–75	5–200
1988	3	25–75	40–200
1989	6	25–75	10–200
1990	7	25–75	25–400

[a] That part of a colony site that terns actually used for nesting, brooding and feeding chicks, and loafing.
[b] Abundant vegetation = >50% cover and >30 cm tall.

G.5.7 Stucker 2012

Findings:

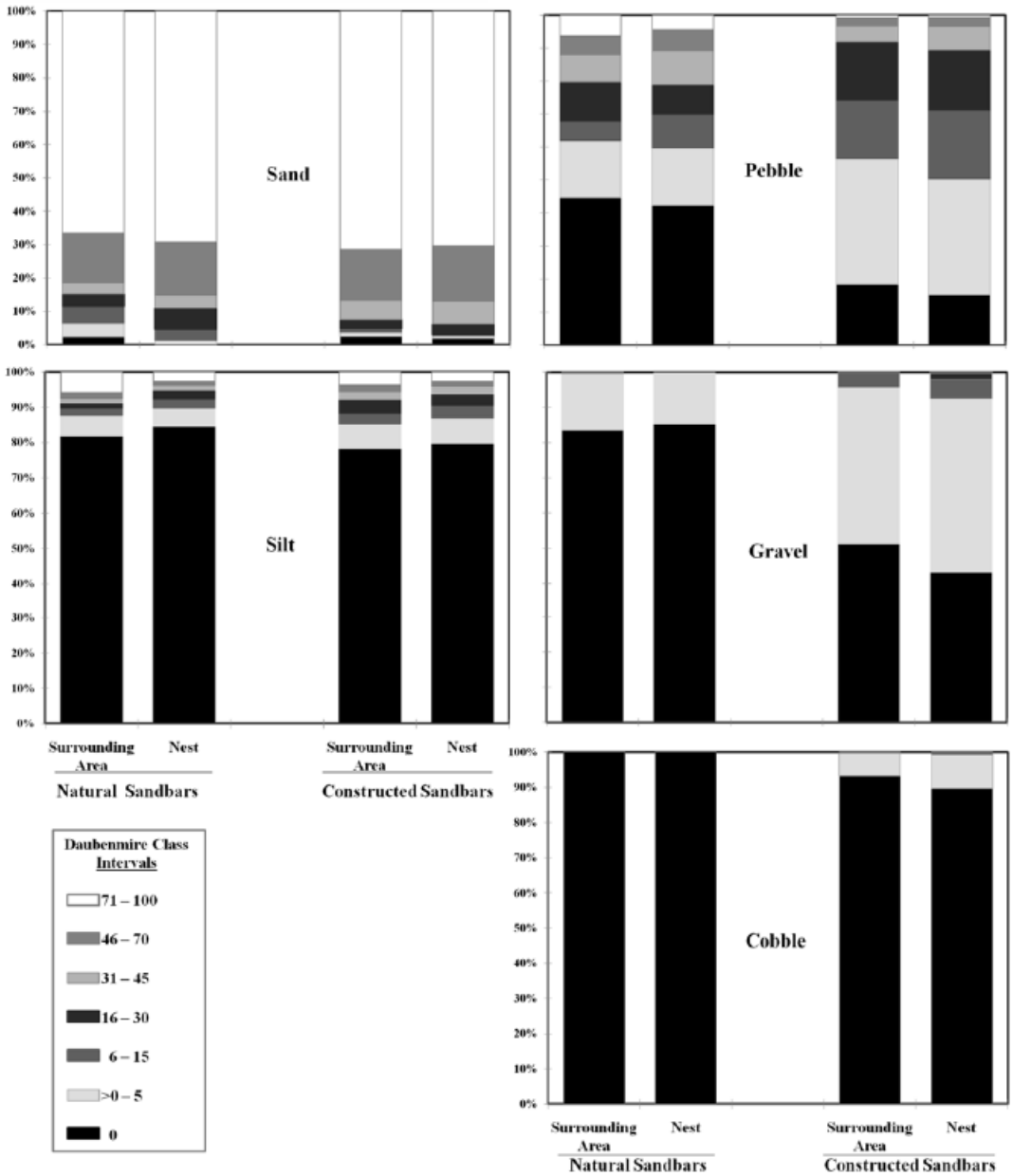


Figure 3. Substrate characteristics for Least Terns nesting on natural and constructed sandbars within the Missouri River (Gavins Point Reach, and Lewis and Clark Lake), 2006 – 2008. Results visually illustrate proportions of substrate type classifications among sampled nest sites and adjoining surrounding areas (3 m from nests) on natural and constructed sandbars.

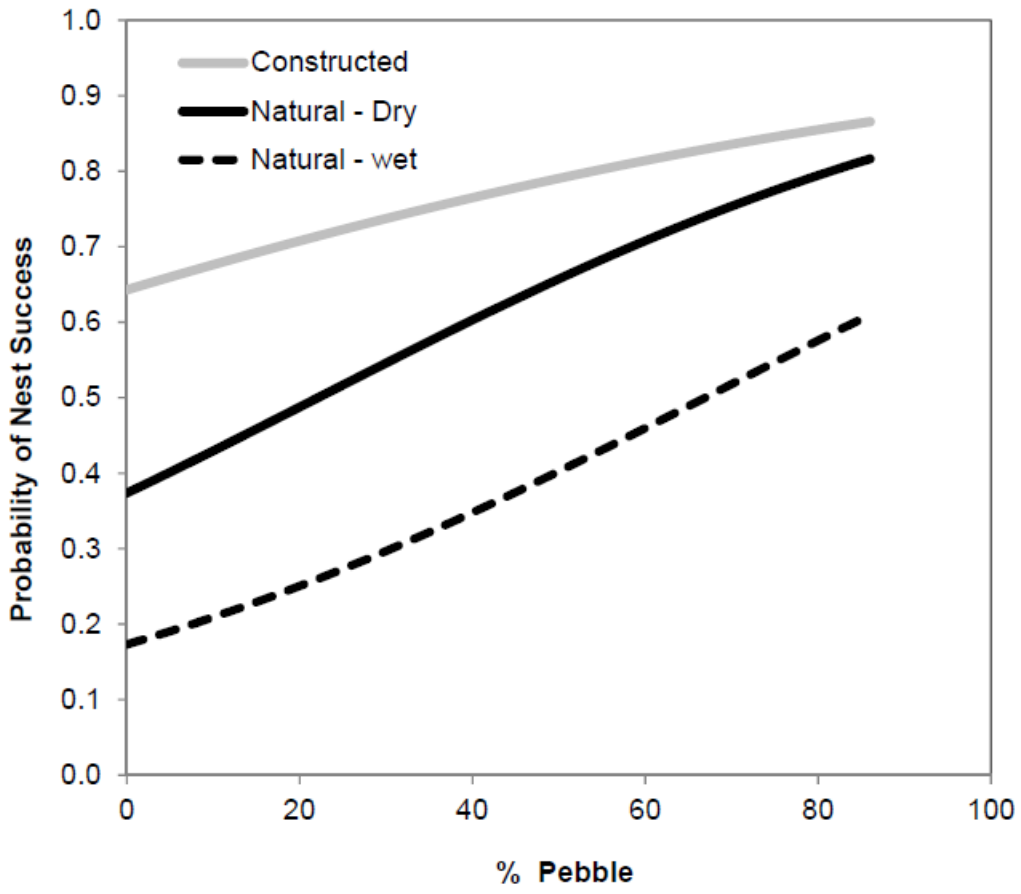


Figure 4. Increasing percentage of pebble at Least Terns nesting sites was associated with increased nest success on constructed and natural sandbars within the Missouri River (Gavins Point Reach, and Lewis and Clark Lake), 2006 – 2008. Substrate moisture differences in surrounding area (3m from nest) for natural sandbar sites are indicated by solid (dry) and wet (dashed) lines.

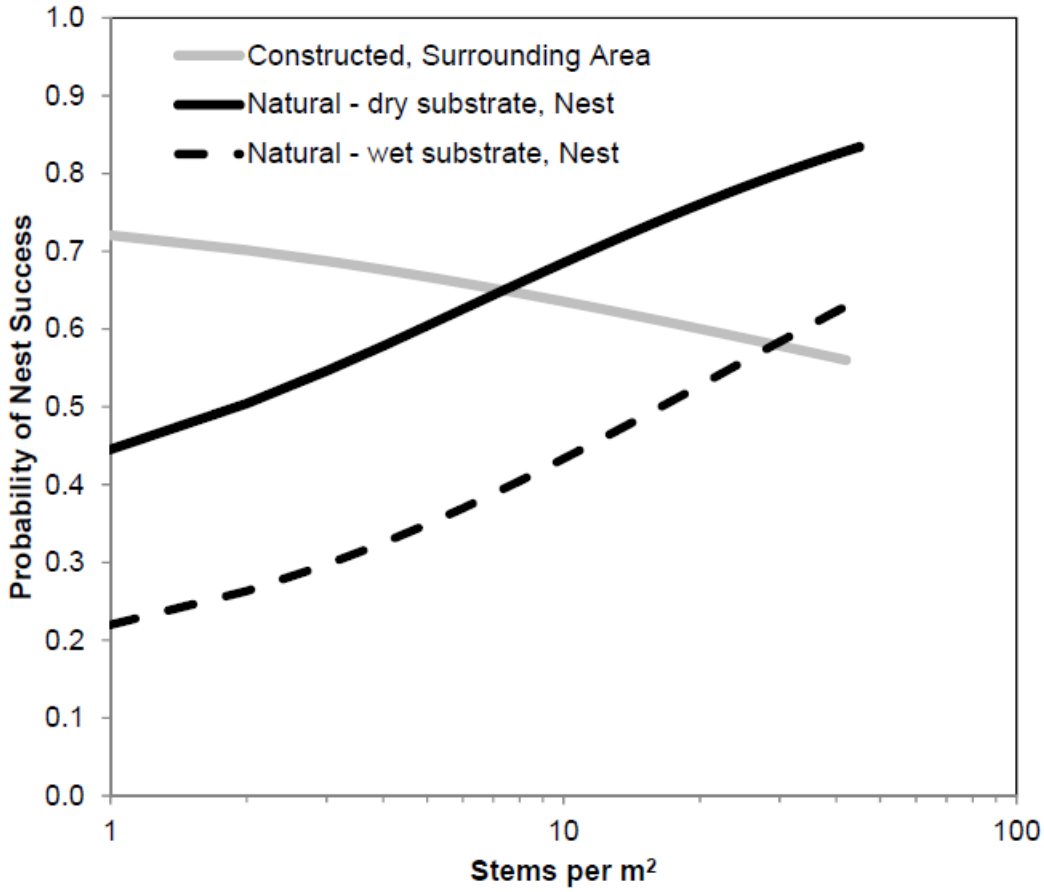


Figure 4. Stem counts were associated with Least Tern nest success within the Missouri River (Gavins Point Reach, and Lewis and Clark Lake), 2006 – 2008. Among natural sandbars increasing stem counts were associated with increased success; substrate moisture differences within surrounding area (3m from nest) are indicated by solid (dry) and wet (dashed) lines. Stem counts from the surrounding area of constructed sandbars were associated with decreased nest success.

Table 8. Mean, standard deviation (), and back-transformed mean [] values of continuous variables included fitting models of airborne Least Tern habitats on the Missouri River 2006-2008.

	2006		2007		2008	
	Random	Tern	Random	Tern	Random	Tern
Sandbar	[b] 0.031	[b] 0.055	[a] 0.031	[a] 0.09	[b] 0.049	[b] 0.140
	(0.082)	(0.087)	(0.101)	(0.175)	(0.088)	(0.150)
Large.Trees	[a] 0.070	[a] 0.007	[b] 0.081	[b] 0.023	[a] 0.033	[a] 0.004
	(0.186)	(0.025)	(0.138)	(0.047)	(0.099)	(0.020)
Water	[a] 0.693	[a] 0.812	[a] 0.674	[a] 0.058	[a] 0.579	[a] 0.558
	(0.403)	(0.231)	(0.406)	(0.111)	(0.420)	(0.357)
Wet sand	[b] 0.008	[b] 0.035	[a] 0.015	[a] 0.738	[b] 0.049	[b] 0.104
	(0.024)	(0.079)	(0.056)	(0.289)	(0.084)	(0.104)
ln Bank Distance					5.145	5.415
					(1.013)	(0.858)
					[171 m]	[224 m]

[a] proportion within 50m, [b] proportion within 200m

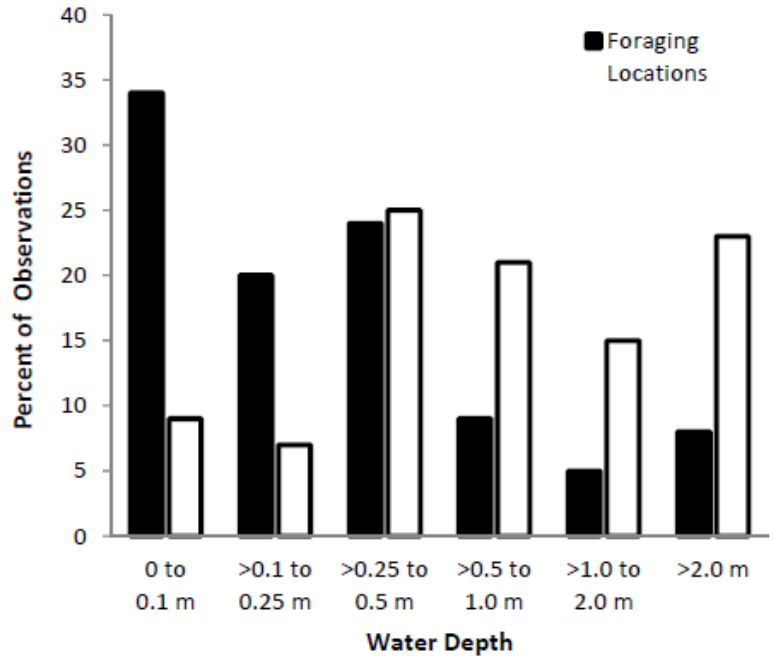
Table 10. Mean, standard deviation (), and back-transformed mean [] of continuous variables included in the best-fitting models identifying probabilities of airborne Least Terns as demonstrating behaviors of foraging or flying on the Missouri River 2007-2008.

	2007		2008	
	Flying	Foraging	Flying	Foraging
Sandbar_200m	0.095 (0.047)	0.084 (0.048)	0.168 (0.173)	0.105 (0.104)
Large.trees_200m	0.022 (0.047)	0.025 (0.048)	0.016 (0.041)	0.026 (0.053)
Water	[b] 0.057 (0.105)	[b] 0.060 (0.119)	[a] 0.522 (0.379)	[a] 0.603 (0.321)
Wetsand distance (ln)	3.588 (1.561)[35m]	3.5 (1.484)[32m]	3.086 (1.563) [21m]	2.844 (1.603) [16m]
Bank distance (ln)			5.521 (0.765) [249m]	5.281 (0.948) [196m]

[a] proportion within 50m

[b] proportion within 200m

(a)



(b)

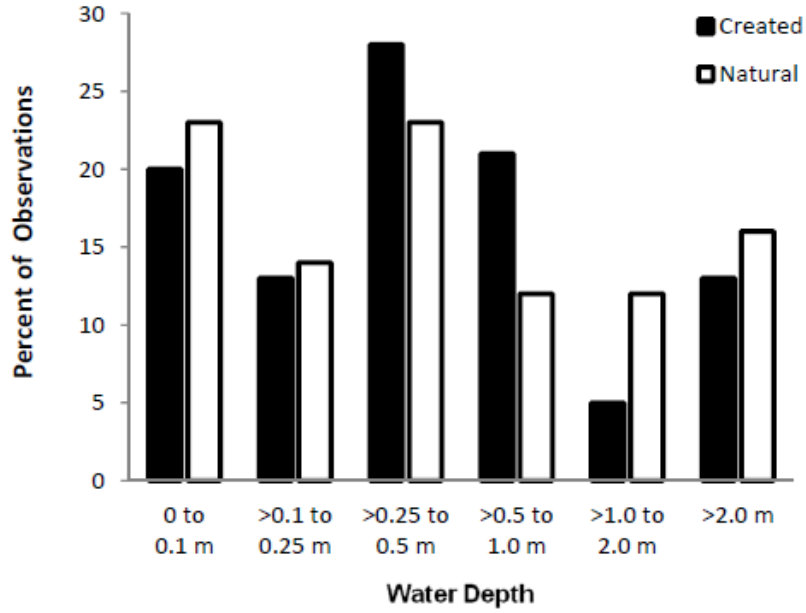


Figure 1. Proportion of observations, by depth, for (a) foraging and random locations, and (b) mechanically created and natural sites.

