# 1993 Waterfowl Nesting Success on Fish Springs National Wildlife Refuge

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## Introduction

The finite rate of population growth ($\lambda$) is the product of life-cycle processes such as age- or stage-specific survival rates. These processes, also known as vital rates, vary in time and space and have varying levels of influence on population growth. For example, in ground-nesting birds, nesting success has been found to be an important determinant of recruitment and population growth rate (Hoekman et al., 2002). In a sensitivity analysis of the midcontinent mallard (*Anas platyrhynchos*) population of North America, nesting success demonstrated the greatest influence on $\lambda$ relative to other vital rates (Hoekman et al., 2002). Population growth in greater prairie chickens is similarly sensitive to changes in nesting success, with the combined product of nesting success and brood survival the primary driver of variation in $\lambda$ (Wisdom and Mills 1997). Such disproportionate effect of a vital rate to $\lambda$ can highlight where conservation actions can have the greatest influence on population management.

Understanding of factors that influence vital rates known to be important drivers of population growth is necessary for effectively managing populations. In ground-nesting birds factors such as weather (Harvey, 1971), food availability (Gloutney and Clark, 1991), and nest parasitism (Lokemoen, 1991) have been demonstrated to influence nesting success, largely through abandonment. However, the greatest cause of nest failure in ground-nesting birds is nest depredation (Klett et al., 1988). Nest failure can be caused by depredation in a variety of ways, depending on the behavior and foraging habits of the nest predators in the area. Some nest predators focus their efforts on consuming eggs, while others prey more frequently on hatchlings and adult

waterfowl (Sargeant et al., 1993).   Not only do predators cause nest failure directly by killing nesting individuals or depredating nests, but their presence can indirectly reduce nesting success. In a study of several passerine species in Arizona experimental predator control resulted in increased parental investment through higher rates of nestling feeding, increased egg size, greater clutch mass, and higher rates of males feeding incubating females (Fontaine and Martin, 2006). Increased predator pressure may reduce these actions as nesting individuals seek to conserve energy in the event that renesting is required, and as individuals attempt to reduce activity that could draw predator attention to the nest (Fontaine and Martin, 2006).

Many studies have explored the role of nest predation on nesting success. Differential effects of individual predator species on nesting success have received considerable attention in the literature. Examples of these studies include those that have explored how nesting success increases in areas inhabited by red fox (*Vulpes vulpes*) when coyotes (*Canis latrans*) are also present (Sovada et al., 1995), and evidence of incidental, rather than purposeful, nest predation by striped skunks (*Mephitis mephitis*) (Vickery et al., 1992).  The influence of nest-site characteristics on nesting success has similarly been well documented (patch size [Horn et al., 2005], landscape structure [Bergin et al., 2000], habitat type [Higgins, 1977], nest-site vegetation structure [Crabtree et al., 1989]). However, the majority of nesting success studies have been conducted in the Prairie Pothole Region (PPR) of the United States and Canada, North America's most significant waterfowl breeding ground (Smith et al., 1964). Exploring nest predation processes in dissimilar or unique nesting habitats can help

determine how general the processes are. For example, Fish Springs National Wildlife

Refuge in the West Desert of Utah includes nearly 10,000 acres of wetland habitat

created by artesian springs. In much of the PPR, agricultural development has reduced

waterfowl breeding habitat to fragmented islands of wetlands and grassland within a

matrix of cropland. Similarly, Fish Springs is a habitat island in the midst of a salt-desert

ecosystem, characterized by salt-tolerant vegetation (e.g., *Sarcobatus vermiculatus,*

*Spartina gracilis, Atriplex confertifolia*). The PPR's matrix of agricultural land use favors a

broad variety of generalist mesopredators, likely at elevated densities due to the greater

and more persistent food resources associated with anthropogenic development

(Fedriani et al., 2001; McKinney, 2002). Fish Springs' location in the West Desert results

in the surrounding area being relatively inhospitable, limiting the spectrum of

mesopredator species present. The suite of mesopredators found at Fish Springs

primarily includes bobcat (*Lynx rufus*), coyote, raven (*Corvus corax*), gopher snake

(*Pituophis*

*melanoleucusl*), Great Basin rattlesnake (*Crotalus oreganus lutosus*), and rarely kit fox

(*Vulpes macrotis*) and badger (*Taxidea taxus*). This creates a unique situation for

waterfowl management and investigations of nesting success.

The mission of the National Wildlife Refuge System is to establish a network of

lands and waters for the management of wildlife resources for future generations

(USDOI, 2000; USFWS, 2009). Each refuge within the National Wildlife Refuge System is

required to prepare and implement a habitat management plan with the overarching

purpose of conserving the wildlife, plants and habitats found on the refuge (USDOI,

2000; USFWS, 2009). An important component of this planning process is gathering baseline information on trust species within the area. An investigation of nesting success and the factors affecting it on Fish Springs NWR will provide important information to inform future management and create a comprehensive refuge management plan, addressing the motivations behind the formation of the Refuge: the protection and support of migratory birds (USFWS, 2004).

## Study Area

Fish Springs National Wildlife Refuge is located in Juab County, Utah, at the southern end of the Great Salt Lake Desert. The Refuge comprises 17,992 acres, and includes 10,000 acres of spring-fed wetlands. Five main artesian springs at the base of the Fish Springs Range provide water to the Refuge, along with several lesser upwellings. Although the springs vary in temperature and salinity, they are all brackish (2010 mean minimum and maximum specific conductivity 3983 and 6203 µS/cm, respectively) with a minimum annual mean pH of 7.4 and a maximum annual mean pH of 7.9. Mean water temperature for all springs is 23.3° C (SD = 2.2° C), with individual mean spring temperatures ranging from 20.2–26.9° C. Conductivity generally increases in the springs and pools from south to north and from west to east (annual mean of 4002 µS/cm in Percy spring, the most southerly spring; annual mean of 6148 µS/cm in North spring, the most northerly spring) due to an increasing salinity gradient. Dry areas of the Refuge are dominated by a salt-desert shrub community with black greasewood (*Sarcobatus vermiculatus)* and fourwing saltbush (*Atriplex canescens)* as the dominant over story, and inland saltgrass (*Distichlis spicata*) and alkali sacaton (*Sporobolus*

*airoides)* as the dominant understory vegetation. Wet areas of the Refuge support

species of rushes (*Juncus* spp.), bulrushes (*Schoenoplectus* spp.), spikerushes (*Eleocharis*

spp.) and cattail (*Typha* spp.). Common vegetation in these areas include saltgrass, Baltic

rush (*Juncus balticus*), hardstem, alkali and Olney's three-square bulrush

(*Schoenoplectus acutus, S. maritimus* and *S. americanus*, respectively), southern and

broadleaf cattail (*Typha domingensis* and *T. latifolia*, respectively) and common reed

(*Phragmites australis)* (Bolen 1964; USFWS, 2004). Trees are located only at the Refuge

office complex, with a small patch of mature trees also established at a picnic area

located at one of the springs. On average, the hottest month is July (mean minimum

temperature 18.1° C ; mean maximum 35.2° C), with January being the coldest month

on the Refuge (mean minimum temperature -7.8° C;  mean maximum 4.2° C). The

coldest recorded temperature at the Refuge is -28.3° C, but the springs do not freeze

over completely (USDOI, 2004).  Annual mean precipitation is 20.3 cm, with the majority

falling in the spring and fall. May is the wettest month, with an average of 2.7 cm

precipitation, contributing 13% of the total average annual precipitation (Western

Regional Climate Center, http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?ut2852).  Frost-

free days generally extend from late April until mid-October.

Shortly after establishment of the Refuge an impoundment system was

constructed, creating nine wetland management units (USFWS, 2004). The

impoundment system resulted in creation of semi-permanent wetland habitat

(hereafter pools) within each unit. Evapotranspiration results in a significant loss of

water from spring through fall in the pools.   Evapotranspiration rates increase through

the summer, causing a transition in early to mid-June from water accumulation (from the springs and slight precipitation) to water loss in the pools. Since 1988, the impoundment units have been managed through a prescribed schedule of rotational drawdowns. Under this system, certain units are completely drained for an entire year before allowing the unit to resume optimal pool levels. Optimum pool levels are maintained throughout the year in the Avocet, Mallard, Shoveler and South Curlew impoundment units, except when one of these units is selected for drawdown. Pools located north of these units are allowed to dry during the deficit period of the year, with the water levels decreasing during the growing and nesting season.