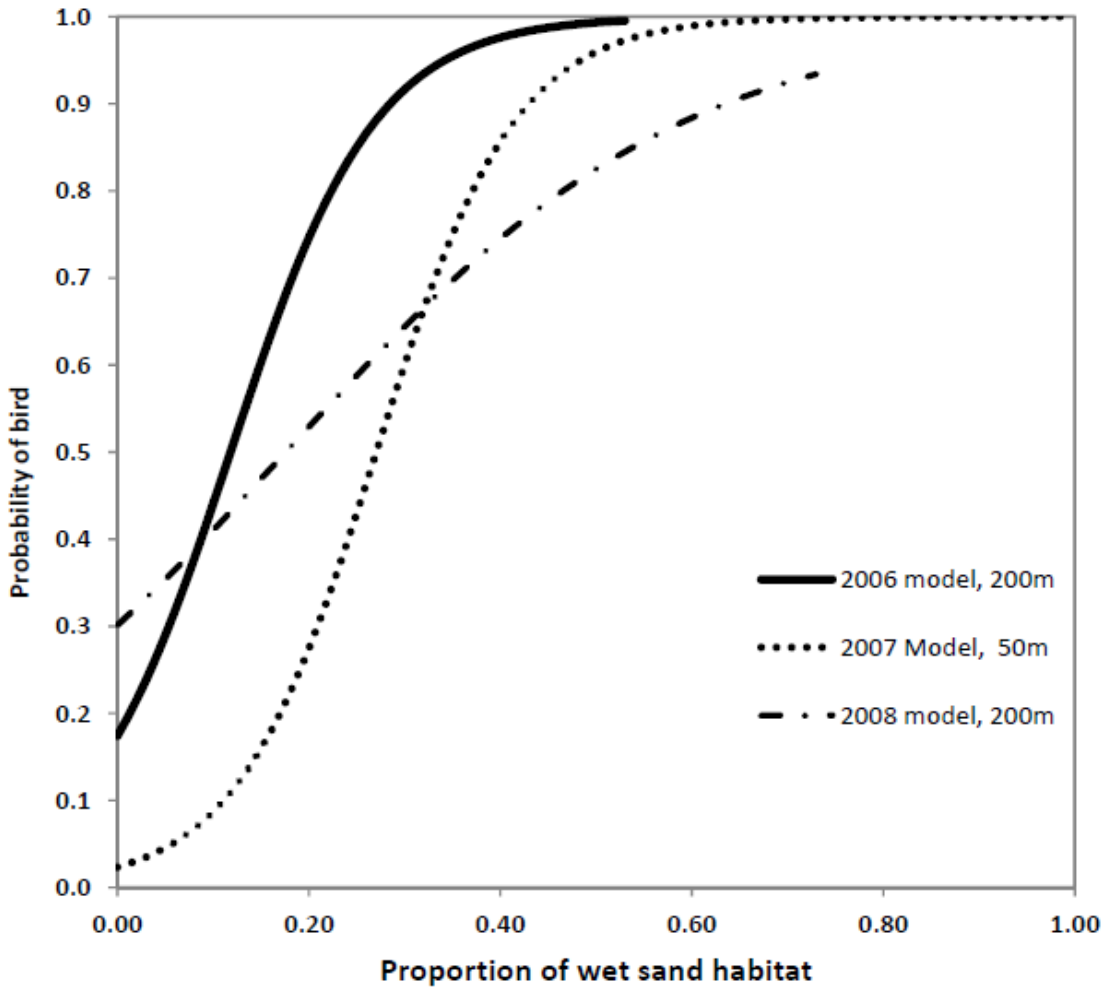


Figure 2. The proportion of wet sand habitats within 50m or 200m and probability of bird use as predicted by best model for each year.

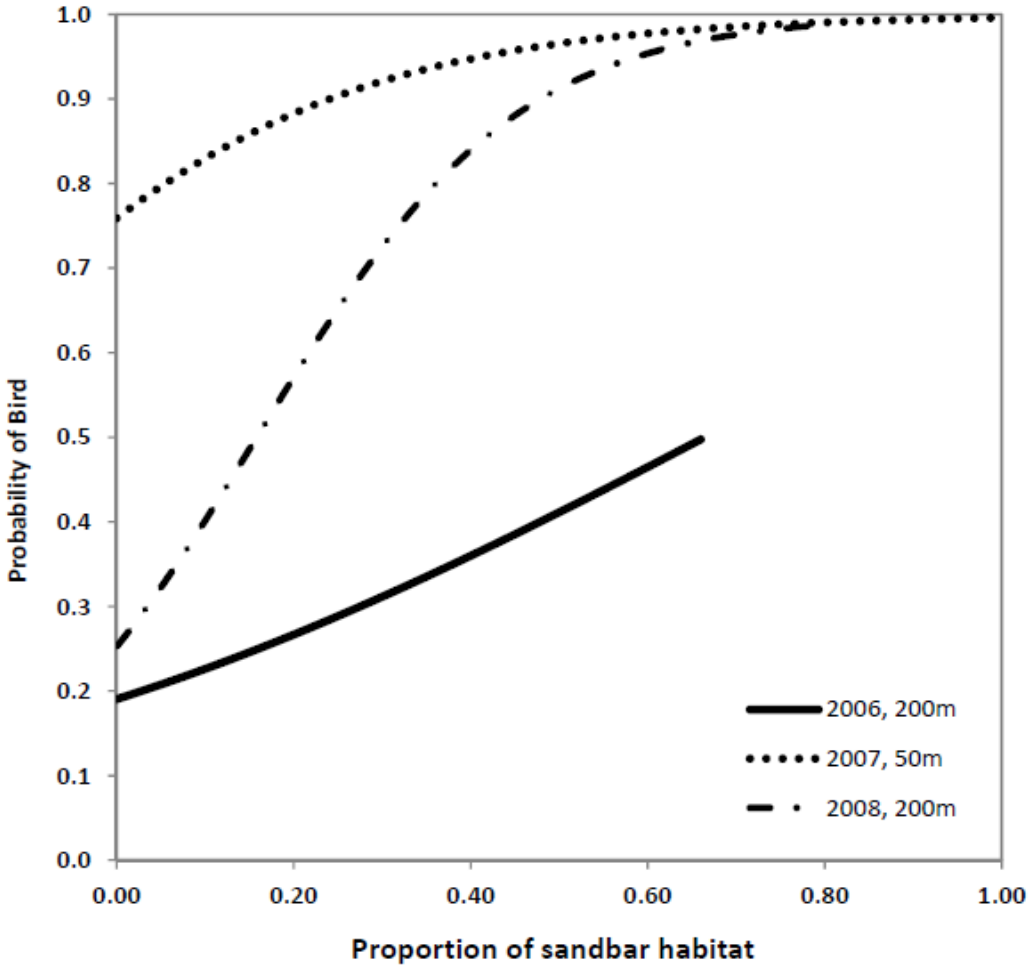


Figure 3. The proportion of sandbar nesting habitats within 50m or 200m and probability of bird use as predicted by best model for each year.

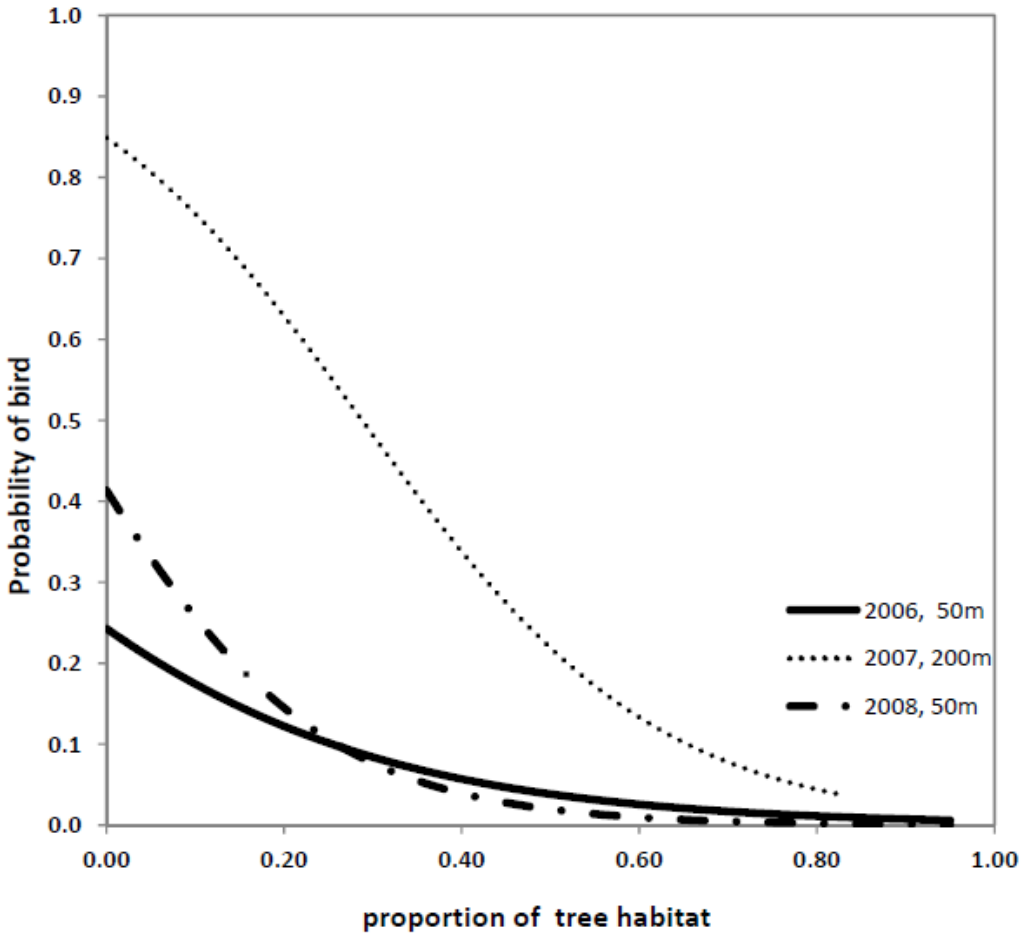


Figure 4. The proportion of large tree habitats within 50m or 200m and probability of bird use as predicted by best model for each year.

G.6 Predation

G.6.1 Shwiff et al. 2005

Findings:

Table 3
Predator-removal and reproduction-monitoring budgets

Year	2003 Dollars[a]					
	Predator budget (US$)	Rank	Monitor budget (US$)	Rank	Total budget (US$)	Rank
1995	76,170	2	71,046	6	147,216	5
1996	78,753	1	68,477	7	147,230	4
1997	73,019	4	76,594	3	149,613	3
1998	73,165	3	81,275	1	154,440	1
1999	70,366	7	75,890	4	146,256	7
2000	71,601	6	75,328	5	146,929	6
2001	72,763	5	80,005	2	152,768	2

[a] Adjusted for inflation.

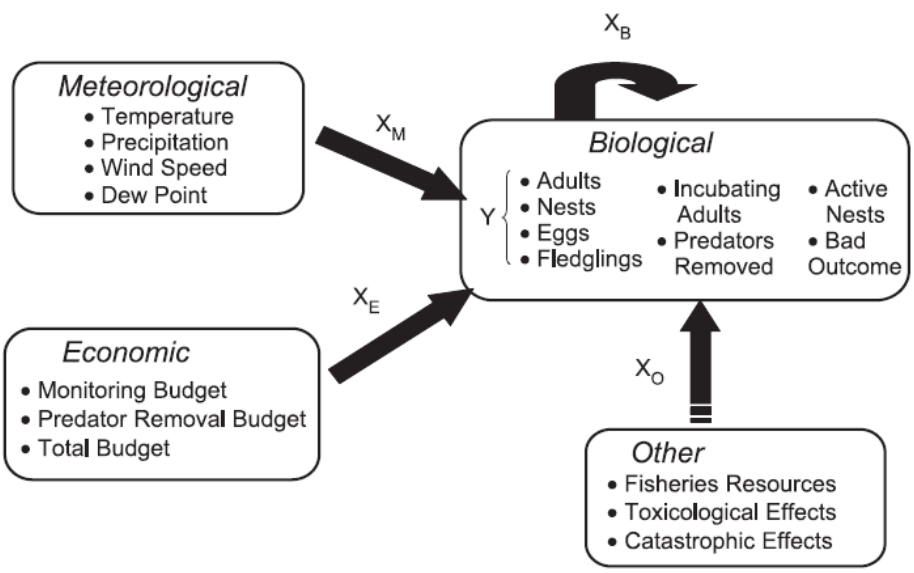


Fig. 2. Schematic of the Tern Reproduction System and interaction of biological, economic, meteorological, and other variables.

Table 6
Percent increase forecast for the dependent variables as a result of an increase in the independent variables

Development stage/ dependent variable	Scaled independent variable								
	Monitor hours (%)			Predator-removal hours (%)			Total hours (%)		
	25	50	100	25	50	100	25	50	100
Adults	7.6	14.1	24.7	4.8	9.1	16.7	8.1	16.1	32.2
Total nests	105.6	211.5	423.2	29.1	58.4	117.1	1.2	2.6	5.5
Eggs	4.2	8.5	16.9	10.4	20.8	41.5	4.2	8.4	16.7
Fledglings	38.6	77.2	154.3	−42.9	−85.8	−171.7	24.9	49.8	99.5

G.6.2 Brunton 1997

Findings:

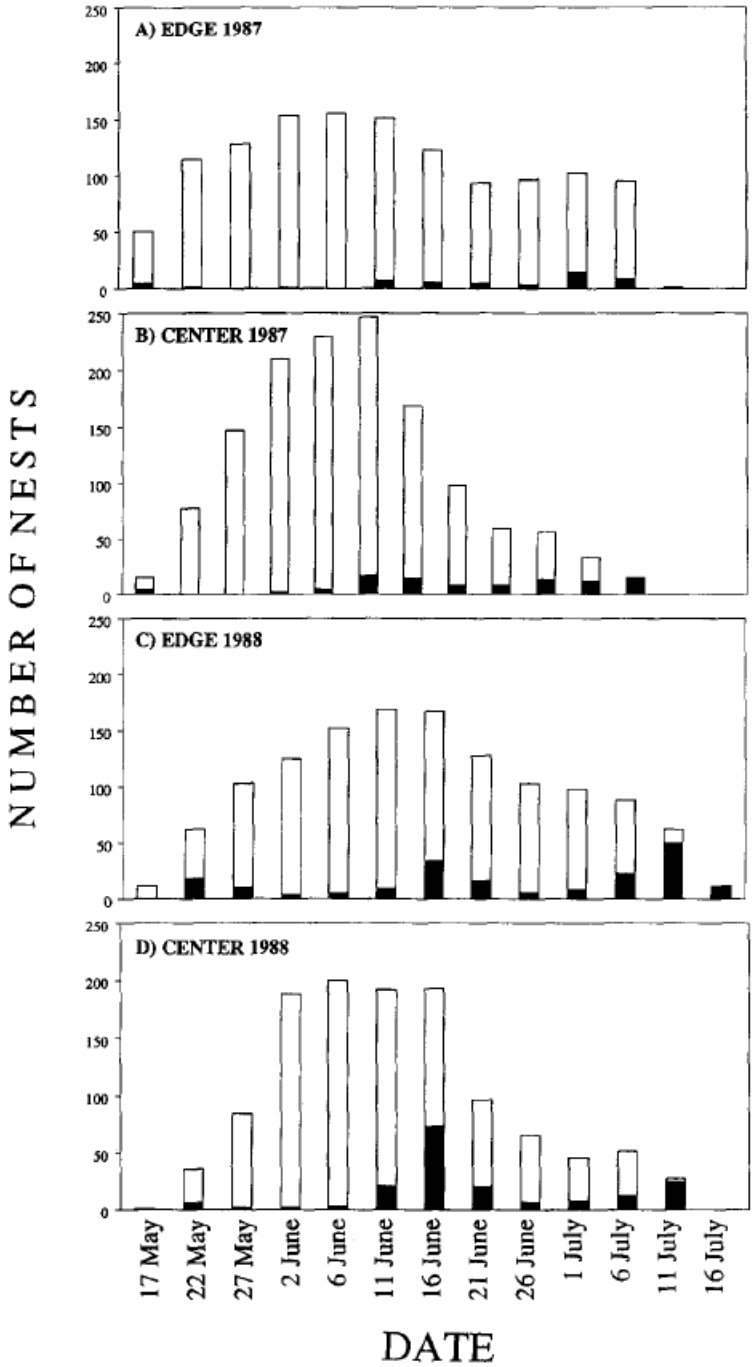


FIGURE 1. The relationship between the number of active nests (open bars) and failed nests (solid bars) in consecutive 5-day intervals throughout the breeding season. Data are shown separately for the edge region: (a) 1987 and (c) 1988, and the center region: (b) 1987 and (d) 1988.

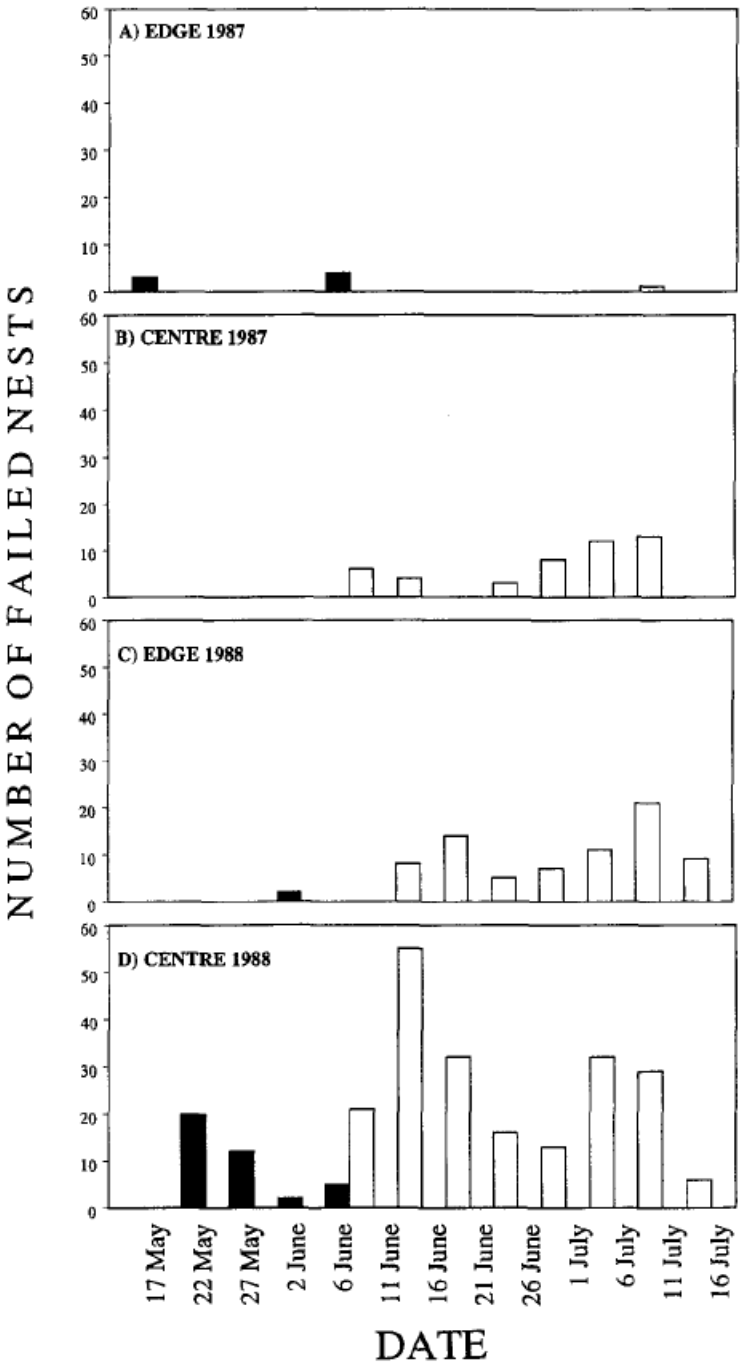


FIGURE 2. The number of nest failures caused by American Crows (solid bars) and Black-crowned Night-Herons (open bars) during consecutive 5-day intervals over the breeding season. Data are shown separately for the edge region: (a) 1987 and (c) 1988, and the center region: (b) 1987 and (d) 1988.

G.6.3 Kirsch 1996

Findings:

Table 17. Mean frequencies of track trails at least tern colony sites on sandbars and sandpits on the lower Platte River, Nebraska, 1987–90. The nesting period ran from nest initiation to when 75% of the nests became inactive (hatched or destroyed). The chick rearing period followed the nesting period and ended when the last chick fledged.

			Total season		Nesting		Chick rearing		Unused sites[a]	
Habitat	Year	n	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Sandbar	1987	3	0.76	0.19	0.82	0.22	0.44	0.10		
	1988	4	0.68	0.16	0.88	0.23	0.30	0.06		
	1989	6	0.66	0.28	0.71	0.34	0.74	0.46		
	1990	6	1.34	0.38	0.57	0.26	2.08	1.10	2.50	0.50
Sandpit	1987	5	0.39	0.04	0.46	0.14	0.32	0.12		
	1988	3	0.47	0.07	0.37	0.03	0.70	0.25		
	1989	6	0.87	0.22	0.62	0.23	1.03	0.36		
	1990	7	1.66	0.40	1.45	0.54	1.73	0.37	3.04	0.87

[a] Data from unused sites were collected in 1990 from 3 sandbar and 4 sandpit sites.

Table 18. Correlations among number of nests, numbers of eggs lost due to predation and abandonment, and frequency of different types of track trails for least tern colonies along the lower Platte River, Nebraska, 1987–90. Correlations between track trail types and other variables were Spearman rank, and all other correlations were Pearson product-moment.

		No. eggs lost to		Frequency of track trails					
Habitat	Number	Predation	Abandon-ment	Terrestrial predator	Human	Owl	ATV[a]	Heavy equipment	All trails
Sandbars	Nests	0.317	0.574**	0.107	0.303	0.233	0.113		0.316
	Depredated eggs		0.288	0.494**	0.170	0.399	−0.144		0.565
	Abandoned eggs			0.039	0.143	−0.027	−0.024		−0.037
Sandpits	Nests	0.049	0.631**	−0.306	0.286	0.182	0.013	0.220	−0.185
	Depredated eggs		0.148	−0.357	−0.018	0.284	−0.392	0.433*	0.235
	Abandoned eggs			−0.529*	0.459	0.247	−0.324	0.433*	−0.308

[a] All-terrain vehicle.
* Significant correlation, $P < 0.05$.
** Significant correlation, $P < 0.01$.

Table 19. Spearman rank correlations of proportions of least tern chicks known alive on each visit and numbers of chicks hatched at a colony, with frequency of different types of track trails on sandbar and sandpit colony sites on the lower Platte River, Nebraska, 1987–90.

		Frequency of track trails					
Habitat	Chick variables	Terrestrial predators	Humans on foot	Owl	ATV[a]	Heavy equipment	All track trails
Sandbars	Proportion of chicks alive (n = 156 visits)	−0.002	−0.010	0.007	−0.223*		−0.117
	Number of chicks hatched (n = 18 sites)	0.242	0.497	0.306*	0.561*		0.569
Sandpits	Proportion of chicks alive (n = 223 visits)	−0.234**	−0.023	−0.106	−0.093	0.016	−0.172
	Number of chicks hatched (n = 18 sites)	0.222	0.567*	0.226	0.386	0.030	0.413*

[a] All-terrain vehicle.
* Significant correlation, $P < 0.05$.
** Significant correlation, $P < 0.01$.

G.7 Stage Survival

G.7.1 Massey et al. 1992

First Winter Survival:

TABLE 1. Demography of the Least Tern population in California and at Venice Beach 1977–1989.

	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989
California													
Number of breeding pairs	775	832	1,000	1,160	1,067	1,130	1,264	1,046	1,003	960	945	1,253	1,240
Estimated number of fledglings	449	418	696	769	733	511	894	518	682	902	632	1,130	764
Productivity ratio[1]	0.58	0.50	0.70	0.66	0.69	0.45	0.71	0.50	0.66	0.94	0.67	0.90	0.62
Venice colony													
Number of breeding pairs	35	67	87	157	150	150	140	82	96	104	109	165	137
Estimated number of fledglings	37	75	145	240	200	60	140	94	113	113	82	192	134
Productivity ratio	1.1	1.1	1.7	1.5	1.3	0.4	1.0	1.2	1.2	1.1	0.8	1.2	1.0
Number of chicks banded	0	87	157	291	139	208	173	84	158	117	92	285	149
Percent of returns[2]	NA[3]	10	10	4	9	2	2	4	9	7	NA[4]	NA[4]	NA[4]

[1] Number of fledglings/pair of adults.
[2] Based on number of chicks banded each year that returned as adults to breed.
[3] Not applicable—no chicks banded in 1977.
[4] Not applicable—chicks banded 1987–1989 did not reach age of major first breeding during the study.

TABLE 4. Demographic analysis.

		Normal years		ENSO years§	
		Estimate	95% Confidence Interval	Estimate	95% Confidence Interval
Return rates					
Hatchling*	S_h	0.16	(0.13, 0.18)	0.03	(0.01, 0.05)
Young breeders†	S_y	0.81	(0.72, 0.87)	0.82	(0.66, 0.96)
Older breeders‡	S_a	0.92	(0.88, 0.95)	0.79	(0.68, 0.89)
Demographic parameters					
Hatchling productivity**	P_H	1.00	(0.84, 1.20)	0.98	(0.88, 1.08)
Breeding life††	L	9.63	(6.55, 14.48)	4.97	(3.46, 8.10)
Lifetime productivity‡‡	P_L	1.49	(0.93, 2.34)	0.15	(0.06, 0.30)

* Percentage of chicks hatched at Venice that return as adults to breed.
† Annual survival rate for birds breeding at ages two and three.
‡ Annual survival rate for birds breeding at ages four or older.
** Hatchlings produced per breeder per year.
†† Expected number of breeding years per adult: $L = 1 + S_y + S_y^2/(1 - S_a)$.
‡‡ Number of progeny per adult expected to survive to breeding age: $P_L = S_h \times P_H \times L$.
§ Estimates based on data from the 1982–1983 El Niño Southern Oscillation.

TABLE 5. Age-dependent survival rates.

Age	*n*	Survival estimate	95% Confidence Interval
2	31	0.82	(0.69, 0.93)
3	73	0.80	(0.72, 0.88)
4	64	0.90	(0.83, 0.96)
5	52	0.92	(0.85, 0.98)
6	43	0.93	(0.85, 0.98)
7	43	0.87	(0.78, 0.96)
8	31	0.87	(0.76, 0.97)
9	20	0.92	(0.80, 1.00)
10	10	0.76	(0.50, 1.00)
11	3	1.00	(1.00, 1.00)
12	2	0.60	(0.17, 1.00)

Adult Survival – Study Design – Sample Size:

TABLE 2. Age of adults at Venice Beach when first trapped on nests (154) or identified by color code (32); $n = 186$.

Number of adults	Age (years)								
	2	3	4	5	6	7	8	9	10
Trapped	37	64	37	6	5	2	2	0	1
Identified by color code	21	11	0	0	0	0	0	0	0
Total	58	75	37	6	5	2	2	0	1
Percent	31	40	20	3.2	2.7	1.1	1.1	0	0.5

TABLE 3. Return rate of banded adults breeding at Venice (both known and unknown age).

	1983	1984	1985	1986	1987	1988	1989
Total returns[1]	0	37	42	58	59	64	76
Total possible returns[2]	0	56	48	65	69	74	104
New birds	56	11	23	11	15	40	32
Return rate (%)	—	66	88	89	86	86	73

[1] Adults that nested at Venice the previous year.
[2] Banded adults trapped for the first time and unbanded mates of banded birds.

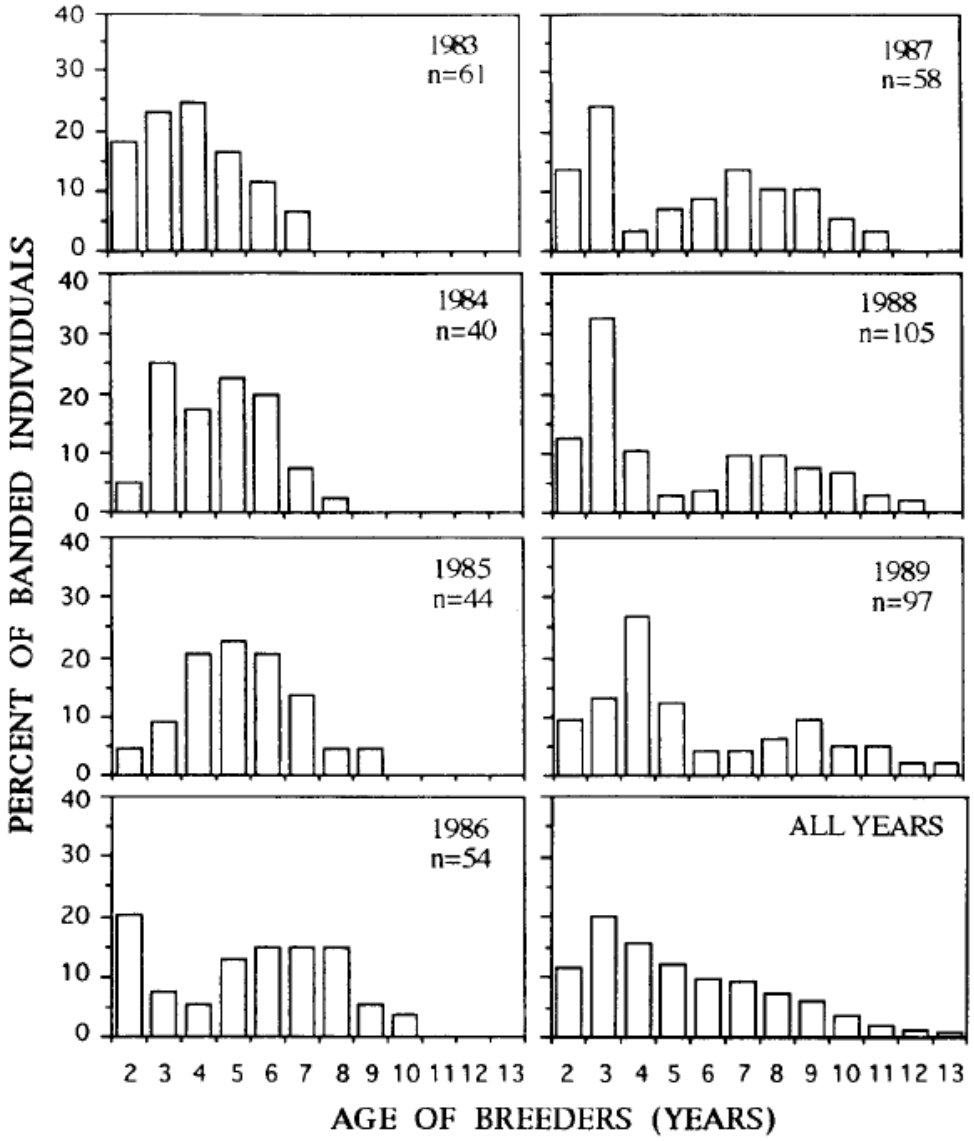


FIGURE 1. Age profile of banded adults breeding at Venice, 1983–1989.

G.7.2 Renken and Smith 1995a

Second Winter Survival – Adult Survival:

TABLE 1. Capture-recapture summary statistics for adult Least Terns in the Mississippi River valley. Definitions of variables are taken from Pollock et al. (1990).

Period	Date	n_i	m_i	R_i	r_i	z_i
1	1987	37	—	37	16	—
2	1988	37	3	37	19	13
3	1989	73	14	73	41	18
4	1990	104	40	104	22	19
5	1991	24	24	23	5	17
6	1992	22	22	22	—	—

n_i = Total number of animals captured in the ith sample (includes captured and resighted birds).
m_i = Number of marked animals captured in the ith sample (resighted birds).
R_i = Number of n_i that are released after the ith sample.
r_i = Number of R_i released at i that are captured again.
z_i = Number of animals captured before i, not captured at i, and captured again later.