## Methods

Limited information was found in historical documents on methods employed in 1993 for collection of field data. However, in documents on similar searches done in 1992, it was recorded that study sites for nest searching were selected based on known duck productivity the year before. Varied habitats and the high proportion of water (50% of the 10,000 acre area within the nine impoundments is shallow-water marsh or open water) on the Refuge resulted in the selection of several small sample areas for nest searches (USFWS, 2004).

To obtain samples for estimating nesting success among units and species, grassland and dry wetland areas were systematically searched by 2 person crews using cable-chain drags towed by all-terrain vehicles (Klett et al., 1986). For sloughs, islands, emergent habitats and other areas not accessible with ATVs, researchers used an airboat

or waded through the areas. In these situations, researchers either dragged a rope between them or beat the vegetation as they walked. Although information on the methods used in 1993 is lacking, the 1992 annual narrative of the Fish Springs Refuge notes that each area was only surveyed once in a similar study in 1992.

All nests found were marked with a flagged stick placed 0.9 to 3.1 meters in front of the nest ("front" being considered the side of the nest the searchers first encountered when moving in the direction of nest searches). For each nest the associated vegetation was noted, distance to water was estimated, and nest initiation date was recorded. Nest initiation date was estimated by field-candling eggs (Weller, 1956). Nests were revisited every 7 to 10 days (with some exceptions) until fate was determined (at least one egg hatched or the nest was destroyed or abandoned). On each visit nest status (active, successful, abandoned, or destroyed) was recorded; for depredated nests predator species (i.e., coyote, raven, snake) was recorded when evidence at the nest site was suitable to make the determination. The location, species and fate of 1993 nests were recorded on hand-drawn maps of the Refuge management units. In 2012, nest locations were digitized in a Geographic Information System (GIS) database (ArcGIS 10.1, ESRI, Inc., Redlands, CA) to estimate distance to nearest road for each nest using the Near Distance tool. Due to the crudeness of the hand-drawn 1993 maps, the calculated distance to road only provided a relative distance from each nest to the nearest road.

*Data Analysis*

We assessed an *a priori* suite of models to determine the amount of support for each of our hypotheses regarding factors influencing nesting success (Burnham and Anderson, 2002). The strength of support for each model was assessed by ranking models with Akaike's Information Criterion corrected for small sample sizes (AICc) and by calculating the normalized relative model likelihoods ($\omega_i$) for each model (Burnham and Anderson, 2002). The models for distance to road and water were also both log-transformed to test for a threshold effect of distance to each of these parameters on nesting success.

Nest daily survival rate (DSR) was modeled following Rotella et al. (2004). Briefly, daily survival rate was estimated using a maximum-likelihood approach that accounted for bias associated with finding nests at various stages (Mayfield, 1975; Johnson, 1979). The fate of nests between investigator visits, i.e., nest interval fate, was modeled with binomially-distributed errors using R 2.14.0 (R Development Core Team, 2012) with package 'RMark' (Laake, 2012) and program MARK (White and Burnham, 1999). Nests abandoned, damaged or destroyed due to investigator activity were not included in analysis. Once estimates for DSR were obtained, nesting success was calculated as $(DSR)^n$, where $n$ is the average number of days from initiation to hatch for ducks ($n = 35$). The R script used to conduct the analysis of nesting success is provided in the Appendix.

## Results

Nest searches were conducted from the second week of May until late June. A total of 97 nests were found and monitored; five nests were not mapped (precluding estimation of distance to the nearest road), resulting in 92 nests used in analysis. Nest

initiation dates ranged from 28 April–6 July, with mean and median initiation dates of 1 and 2 May, respectively. Of the nests monitored, 37 were gadwall (*Anas strepera*), 26 were northern pintail (*Anas acuta*), 9 were mallard (*Anas platyrhynchos*), 8 were unidentified teal species, 5 were northern shoveler (*Anas clypeata*), 4 were redhead (*Aythya americana*), 2 were canvasback (*Aythya valisineria*), and 1 was a blue-winged teal (*Anas discors*).

Mean distance from nests to the nearest water was 67 m (SD = 102.9 m; median 20 m), ranging from 0 to 500 m. The mean distance from nests to the nearest road was 122 m (SD = 154.9 m; median 20 m), ranging from 1 to 808 m.