G.7.3 Whittier and Leslie 2009

First Winter Survival:

TABLE 1—Movement pattern and fate of chicks (%) of the least tern (*Sterna antillarum*) after they abandoned their nest at Salt Plains National Wildlife Refuge, Alfalfa Co., Oklahoma. Two chicks were excluded because they were tracked for <3 days and never moved >30 m from the nest.

Movement pattern	All chicks	Fledged	Died	Unknown
Traveler (travel in a definite direction; $n = 5$)	24	20	20	60
Wanderer (wander randomly from the nest; $n = 14$)	66	36	14	50
Sedentary (remain in the vicinity of nest; $n = 2$)	10	50	50	0

G.7.4 Bailey and Servello 2008

First Winter Survival:

Table 3. Estimates of total numbers of banded fledglings and fledgling success for three count methods compared to values based on estimates of fledgling numbers from mark-resighting analyses at two Least Tern breeding colonies in Maine, 2002-2003. Fledgling count data are presented in Table 2.

	Crescent Surf-Laudholm Beach (No. chicks banded = 96)		Higgins Beach (No. chicks banded = 62)	
	No. of fledglings	Fledging success[1]	No. of fledglings	Fledging success[1]
Based on total banded fledglings[2]	70	0.73	46	0.74
Based on highest daily number of banded fledglings (single high-count method)[3]	56	0.58	32	0.52
Based on highest daily number of banded fledglings plus highest number 14 d before and after the highest count	91	0.95	36	0.58
Based on Thompson and Slack (1984) method[4]	60	0.63	29	0.48

[1]Fledging success is calculated as number of fledglings divided by the number of chicks hatched (i.e., no. of chicks banded for this analysis).

[2]Total numbers of banded fledglings present during the nesting season were calculated from chick survival estimates for the interval hatch to fledging and known numbers of chicks banded. Note: estimates of total number of banded fledglings based on mark-resighting analyses and total numbers of individual banded fledglings identified in field surveys were nearly identical for the two colonies (70 vs. 71 and 46 vs. 44, respectively).

[3]The number of banded fledglings potentially present on a single day on a resighting survey (Table 1).

[4]$Count_2 + (Count_4 - 0.55\ [Count_2])$ where: $Count_2$ = the potential number banded fledglings in the second week of the fledgling period, $Count_4$ = the potential number of banded fledglings in the fourth week of the fledgling period. No fledglings were present in subsequent two-week periods.

Study Design – Sample Size:

Table 1. Chick survival estimates for three Least Tern breeding colonies in Maine, 2002-2003.

Colony	Number of chicks banded	Proportion of banded chicks resighted as fledglings[1]	Chick survival (Φ) based on mark-resighting analyses: Daily (Φ ± SE)	Chick survival (Φ) based on mark-resighting analyses: 21-d chick period
Crescent Surf-Laudholm Beach	96	0.74	0.9864 ± 0.0069	0.73
Higgins Beach	62	0.71	0.9855 ± 0.0042	0.74
Crescent Surf Beach	53	0.15	0.9098 ± 0.0148	0.14

[1]Chicks were classified as fledglings when age was ≥21 d.

G.7.5 Dugger et al. 2000

First Winter Survival:

TABLE 2. Laridae fledging success reported as a percentage of the number of hatched chicks that fledged.

Species	Fledging success	Location source
Great Skua (*Catharacta skua*)	7.8%–95.3%	Hamer et al. (1991)
Glaucous-winged Gull (*Larus glaucescens*)	52%–74%	Hunt and Hunt (1976)
Roseate Tern (*Sterna dougallii*)	90%–97%	Nisbet et al. (1995)
Common Tern (*Sterna hirundo*)	59%–88%	Langham (1972)
Common Tern	68.3%–78.9%	LeCroy and LeCroy (1974)
Common Tern	80%	Nisbet (1978)
Common Tern	0%	Uttley et al. (1989)
Arctic Tern (*Sterna paradisaea*)	0%	Uttley et al. (1989)
Least Tern	43%–62%	This Study

G.7.6 Sidle et al. 1992

Study Design – Sample Size:

Table 1. The number of monitored Least Tern (LT) and Piping Plover (PP) nests inundated along the Platte River between the Loup River confluence and the Missouri River, 1986-90.

Yr	No. Nests Before First Flood	No. Flooded	% Flooded	No. Renests[1]	No. Renests Flooded	Total % Flooded	Maximum Daily Flow (cms) NB/LV[2]
	LT/PP	LT/PP	LT/PP	LT/PP	LT/PP	LT/PP	
1986[3]	136/28	18/6	13/21	0/0	0/0	13/21	421.7/793.0
1987[4]	69/17	5/0	7/0	0/0	0/0	7/0	302.8/484.3
1988[3]	77/32	3/1	4/3	0/0	0/0	4/3	143.2/162.6
1989[3]	61/13	21/7	34/54	24/5	0/0	28/39	331.1/320.0
1990[5]	94/28	94/28	100/100	93/13	4/6	52/85	911.3/1713.3

[1]Least terns and piping plovers renested after losing eggs or chicks from other causes in 1986 and 1988 but initiated no new nests on the river after flooding.
[2]Flows measured at U.S. Geological Survey gauging stations at North Bend and Louisville, Nebraska.
[3]Flooding occurred in late June after most nests had hatched.
[4]Flooding occurred in late May-early June before birds had initiated many nests.
[5]The first flood inundated sandbars along the entire river and occurred during peak nesting. The second flood inundated sandbars below Salt Creek (U.S. Army Corps of Engineers river mile 26) and occurred in late July after most tern but no plover renests had hatched. The second flood killed all chicks on sandbars below river mile 26.

G.7.7 Akcakaya et al. 2003

First Winter Survival:

Table 2. Hatching to fledging survival rate of California least tern at Venice beach, California, USA.

	From Massey et al. (1992)			From Minsky (1984)				
Year	Nests	Fledglings	Fledglings per nest	Nests	Total eggs	Failed eggs	Hatchlings per nest	S_h[a]
1981	150	200	1.3333	170	325	16	1.8179	0.7335
1982	150	60	0.4000	189	329	49	1.4817	0.2700
1983	140	140	1.0000	152	291	20	1.7834	0.5607
1984	82	94	1.1463	87	184	11	1.9866	0.5770
Mean[b]			1.1599				1.8626	0.6237

[a] Fledglings per hatchling, or the hatchling to fledgling survival rate (S_h), is calculated as the ratio fledglings/nest:hatchlings/nest.
[b] Mean excludes 1982.

Second Winter Survival – Adult Survival:

Table 3. Survival (S; subscripts indicate age class) parameters of the California least tern for normal and El Niño/Southern Oscillation (ENSO) years (95% CI in parentheses).

Parameter		Normal years	ENSO years	Reference or calculation
S_h S_0 S_1 S_2	(hatchling to first breeding)	0.16 (0.13 to 0.18)	0.03 (0.01 to 0.05)	Massey et al. (1992)
S_2 and S_3		0.81 (0.72 to 0.87)	0.81[a]	Massey et al. (1992)
S_{4+}		0.92 (0.88 to 0.95)	0.79 (0.68 to 0.89)	Massey et al. (1992)
S_h	(hatchling to fledgling)	0.6237	0.2700	Massey et al. (1992) and Minsky (1984)
S_0 S_1	(fledglings to age 2)	0.3167	0.1372	$= (S_h S_0 S_1 S_2)/(S_h S_2)$
S_0 and S_1		0.5627	0.3704	$= \sqrt{S_0 S_1}$

[a] Massey et al. (1992) estimated S_2 and S_3 for ENSO years as 0.82 (95% CI = 0.66, 0.96). Because survival is unlikely to increase in an ENSO year, in this model we assumed these survival rates for ENSO years to be equal to the average they estimated for normal years (0.81).

Table 5. Survival rate of adult California least terns at Camp Pendleton, California, USA (from Collins et al. 1998).

Yr	Survival
1987	0.926
1988	0.921
1989	0.920
1990	0.866
1991	0.899
1992	0.781
1993	0.894
1994[a]	0.731
Mean	0.8867
SD	0.0510
CV	0.0576

[a] The estimate for 1994 is the product of survival rate and resighting probability, and is not used in calculating the average and the variability.

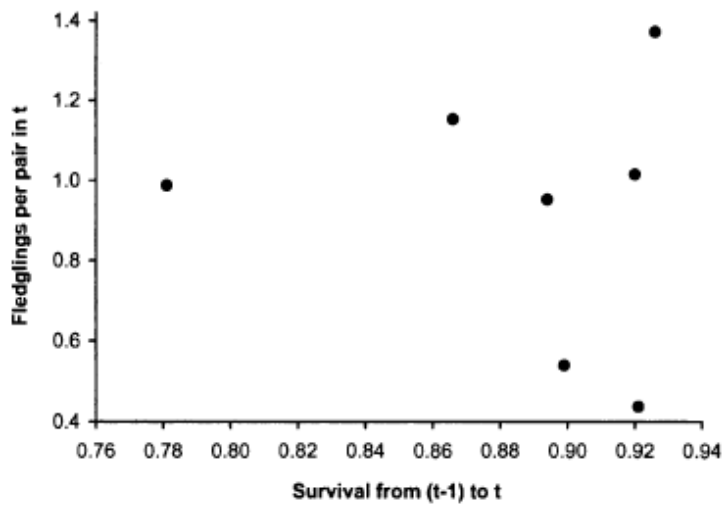


Fig. 4. Number of fledglings per pair of California least tern in year *t*, in the North San Diego population (California, USA), as a function of the adult survival rate from year $t-1$ to t in Camp Pendleton (which is part of this population) for years 1988–1994. The coefficient of correlation is –0.12 (SE = 0.44).

G.7.8 Brooks et al. 2013

First Winter Survival:

Table 3. Daily survival rate (SE) and probability of success for Least Tern (*Sternula antillarum*) nests in Cape Romain National Wildlife Refuge, South Carolina, 2009-2010. Probability of success is the daily survival rate raised to an exponent equal to 21 (incubation days). All estimates were calculated by using coefficients and standard errors of the most-supported model from Table 1.

Variable	Daily Nest Survival	Probability of Success
Colony site		
Lighthouse Island	0.916 (0.006)	0.157
Cape Island	0.930 (0.003)	0.218
Middle White Banks	0.996 (0.001)	0.926
Year		
2009	0.959 (0.003)	0.412
2010	0.977 (0.001)	0.613
Predation risk		
No	0.986 (0.001)	0.749
Yes	0.916 (0.004)	0.160

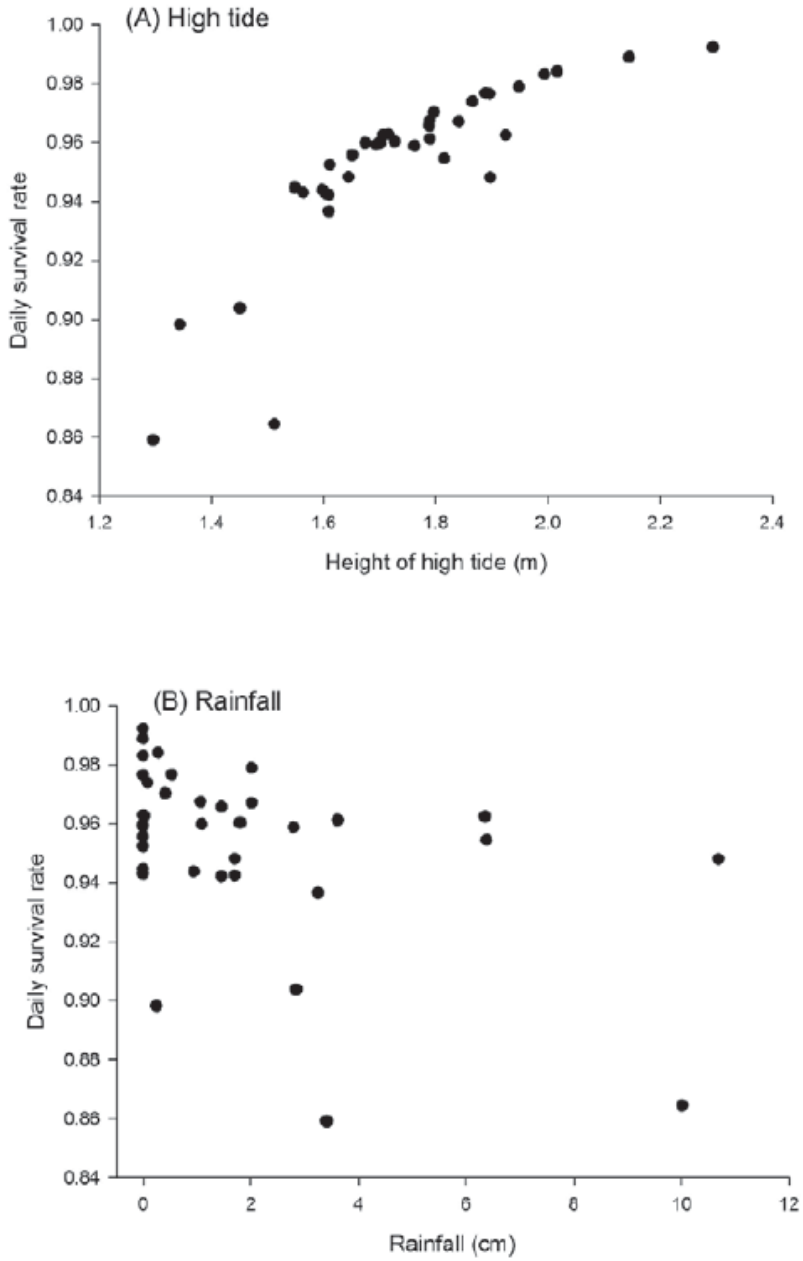


Figure 3. Daily survival rate (DSR) of Least Tern (*Sternula antillarum*) chicks in Cape Romain National Wildlife Refuge, South Carolina, 2009-2010.

G.7.9 Kirsch 1996

First Winter Survival:

Table 14. Comparisons between sandbars and sandpits of instantaneous rates of laying, egg, and chick mortality for least tern young along the lower Platte River, Nebraska, 1987–90.

	Habitat								
	Sandbar			Sandpit					
Mortality rate	n	$\bar{x}$	SE	n	$\bar{x}$	SE	Student's t	df	P
Laying	19	0.216	0.017	21	0.207	0.024	0.320	38	0.751
Egg	19	0.548	0.083	21	0.471	0.082	0.661	38	0.513
Chick	18	0.873	0.077	19	0.722	0.076	1.394	35	0.174
Overall	19	1.225	0.066	21	1.082	0.048	1.785	38	0.087

G.8 Site Fidelity, Immigration, and Emigration

G.8.1 Lott et al. 2013

Findings:

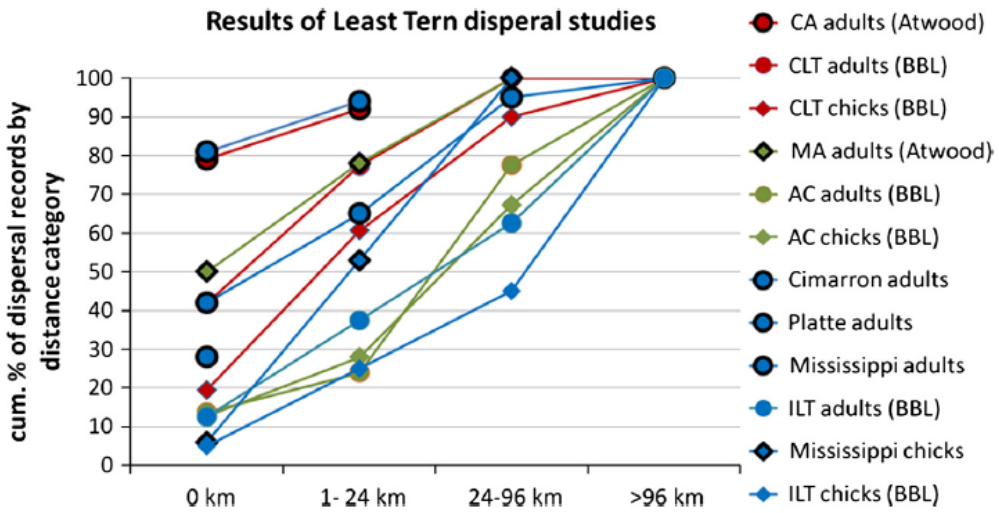


Figure 3. Dispersal distance frequencies, by age class, from published literature (Atwood and Massey 1988, Boyd 1993, Lingle 1993; Renken and Smith 1995a; Akcakaya et al. 2003) and 1261 band recovery records from 1923 through 2011, obtained from the U.S. Geological Survey, Patuxent Wildlife Research Center, Bird Banding Laboratory. Published studies had variably sized study areas, all of which were restricted to <96 km from the original banding site. These studies would only produce long-distance band recoveries in the unlikely event that banded individuals were found dead outside of these small study areas or captured again by researchers in distant locations.

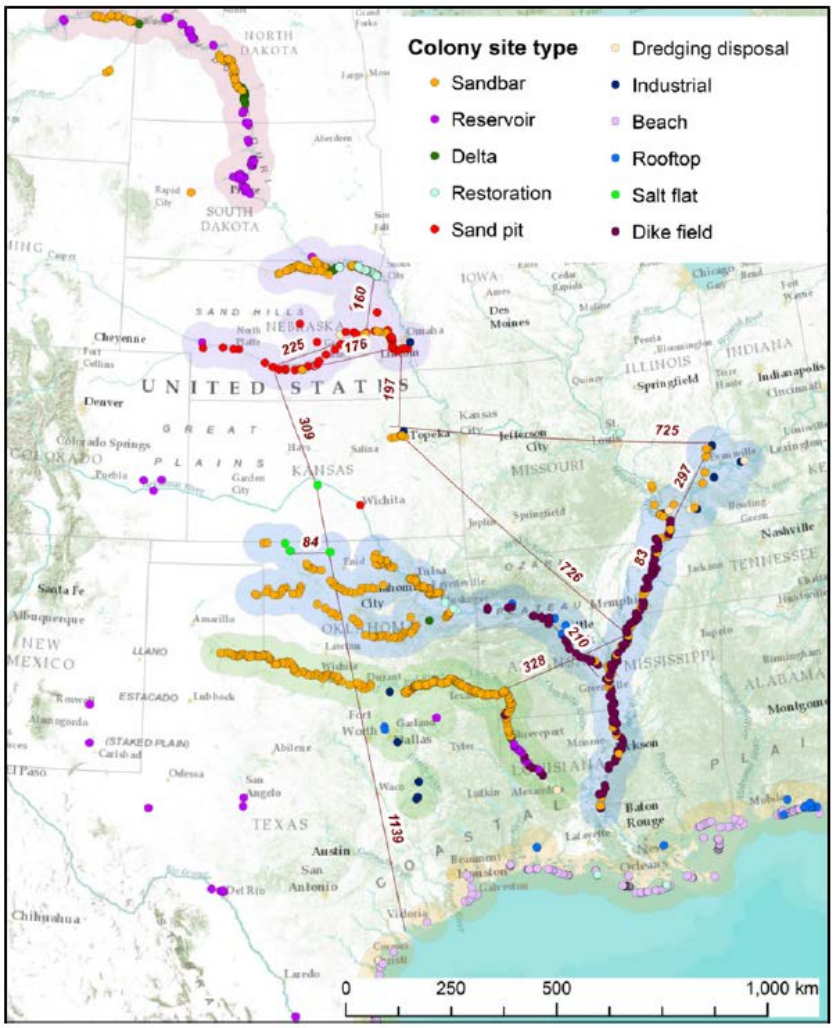


Figure 2. This map displays the 4 major ILT "populations" identified in our GIS analysis: Blue shading, the Lower Mississippi, Arkansas, Canadian, and Cimarron rivers; Green, the Red and Trinity Rivers; Purple, the Platte, Niobrara, and Missouri Rivers (below Fort Randall and Gavins Pt. dams); Pink, Northern Missouri River. The Gulf Coast Least Tern population is shown (beige shading) for spatial reference. Brown lines labeled by brown numbers illustrate 13 ILT band recoveries reflecting dispersal distances >80 km (unpublished data from the U.S. Geological Survey, Patuxent Wildlife Research Center, Bird Banding Laboratory). The true frequency of dispersal events >80 km, which would connect many ILT populations and some ILT populations with Gulf Coast populations, is unknown due to the absence of unbiased dispersal information for this species.

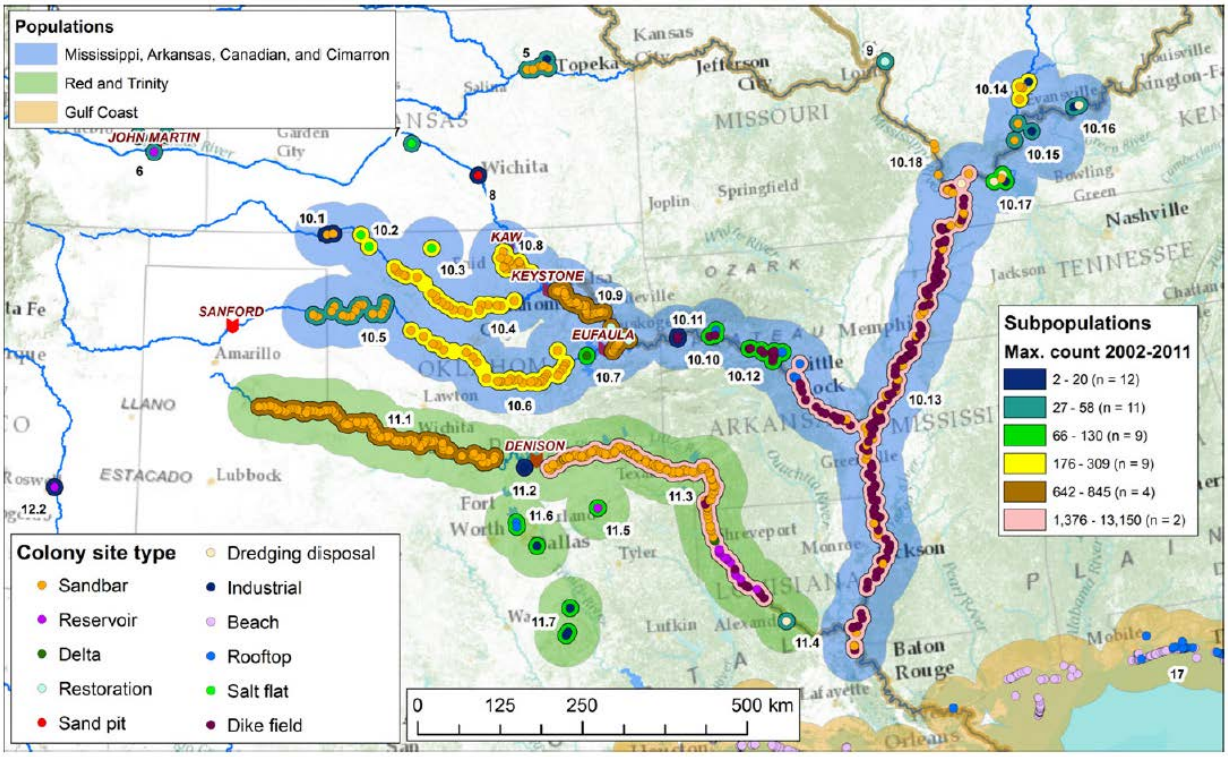


Figure 4. Detailed map of ILT populations 5–11 (and their constituent numbered subpopulations) spanning the Southern Great Plains and the Lower Mississippi River valley. See Table 1 for more information. All-caps text above red chevrons gives major dam names.

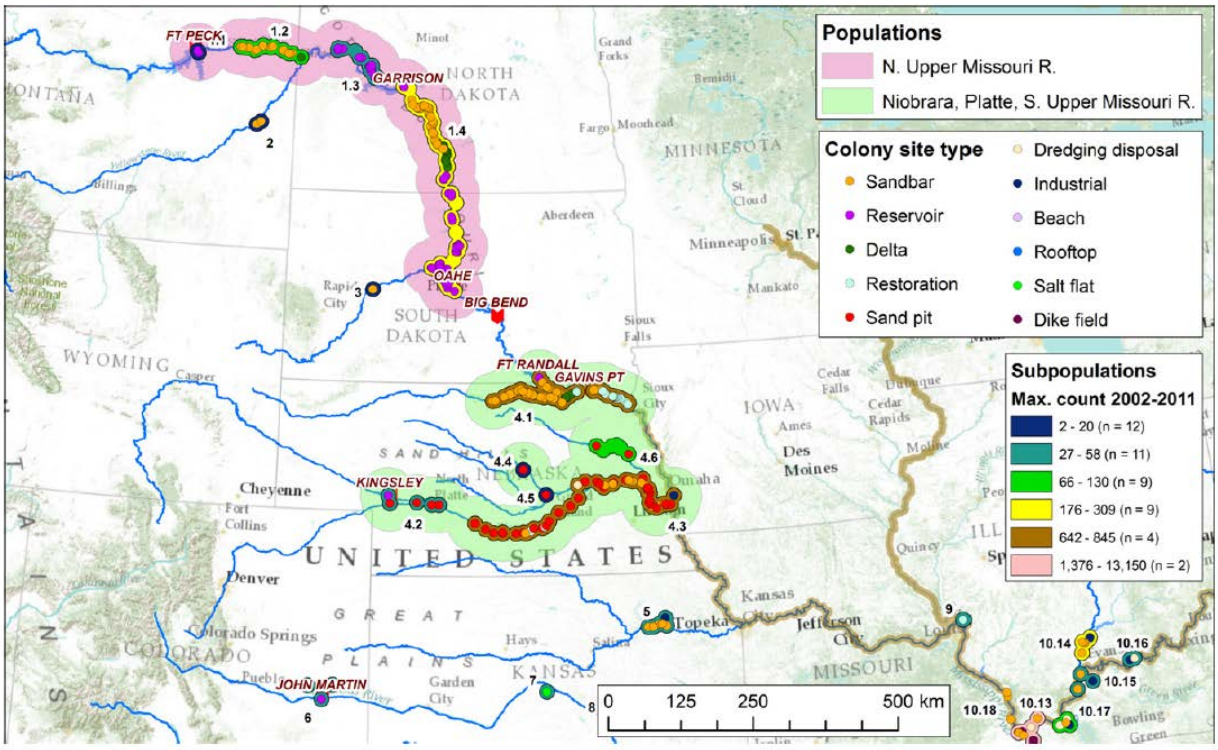


Figure 5. Detailed map of ILT populations 1-4 (and their constituent numbered subpopulations) spanning the Northern Great Plains. See Table 1 for more information. All-caps text above red chevrons gives major dam names.

G.8.2 Atwood and Massey 1988

Findings:

TABLE 1. Numbers of Least Tern chicks banded in California, 1973 to 1984.

Colony[a]	Year											
	73	74	75	76	77	78	79	80	81	82	83	84
HB	4[b]	—	—	—	35	115	98	105	66	61	69	104
VB	—	—	—	—	—	112	168	218	213	178	173	47
TI	—	—	5	44	111	—	—	—	19	48	40	7
MISC	87	44	21	301	151	159	355	360	401	277	633	596
Total	91	44	26	345	297	386	621	683	699	564	915	754

[a] Colony site abbreviations as follows: HB = Huntington Beach; VB = Venice Beach; TI = Terminal Island; MISC = colonies other than HB, VB, or TI.
[b] Number of chicks banded.

TABLE 2. Fidelity to natal colony sites in Least Terns.

Colony[a]	Year	Total recoveries	Observed recoveries from natal colony site	Expected recoveries from natal colony sites[b]	χ^2
HB	1983	24	11	4.30	12.72**
HB	1984	27	9	4.54	5.36*
HB	1985	15	5	2.52	2.93
HB	1986	26	9	3.69	8.90**
VB	1983	52	30	16.74	15.45**
VB	1984	44	28	13.82	21.21**
VB	1985	44	28	13.33	23.16**
VB	1986	50	30	14.55	23.13**

[a] Colony abbreviations as follows: HB = Huntington Beach, VB = Venice Beach.
[b] Assuming random distribution of breeders relative to their natal colony sites.
[c] Based on 2 × 1 contingency table comparing observed vs. expected numbers of recoveries at natal and nonnatal colony sites. Chi-square significance levels (df = 1) indicated as follows: ** = $P < 0.01$; * = $P < 0.05$.

TABLE 3. Annual and intercolony variation in site fidelity in breeding Least Terns.

Colony	Year				$\bar{x}$
	1983[a]	1984	1985	1986	
Venice Beach	86 (21)	58 (59)	82 (54)	86 (66)	78
Huntington Beach	54 (13)	42 (31)	52 (31)	79 (19)	57
Terminal Island	50 (22)	38 (34)	36 (50)	47 (38)	43

[a] Percent of the previous year's banded population (n indicated in parentheses) that was documented as breeding during the specified year. Thus, of 21 banded individuals that nested at Venice Beach in 1982, 86% returned to breed at this site in 1983.

G.8.3 Akcakaya et al. 2003

Findings:

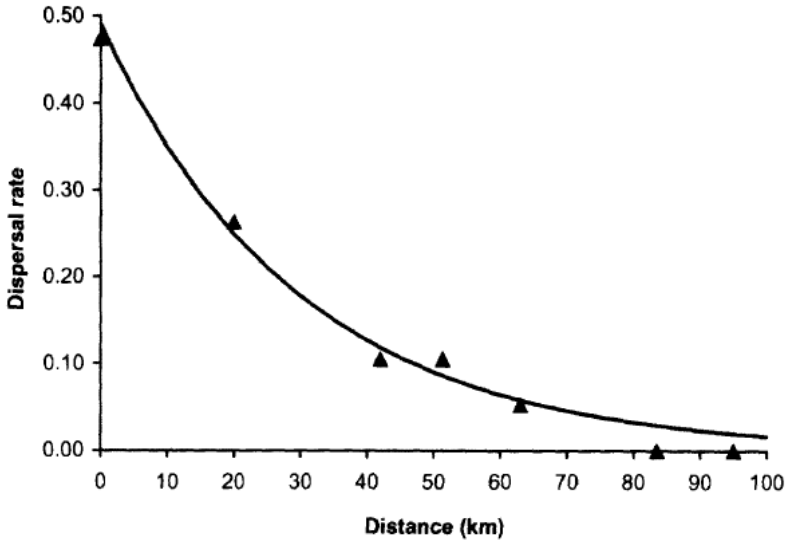


Fig. 6. Dispersal-distance function based on data from a Massachusetts, USA, population of least tern (data from Atwood 1999). The dispersal model assumes a maximum dispersal distance of 80 km. The curve shows the function $y = a \cdot \exp(-x / b)$, where $a = 0.49$ and $b = 29$ were estimated from a regression of ln(dispersal rate) on distance in km ($R^2 = 0.97$).

Table 6. Dispersal rates used in our California least tern metapopulation model: percentage of individuals moving to other populations in 5 distance classes.

	Dispersal model		
Distance from source population (km)	Low	Medium	High
0 (remain in the source population)	90	70	50
<30 km	10	15	25
30–60 km	0	10	15
60–90 km	0	5	10
>90 km	0	0	0

G.9 Population Trends through Time

G.9.1 Boyce et al. 2002

Observed Trend:

Table 1. Least tern numbers and estimates for model parameters of the central and lower reaches of the Platte River.

Year	Parameter estimate	Number of terns	
		Central	Lower
1986		114	324
1987		171	420
1988		123	497
1989		157	409
1990		149	361
1991		197	487
1992		191	427
1993		178	451
1994		169	426
1995		119	190†
1996		134	290
	λ^*	1·050	1·035
	k	5·0625	6·073
	r	0·016	0·034
	V_r	0·0556	0·0316

†This estimate was excluded from analyses because Nebraska Game and Parks Commission was unable to survey least terns nesting on sandbars in 1995 before the river flooded.

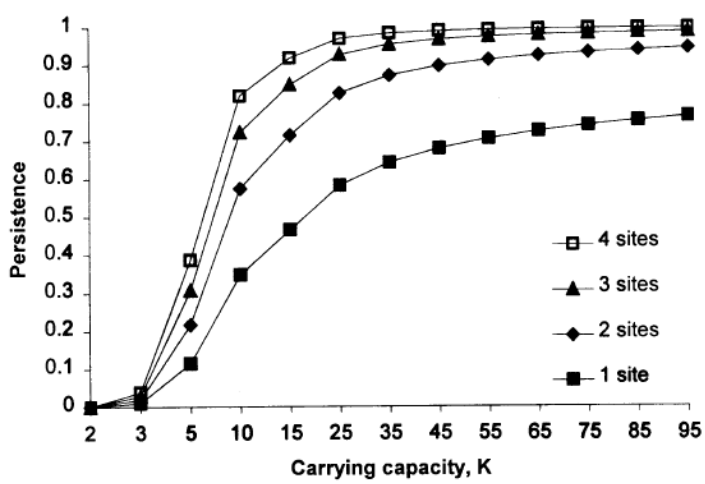


Figure 2. Effects of carrying capacity (K) and the number of populations (sites) on the probability of persistence for interior least terns.

G.9.2 Brown et al. 2001

Observed Trend:

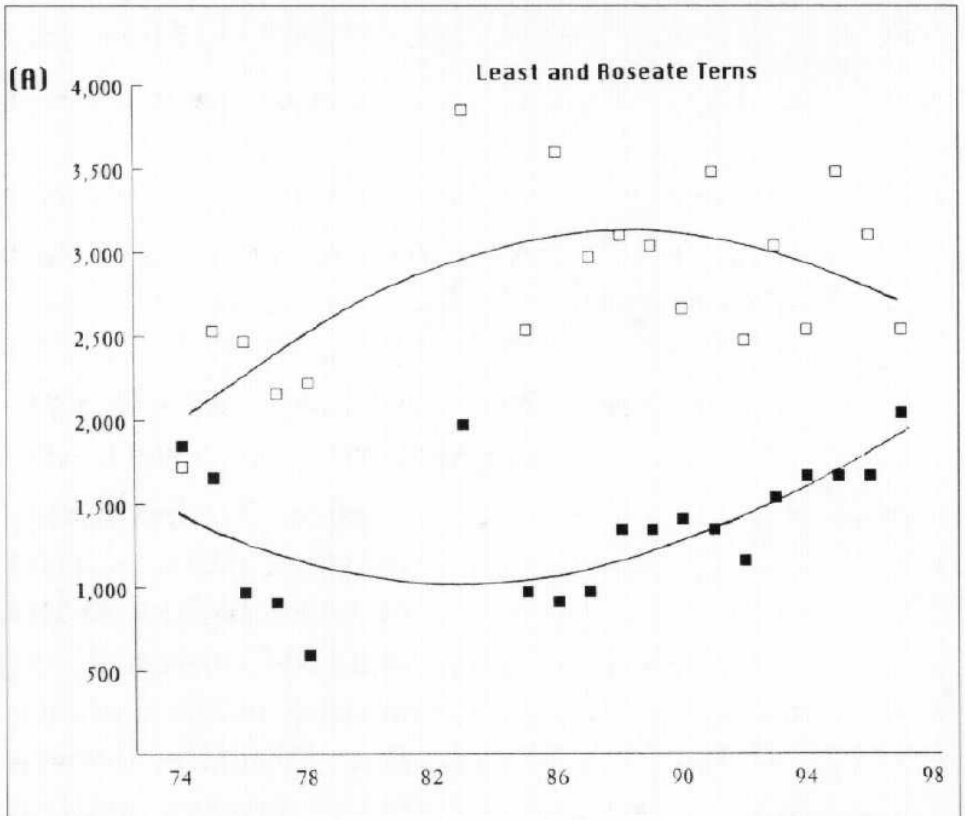


Figure 4. Estimates (nesting pairs, y-axis) of (A) Least (open boxes) and Roseate (solid boxes) Terns and (B) Black Skimmers nesting on Long Island, from 1974 to 1997. The lines represent the regression models that best fit these data ($P < 0.05$).

G.9.3 Burger 1989

Observed Trend:

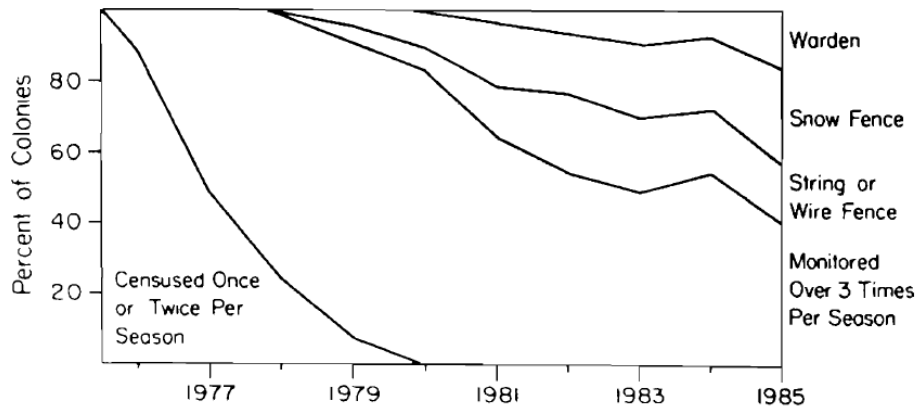


Figure 1. Changes in population levels, colony numbers and reproductive success of least terns in New Jersey from 1976-1985.

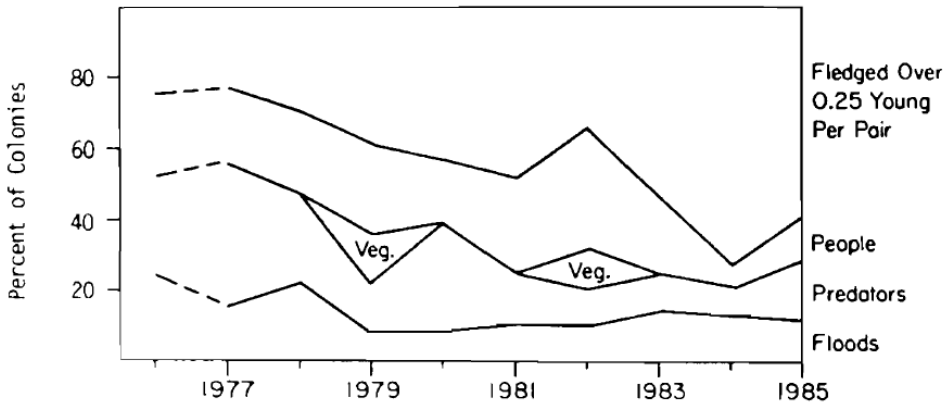


Figure 2. Causes of colony failures of least terns from 1976-1985. A colony failure occurred when fewer than 0.25 young were fledged per pair.

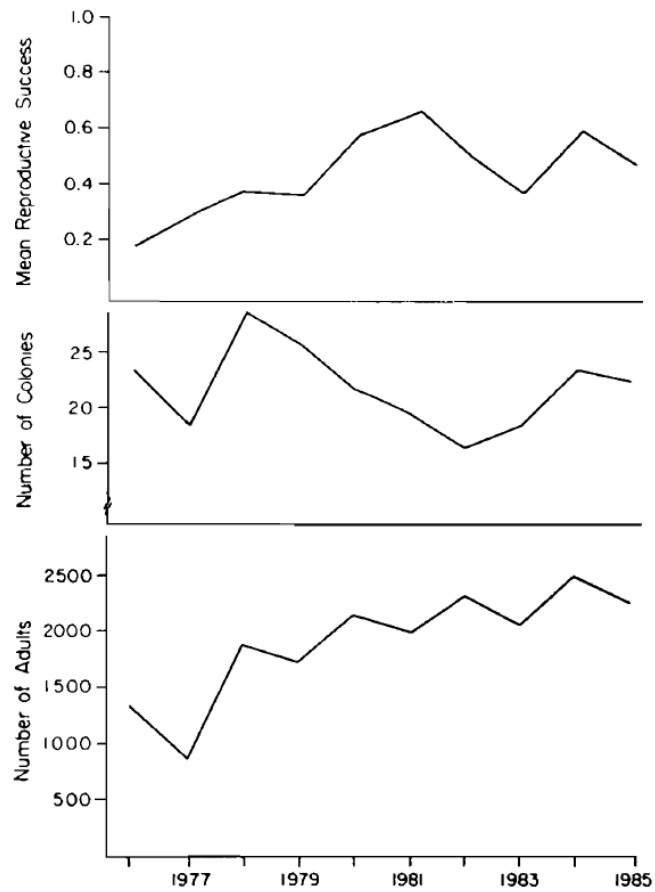


Figure 3. Percent of least tern colonies in New Jersey exposed to different management practices, 1976-1985.

Table 3. *Results of decoy experiments to attract Least Terns to abandoned sites.*

		1983		1984	
Island	Date Colony was Last Used	Number Nesting	Success[a]	Number Nesting	Success
Experimental Islands					
Mike's	1980	20	0.85	42	1.19
Island Beach[b]	1981	1[c]	—		
Cedar Bonnet	1981	7	1.0	62	0[d]
Experimental Plot[e]	1980	3[f]	0	12	1.20

[a] Number fledged per pair.
[b] No experimentation in 1984.
[c] Landed but did not nest.
[d] A dog in the colony one weekend killed all young chicks and the adults deserted.
[e] At Brigantine beach.
[f] Washed out by heavy rain.

G.9.4 Palacios and Mellink 1996

Observed Trend:

TABLE 1. Number of pairs of Least Terns and type of habitat at each known nesting site, in the Gulf of California. Before 1985, the number in parenthesis indicates the year of observation. A dash indicates that no information is available and an asterisk indicates that adults were present but no breeding evidence was found. The names of the colonies have been shortened for convenience; see Appendix 1 for complete names. Substrate: BB = barrier beach, SB = sandy beach, DS = dredge spoil, MF = mudflat, SM = sand bars with shell fragments or shell mounds on extensive mudflat and SF = saltflat.

Colony	Year <85	85	86	87	88	89	90	91	92	93	94	Substrate
Baja California Sur												
San José del Cabo												
San José	15(23)	—	—	3	—	—	10	—	7	—	—	BB
Punta Arena de la Ventana												
Ventana	—	—	—	—	20	4	—	—	2	—	—	SB
Ensenada de La Paz												
Afegua	—	163	147	122	25	19	—	4	0	—	—	DS
Fidepaz	—	12	5	1	3	67	9	3	0	—	—	DS
Chametla	—	20	—	—	—	2	44	15	4	—	—	SM
Centenario	—	2	0	0	0	0	0	0	0	—	—	SM
Zacatecas	10(84)	—	1	—	—	8	—	3	0	—	—	SM
Baja California												
Laguna Percebú												
LP Sur	+(83),17(84)	31	—	—	—	20	12	31	12	30	8	BB
LP Oeste	—	—	—	—	—	—	8	4	37	0	0	MF
LP Norte	—	15	—	—	—	15	4	18	27	30	7	BB
Punta Estrella												
Estrella	—	—	—	—	—	—	—	—	6	5	11	BB
Esteros La Bolsa												
La Bolsa	—	—	—	—	—	—	—	—	11	—	—	BB
Delta del Río Colorado												
Isla Montague												
Montague	—	—	—	—	—	—	—	5	20	6	10	SM
El Doctor	—	—	—	—	—	—	—	—	—	—	4	SF
La Flor*	—	—	—	—	—	—	—	—	—	—	2	SF
Sonora												
La Salina												
Salina N	—	—	—	—	—	—	—	—	1	—	—	SF
Salina S	—	—	—	—	—	—	—	2	—	—	—	SF
Estero Morúa												
???	+(78)											
Morúa P	—	—	—	—	—	—	—	—	16	+	—	BB
Morúa D	—	—	—	—	—	—	—	—	15	—	1	BB
Bahia de San Jorge												
La Pinta	—	—	—	—	—	—	—	—	—	—	+	BB
Purinera	—	—	—	—	—	—	—	60	110	174	83	BB
Francisquito	—	—	—	—	—	—	+	16	36	3	4	BB
Estero Los Tanques												
Tanques*	—	—	—	—	—	—	—	—	4	—	—	BB
Lobos	—	—	4	—	—	—	—	—	—	—	—	SB
Estero del Soldado												
Soldado	25(79)	—	—	—	—	—	—	—	15	—	—	MF

TABLE 1. Continued.

Colony	Year											Substrate
	<85	85	86	87	88	89	90	91	92	93	94	
Estero El Tóbari												
Estero SJ	—	—	—	—	—	—	—	—	—	—	15	MF
Salitral SJ	—	—	—	-	—	—	—	—	—	—	15	SF
Tóbari I*	—	—	—	—	—	—	—	—	—	—	14	BB
Tóbari N*	—	—	—	—	—	—	—	—	—	—	17	BB
Tóbari S*	—	—	—	—	—	—	—	—	—	—	41	BB

G.9.5 Parnell et al. 1997

Observed Trend:

Table 1. Status of colonial waterbirds nesting in North Carolina, 1900 to present.

			Census year									
			1977		1983		1988		1993		1995	
Species	Nesting 1900 to 1919[1]	First nesting record	No. sites	No. nests	No. sites	No. nests	No. sites	No. nests	No. sites	No. nests	No. sites	No. nests
Brown Pelican	no record	1929[2]	2	82	5	1333	7	2451	7	3327	5	4031
Green-backed Heron	"common"	pre-1900	7	42	3	24	3	22	3	8	0	0
Little Blue Heron	"abundant"	pre-1900	17	802	12	1164	12	544	15	1727	12	1407
Cattle Egret	no record	1956[3]	9	1137	7	1448	8	1919	12	2271	7	1517
Great Egret	1 site 20 pr	pre-1900	15	494	17	839	13	673	21	1945	19	1901
Snowy Egret	10-12 pr 1908	pre-1900	17	1034	13	715	11	497	20	904	13	672
Tricolored Heron	1000+ pr	pre-1900	17	1479	15	1399	10	869	19	1938	12	1716
Black-cr N.-Heron	common	pre-1900	10	230	13	269	6	185	14	251	13	204
Yellow-cr N.-Heron	no record	1939[4]	0	0	2	9	1	12	1	18	1	18
Glossy Ibis	no record	1940[5]	12	404	7	291	4	84	13	502	10	279
White Ibis	no record	1950[6]	3	1939	2	3825	2	6312	6	10453	7	9428
Herring Gull	no record	1962[7,10]	10	433	15	451	13	965	16	960	18	516
Great Black-b. Gull	no record	1972[8]	3	10	?	?	1	3	7	47	8	92
Laughing Gull	700 chicks 1908	pre-1900	22	9369	16	22903	14	18978	19	17970	16	23667
Gull-billed Tern	1pair	1909	20	268	10	226	15	151	8	155	9	249
Forster's Tern	no record	1972[9]	29	1138	15	936	13	958	25	1548	20	1117
Common Tern	abundant	pre-1900	45	2761	28	2227	24	2610	27	2122	22	1699
Least Tern	4000-5000 prs	pre-1900	38	1925	31	1633	19	1528	21	2154	29	1931
Royal Tern	2000 adults 1907	pre-1900	9	9755	8	17029	8	11793	7	14611	7	14150
Sandwich Tern	108 eggs 1908	pre-1900	6	1190	7	1850	6	1465	5	2700	6	2905
Caspian Tern	no record	1972[11]	0	0	2	6	1	11	2	33	1	37
Black Skimmer	breeds at several sites	pre-1900	26	976	18	732	18	643	18	1084	17	819

[1]Pearson *et al.* 1919
[2]Pearson *et al.* 1942
[3]Quay and Adams 1956
[4]Stephens 1950
[5]Quay and Adams 1956, Davis 1957
[6]Stephens 1950
[7]Hailman 1963
[8]Parnell and Soots 1975
[9]Fussell 1974
[10]Davis 1993
[11]Parnell and Soots 1976
?See species account in text

Table 2. Changes in site types and numbers of occupied sites for colonially nesting birds, 1977 to 1995.

	Year				
Site type	1977	1983	1988	1993	1995
Barrier beach	24	20	17	22	26
Natural estuarine	42	22	23	36	34
Dredged undiked	36	21	16	18	19
Dredged diked	22	18	8	5	6
Other	0	0	0	1	3
Total sites	124	81	64	82	88

Table 3. Turnover rates for colonial waterbirds in North Carolina 1983 - 1995

	1983-1988							1988-1993							1993-1995							
	Sites				Sites			Sites				Sites			Sites				Sites			
Species	1983	d	r	n	1988	T_5	T	1988	d	r	n	1993	T_5	T	1993	d	r	n	1995	T_2	T	MAT
Brown Pelican	5	1	4	3	7	0.31	0.06	7	1	6	1	7	0.14	0.03	7	3	4	1	5	0.31	0.16	0.08
Great. Egret	17	7	10	3	13	0.32	0.06	13	1	12	9	21	0.25	0.05	21	3	18	1	19	0.10	0.05	0.05
Snowy Egret	13	6	7	4	11	0.41	0.08	11	2	9	11	20	0.37	0.07	20	8	12	1	13	0.24	0.12	0.09
Cattle Egret	7	3	4	4	8	0.46	0.09	8	1	7	5	12	0.27	0.05	12	6	6	1	7	0.32	0.16	0.10
Little Blue Heron	12	4	8	4	12	0.33	0.07	12	6	6	9	15	0.55	0.11	15	4	11	1	12	0.18	0.09	0.09
Tricolored Heron	15	8	7	3	10	0.42	0.08	10	3	7	12	19	0.47	0.09	19	8	11	1	12	0.25	0.13	0.10
Black-cr. N.-Heron	13	10	3	3	6	0.63	0.13	6	2	4	10	14	0.52	0.10	14	4	10	3	13	0.26	0.13	0.12
White Ibis	2	1	1	1	2	0.50	0.10	2	1	1	5	6	0.67	0.13	6	0	6	1	7	0.07	0.04	0.09
Glossy Ibis	7	5	2	2	4	0.61	0.12	4	0	4	9	13	0.35	0.07	13	5	8	2	10	0.29	0.15	0.11
Laughing Gull	16	7	9	5	14	0.40	0.08	14	3	11	8	19	0.32	0.06	19	5	14	2	16	0.19	0.10	0.08
Herring Gull	15	8	7	6	13	0.50	0.10	13	5	8	8	16	0.44	0.09	16	3	13	5	18	0.23	0.12	0.10
Gull-billed Tern	10	7	3	12	15	0.75	0.15	15	12	3	5	8	0.71	0.14	8	6	2	7	9	0.76	0.38	0.22
Royal Tern	8	5	3	5	8	0.63	0.13	8	3	5	2	7	0.33	0.07	7	3	4	3	7	0.43	0.21	0.14
Sandwich Tern	7	4	3	3	6	0.54	0.11	6	3	3	2	5	0.45	0.09	5	2	3	3	6	0.45	0.23	0.14
Common Tern	28	18	10	14	24	0.61	0.12	24	14	10	17	27	0.61	0.12	27	19	8	14	22	0.67	0.34	0.10
Forster's Tern	15	11	4	9	13	0.71	0.14	13	7	6	19	25	0.65	0.13	25	14	11	9	20	0.51	0.25	0.17
Least Tern	31	20	11	8	19	0.53	0.11	19	9	10	11	21	0.50	0.10	21	10	11	18	29	0.55	0.27	0.16
Black Skimmer	18	15	3	15	18	0.83	0.17	18	13	5	13	18	0.72	0.14	18	12	6	11	17	0.66	0.33	0.21

MAT = Mean Annual Turnover, d = Deserted colony sites, r = Reused colony sites, n = New colony sites

G.9.6 Kirsch and Sidle 1999

Observed Trend:

Table 1. Estimated numbers of interior least terns throughout their interior breeding range, 1984–95.

Area	Year											
	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995
Yellowstone River, Montana; to Lake Sakakawea, North Dakota			23	20	36	29	24	16	14	19	40	21
Missouri River, Fort Peck Reservoir, Montana				4	3	4	6	10	0	7	9	2
Missouri River, Fort Peck Dam, Montana, to Lake Sakakawea, North Dakota					25	47	92	66	110	31	58	95
Missouri River, Lake Sakakawea, North Dakota					7	15	6	8	29	17	35	7
Missouri River, Garrison Dam to Lake Oahe, North Dakota	136	114	147	166	142	121	174	195	198	145	217	284
Missouri River, Lake Oahe, North Dakota and South Dakota		38[a]	30[a]	82	97	100	143	124	125[a]	160	84	
Missouri River, Fort Randall Dam, South Dakota to Niobrara, Nebraska			11	32	0	4	25	32	13	38	43	10
Missouri River, Lewis and Clark Lake, South Dakota			14	28	45	29	63	55	29	76	44	16
Missouri River, Gavins Point Dam, Nebraska and South Dakota to Ponca, Nebraska			181	232	252	210	166	193	187	272	211	93
Missouri River, Sioux City, Iowa			2	0		0		0	12		1	12
Missouri River, Council Bluffs, Iowa	18	18	28	22	22		0	20	9	0	4	4
Cheyenne River, South Dakota			31	54	27	12						32
Niobrara River, Nebraska		174		143	217			291				
Loup River, Nebraska			12[b]	100	155	29[a]	42[a]	87	188	46		150
North Platte River, Lake McConaughy, Nebraska					0	0	8	16	24	10	12	8
Central Platte River, Nebraska	24[a]	120	114	153	102	152	116	172	191	178	169	119
Lower Platte River, Nebraska	203	158[a,b]	324	426	533	436	366	284	427	451	426	180[a]
South Platte River, Nebraska and Colorado								5				
Elkhorn River, Nebraska		2	8	4				26	35	38	24	35
Arkansas River watershed reservoirs, Colorado			6	10	6	24	30	46	42	30	22	
Quivira NWR, Kansas	44	48	48	54		46	68	54		48	46	50
Cimarron River, Kansas and Oklahoma	108[a]	82[a]	150[a]	132[a]		385[d]	386[a]	67[b]	452	16[a]	22[a]	16[a]
Optima NWR, Oklahoma	30	46	52	64	38	0	0	0	16			
Salt Plains NWR, Oklahoma	115[c]		130[c]	210[c]		220	240	82	158[c]	168	90	200
Arkansas River, Oklahoma	3[a]	43	88	20[a,c]	161[a]	45[a,c]	209[a]	320[c]	368[c]	447	471	224
Arkansas River, Arkansas		80	0	90	119							
Canadian River, Oklahoma		127	182	136	74[a]	7[a]	188	38[a]		135[c]	78	120
Canadian River, Texas		35		8	16	12	14					
Red River, Oklahoma and Texas				12[a]	16[a]			333[c]			243[a,c]	
Prairie Dog Town Fork, Red River, Texas		44	50	12	16	12	18					
Mississippi River, Illinois, Missouri, Kentucky, Mississippi, Arkansas, Louisiana		568	2,470	2,488	2,356	2,503	5,038	4,297	3,653	4,589[c]	6,776	6,971
Ohio River, Kentucky, Illinois, Indiana	31						0	0			44	
Wabash River, Gibson Lake, Indiana			4	4	0	6	6	10	9	34	30	24
Bitter Lake NWR, New Mexico	10	6	8	6	6	6	6	10	12	14	11	14
Rio Grande, Falcon Reservoir, Texas		500	150	62	164	426	386			768[c]		
Rio Grande, Lake Casa Blanca, Texas		5		14	50	20						
Rio Grande, Amistad Reservoir, Texas		20	9		14	30						
Dallas County, water treatment plant, Texas									20[e]	20[e]	20[e]	20[e]
Total	722	2,208	4,242	4,745	4,669	4,933	7,789	6,833	6,319	7,606	8,971	8,859

[a] Coverage was incomplete, and data from these surveys were not included in the statistical analysis unless counts were within 75% of the mean for years of complete surveys.
[b] Counts from aerial surveys were not included in analysis, because they are known to be underestimates.
[c] Midpoints of reported numbers are listed and used in analysis.
[d] A complete survey of the Cimarron River was done in 1989; however, this year was deleted from analysis of trend because other surveys of this area were incomplete.
[e] Approximate counts (surveys were cursory).

Table 2. Recovery Plan (USFWS 1990) population objectives for specified areas and survey data from local areas defined in this study for the interior population of least terns, 1986–95.

Recovery Plan		Survey data			
Area	Population objective	Area	$\bar{x}$	SD	No. years
Missouri River system, Montana	50	Fort Peck Reservoir, Montana[a]	5	3.3	9
		Fort Peck Dam, Montana, to Lake Sakakawea, North Dakota[a]	65	32.4	9
		Yellowstone River,[a] Montana	24	8.4	10
Missouri River system, North Dakota	250	Lake Sakakawea,[a] North Dakota	16	11.1	8
		Garrison Dam to Lake Oahe, North and South Dakota	179	47.4	10
Missouri River system, South Dakota	680	Missouri River System, South Dakota			
Lake Oahe, North and South Dakota	100	Lake Oahe, North and South Dakota	114	28.2	10
Fort Randall Dam, South Dakota to Niobrara, Nebraska[a]	80	Fort Randall Dam, South Dakota to Niobrara, Nebraska	21	15.0	10
		Lewis and Clark Lake,[a] South Dakota and Nebraska	40	20.3	10
Gavins Point Dam, South Dakota and Nebraska to Ponca, Nebraska	400	Gavins Point Dam, South Dakota and Nebraska to Ponca, Nebraska	200	49.8	10
Other Missouri River sites	20	Cheyenne River, South Dakota	24	10.4	3
		Sioux City, Iowa[a]	4	5.6	7
		Council Bluffs, Iowa[a]	12	10.9	9
Niobrara River, Nebraska	200	Niobrara River, Nebraska	217	74.0	3
Platte River,[d] Nebraska	750	Lower Platte,[a] Nebraska.	408	73.7	9
		Central Platte,[a] Nebraska	147	31.5	10
		North Platte River, Lake McConaughy,[a] Nebraska	8	8.1	10
		South Platte,[a] Colorado and Nebraska	4	2.1	2
		Elkhorn River,[a] Nebraska	21	13.5	7
Loup River, Nebraska	170	Loup River, Nebraska	121	52.4	6
Lower Mississippi and Ohio rivers, Illinois, Missouri, Kentucky, Mississippi, Arkansas, Louisiana	2,500	Lower Mississippi River, Illinois, Missouri, Kentucky, Mississippi, Arkansas, Louisiana	4,283	1,762.3	9
		Ohio River, Illinois, Indiana, Kentucky	15	25.4	3
		Wabash River, Gibson Lake,[a] Indiana	12	12.2	10
Arkansas River, Oklahoma	250	Arkansas River, Oklahoma	401	69.6	4
Arkansas River, Arkansas	150	Arkansas River, Arkansas	104	20.5	2
		Arkansas River watershed reservoirs, Colorado[a]	23	15.5	8
Quivira NWR, Kansas	100	Quivera NWR, Kansas	53	7.4	7
Salt Plains NWR, Oklahoma	300	Salt Plains NWR, Oklahoma	161	63.7	7
Cimarron River, Kansas and Oklahoma	400	Cimarron River, Kansas and Oklahoma	280	162.9	4
Canadian River, Oklahoma	300	Canadian River, Oklahoma	140	40.9	6
		Canadian River, Texas	12	3.4	5
Beaver and North Canadian rivers, Oklahoma	100	Optima NWR,[a,b] Oklahoma	24	27.0	7

Table 2. Continued.

Recovery Plan		Survey data			
Area	Population objective	Area	$\bar{x}$	SD	No. years
Prairie Dog Town Fork, Red River, Texas	300	Prairie Dog Town Fork, Red River, Texas	22	16.1	5
Rio Grande system	500	Rio Grande, Amistad Reservoir,[a] Texas	18	11.0	3
		Rio Grande, Lake Casa Blanca,[a] Texas	28	19.3	3
		Rio Grande, Falcon Reservoir,[a] Texas	238	159.2	5
		Dallas County, water treatment plant, Texas[a]	20	0.0	4
		Bitter Lake NWR area,[a] New Mexico	9	3.3	10
Total	7,000				

[a] Numbers of least terns were not specifically stated in the Recovery Plan for these areas.
[b] Optima NWR is located at the confluence of 2 tributaries of the Beaver River but is not actually on the Beaver River.

Table 3. Significant estimated population trends for interior least terns in local areas, 1986–95. Instantaneous rate of population change, r, was estimated as the slope of the regression of year on the natural log of least tern numbers. Finite trend was estimated as $\lambda = e^r$. A deterministic population model for least terns (Kirsch 1996) was used to calculate the "required" fledglings per pair needed to support the observed trend.

Local area	r	SD(r)	p	Finite trend λ	No. years[a]	% change /year	Required fledglings /pair
Missouri River, Garrison Dam to Lake Oahe, North Dakota	0.057	0.0161	0.024	1.059	10	5.9	0.740
Missouri River, Council Bluffs, Iowa	−0.222	0.0389	0.002	0.801	7	−19.9	0.075
Elkhorn River, Nebraska	0.212	0.0524	0.010	1.236	7	23.6	1.699
Arkansas River watershed reservoirs, Colorado	0.230	0.0835	0.033	1.259	9	25.9	1.850
Optima NWR, Oklahoma	−0.222	0.0486	0.045	0.801	4	−19.9	0.075
Wabash River, Gibson Lake, Indiana	0.305	0.0958	0.013	1.357	8	35.7	2.580
Lower Mississippi River, Illinois, Missouri, Kentucky, Mississippi, Arkansas, Louisiana	0.120	0.0216	0.001	1.128	9	12.8	1.065

[a] Number of years during 1986–95 for which survey data were available for analysis.

G.9.7 Schuetz 2011

Observed Trend:

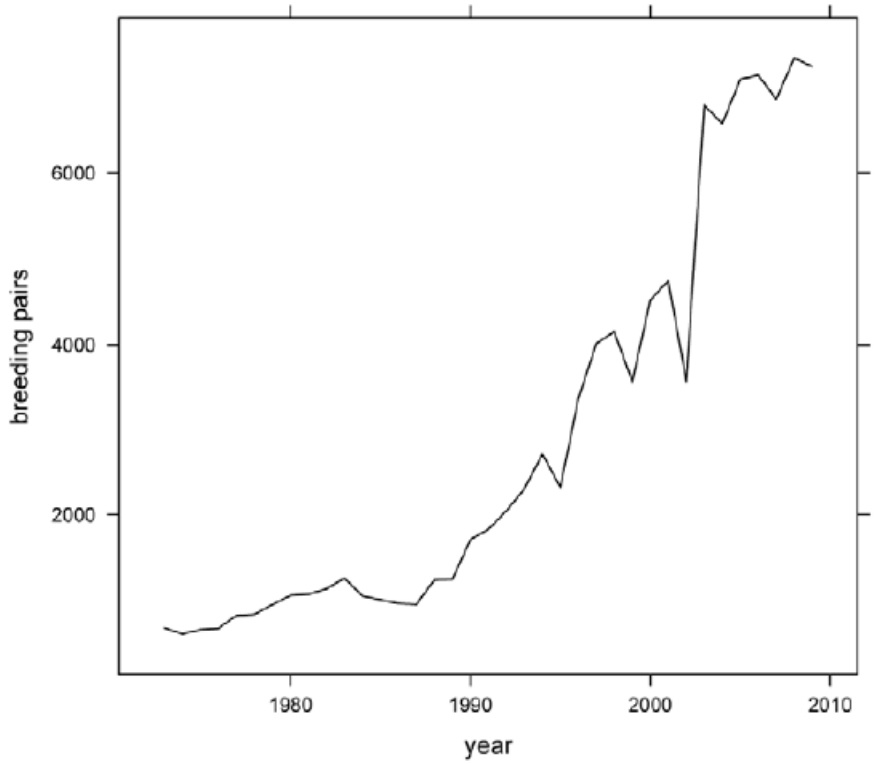


Figure 1. Number of California least tern pairs breeding in California from 1973–2009.
doi:10.1371/journal.pone.0019489.g001

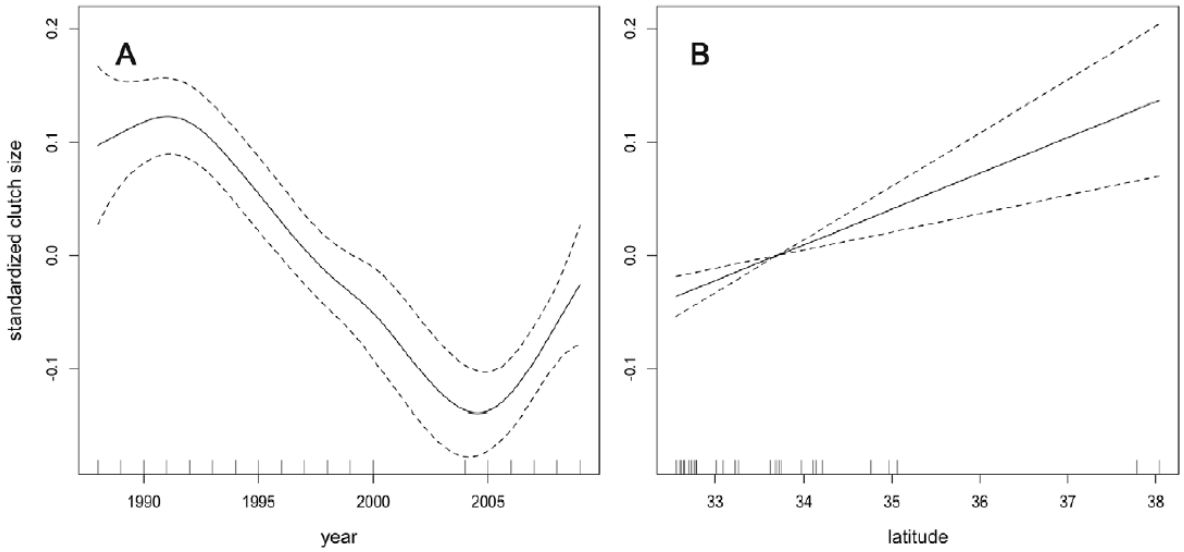


Figure 5. Partial dependence plots showing marginal effect of A) year and B) latitude from analysis of clutch size. The solid lines show fits derived from the generalized additive mixed model. Dotted lines indicate 95% confidence intervals. Clutch size data were unavailable from 2001–2003.

doi:10.1371/journal.pone.0019489.g005

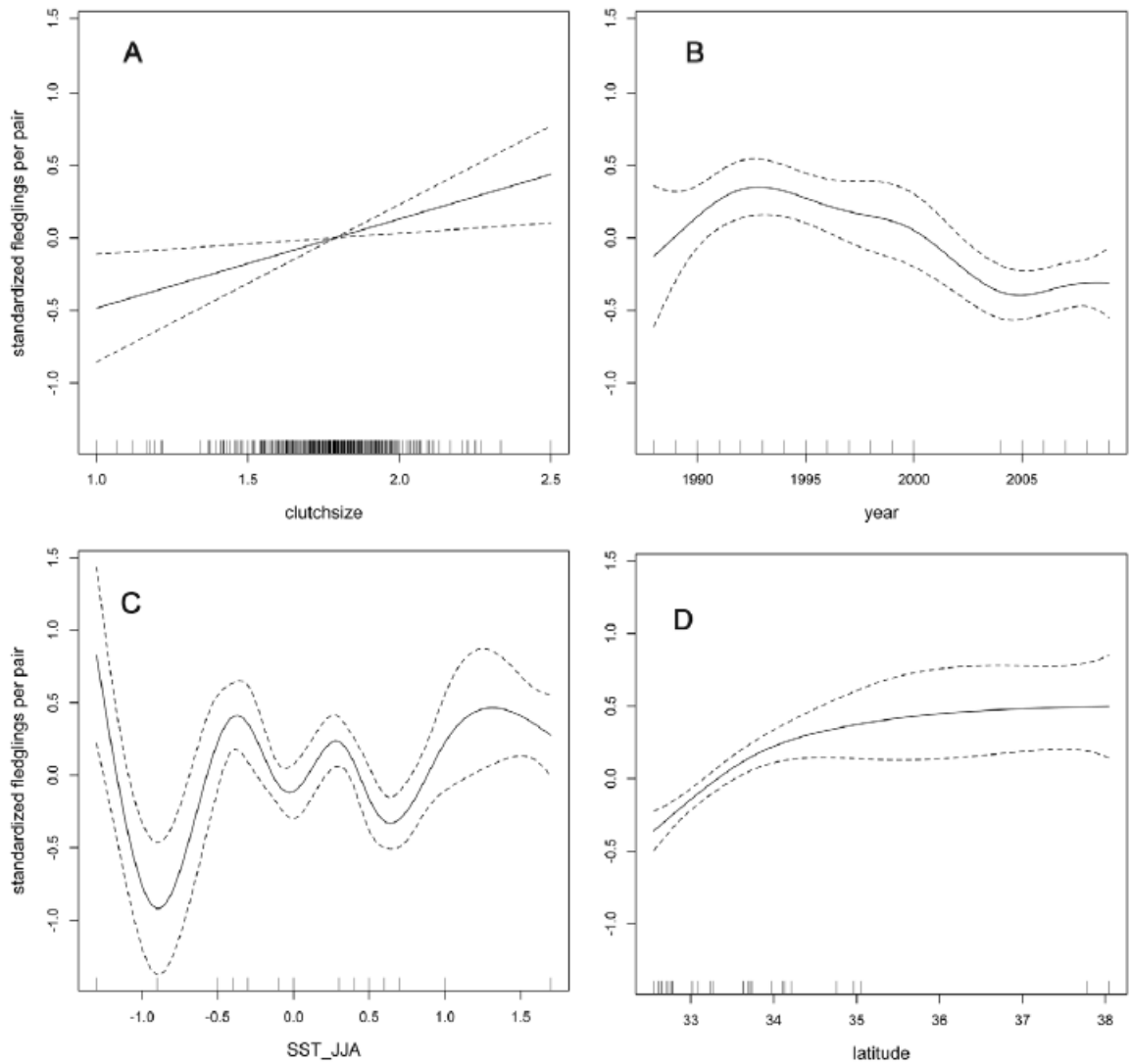


Figure 6. Partial dependence plots showing marginal effect of A) clutch size, B) year, C) SST-JJA, and D) latitude from analysis of fledgling productivity. The solid lines show fits derived from the generalized additive mixed model. Dotted lines indicate 95% confidence intervals. Clutch size data were unavailable from 2001–2003, precluding analysis of productivity in those years.
doi:10.1371/journal.pone.0019489.g006

G.9.8 Aron 2005

Observed Trend:

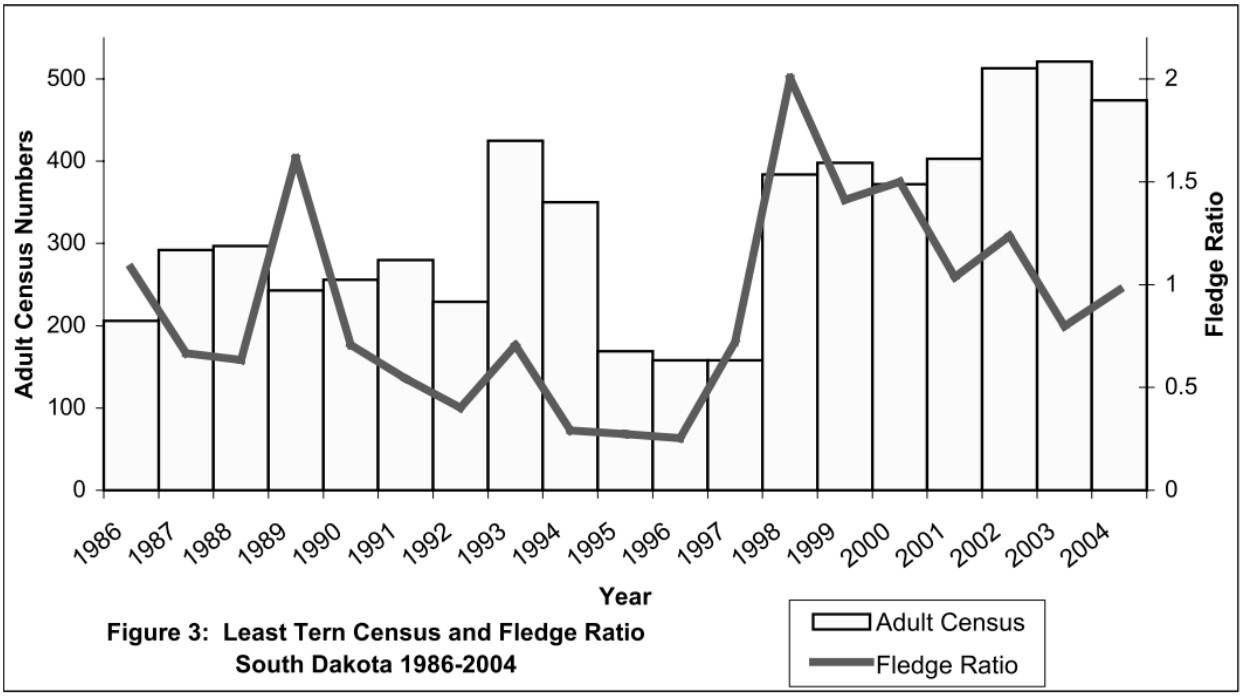


Figure 3: Least Tern Census and Fledge Ratio South Dakota 1986-2004

G.9.9 Elliott et al. 2007

Observed Trend:

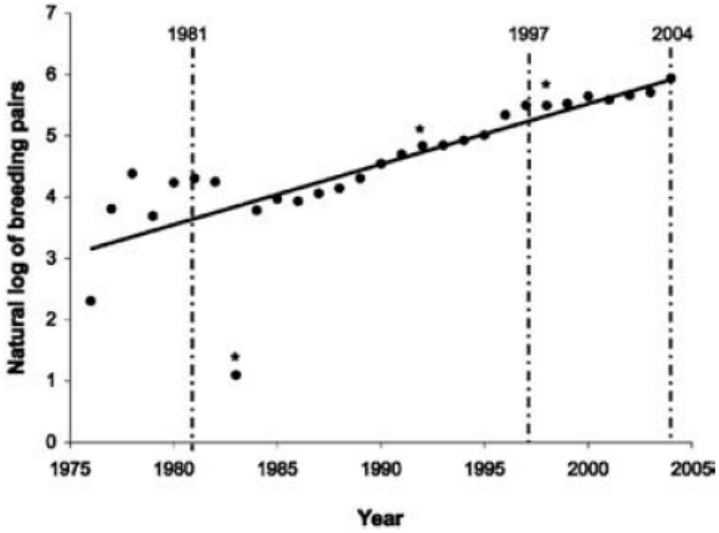


Figure 2. Estimated population size of the Alameda Point Least Tern colony from 1976-2004. (* = El Niño years. Years of important site changes are noted: 1981 = electric fence was erected; 1997 = Naval Air Station closure; 2004 = chain link fence erected.)

G.9.10 Sherfy et al. 2008

Observed Trend:

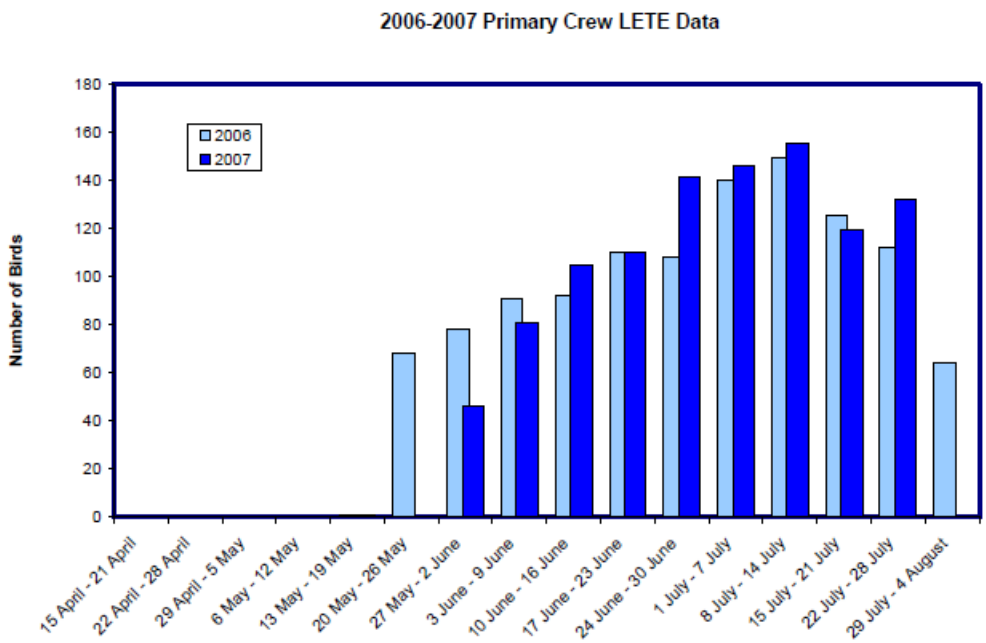


Figure 11. Number of adult least terns observed by primary crews during weekly surveys of the Garrison Reach of the Missouri River during 2006 (light bars) and 2007 (dark bars). Numbers are not adjusted for detection probability based on secondary crew observations, nor is any information available on the number of breeding pairs represented by these counts.

G.9.11 Kirsch 1996

Observed Trend:

Table 22. Estimated population trend (*r*) for model least tern populations with 0.50 fledglings per pair, 6 levels of survival from fledging to 2 years of age, and 6 levels of annual adult survival.

Adult survival	Survival from fledging to 2 yrs					
	0.30	0.40	0.50	0.65	0.70	0.80
0.70	−0.260	−0.225	−0.191	−0.166	−0.143	−0.120
0.75	−0.209	−0.175	−0.142	−0.119	−0.096	−0.074
0.80	−0.161	−0.128	−0.096	−0.074	−0.052	−0.031
0.85	−0.116	−0.083	−0.053	−0.031	−0.010	0.010
0.90	−0.072	−0.041	−0.011	0.010	0.030	0.050
0.95	−0.003	0.000	0.029	0.049	0.069	0.088

G.10 Detection Probabilities

G.10.1 Shaffer et al. 2013

Findings:

Table 5. Daily detection probabilities for least tern and piping plover nests on three Missouri River study areas, 2006–09, along with probability that a nest would be detected in three and seven visits.

[LETE, least tern; PIPL, piping plover; GRR, Garrison River Reach; GVP, Gavins Point River Reach; SAK, Lake Sakakawea]

Species	Study area	Year	Number of opportunities	Number of detections	Daily detection probability	95-percent confidence interval	Three visits	Seven visits
LETE	GRR	2006	241	117	0.49	0.42–0.55	0.86	0.99
LETE	GRR	2007	227	107	0.47	0.41–0.54	0.85	0.99
LETE	GVP	2008	436	194	0.44	0.40–0.49	0.83	0.98
LETE	GVP	2009	290	136	0.47	0.41–0.53	0.85	0.99
PIPL	GRR	2006	366	237	0.65	0.60–0.70	0.96	1.00
PIPL	GRR	2007	411	205	0.50	0.45–0.55	0.87	0.99
PIPL	GVP	2008	572	237	0.41	0.37–0.45	0.80	0.98
PIPL	GVP	2009	314	194	0.62	0.56–0.67	0.94	1.00
PIPL	SAK	2007	202	107	0.53	0.46–0.60	0.90	0.99
PIPL	SAK	2008	167	86	0.51	0.44–0.59	0.89	0.99

Table 26. Average number of attempts to relocate a fledging-age chick (14–18 days for terns, 18–25 days for plovers), and the estimated detection probability from simulation by species and study area.

[Detection probability values and confidence intervals interpolated from table 25. LETE, least tern; PIPL, piping plover; GRR, Garrison River Reach; GVP, Gavins Point River Reach; SAK, Lake Sakakawea; NA, not applicable]

Species	Study area	Year	Attempts	Detection probability	95-percent confidence interval
LETE	GRR	2006	2.33	0.54	0.47–0.61
LETE	GRR	2007	1.97	0.50	0.43–0.57
LETE	GVP	2008	2.46	0.49	0.46–0.53
LETE	GVP	2009	1.98	0.42	0.39–0.46
PIPL	GRR	2006	NA	NA	NA
PIPL	GRR	2007	2.54	0.86	0.84–0.89
PIPL	GVP	2008	2.45	0.70	0.67–0.73
PIPL	GVP	2009	4.54	0.80	0.78–0.82
PIPL	SAK	2007	3.38	0.99	0.99–0.99
PIPL	SAK	2008	3.41	0.99	0.99–0.99

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