Considerable model uncertainty existed in our *a priori* suite of models with all models being within 2 $AIC_c$ units of the top model (Table 1). A general rule of thumb for model selection using $AIC_c$ is models within 2 units of the top model have substantial support given the data (Burnham and Anderson 2002). Therefore, our *a priori* suite of models explained the observed patterns in nesting success at Fish Springs in 1993 nearly equally well. We focused our inference on the top four models, all of which were within 1 $AIC_c$ of each other, primarily for brevity. Additional nest data should facilitate greater model selection certainty, assuming a model representative of nesting success processes at Fish Springs is within our current model suite.

Our top model indicated nesting success was consistent among species and across the Refuge during the nesting season. The estimate of nesting success from the top model was 4.9% (95% CI 2.2–9.2%), similar to the Mayfield estimate of 5% provided in the 1993 Fish Springs NWR annual narrative (USFWS, 1993). Models within 1 $AIC_c$ unit of the top model included initiation date, distance to road, and distance to water (Table

1). These latter models indicated negative relationships between nesting success and initiation date, distance to road, and distance to water, respectively (Table 2). Species specific nesting success, and threshold effects of distance to road and water were less well supported (Table 1).

**Table 1.**
Results of models used to explore variation in nesting success during the 1993 breeding season on Fish Springs National Wildlife Refuge, Utah, 1993. Distance to nearest road and water were quantified for each nest; see methods for details. Species-specific nesting success was considered for the most common species found during nest searching: gadwall, northern pintail, mallard, and other.

| Model | $k^a$ | $\Delta AICc^b$ | $\omega_i^c$ |
|---|---|---|---|
| Null | 1 | 0.00 | 0.225 |
| Nest initiation date | 2 | 0.47 | 0.178 |
| Distance to road (m) | 2 | 0.53 | 0.172 |
| Distance to water (m) | 2 | 0.94 | 0.140 |
| Species | 3 | 1.61 | 0.101 |
| lnDistance to road | 2 | 1.74 | 0.094 |
| lnDistance to water | 2 | 1.83 | 0.090 |

[a]Number of model parameters
[b]Difference in Aikaike's Information Criterion (corrected for small sample sizes) between each model and the best model (Null)
[c]Normalized relative model weight (Burnham and Anderson, 2002)

**Table 2.** Estimated effect (and standard error) of predictor variables on duck nesting success at Fish Springs National Wildlife Refuge, Utah, 1993, based on the four most parsimonious nesting success models. Distance to nearest road and water were quantified for each nest; see methods for details.

| Model | Intercept (SE) | $\beta$ (SE) |
|---|---|---|
| Null | 2.4 (0.13) | - |
| Nest initiation date | 2.8 (0.33) | -0.009 (0.008) |

| | | |
|---|---|---|
| **Distance to road (m)** | 2.5 (0.17) | -0.001 (0.001) |
| **Distance to water (m)** | 2.5 (0.15) | -0.001 (0.001) |

## Discussion

Our re-analysis of nesting success data from 1993 provided the opportunity to explore patterns in variation previously not examined. We tested competing hypotheses regarding drivers of nesting success at Fish Springs National Wildlife Refuge. Considerable model selection uncertainty was apparent in our results, but we still believe important insights can be gained from the results. Moreover, our results will help refine future nesting success work on the Refuge. The unique location of the Fish Springs Refuge requires consideration of the distinctive relationships and behavioral patterns that are especially specific to desert ecosystems. Due to the fact that the majority of nesting failures in the 1993 survey were the result of predation, examining potential mechanistic explanations with an emphasis on predation and predator efficiency is appropriate.

Our top model indicated nesting success was consistent among species and across the Refuge during the nesting season. We estimated nesting success as 4.9% (95% CI 2.2–9.2%) from the best (null) model, consistent with the estimate of nesting success provided in the 1993 Refuge Annual Narrative (USFWS, 1993). It is notable that predator control was conducted in 1993, a continuation of the practice beginning in the mid-1960s. In 1993, predator control occurred just prior to the onset of duck nesting through the end of incubation, removing snakes, coyotes and ravens. In spite of this,

nesting success was still low, well below the 15% threshold necessary for population stability (Cowardin et al., 1985).

Some studies have indicated that reducing predator numbers may actually cause predator abundance to increase as the amount of resources available to remaining individuals increases. This can lead to higher survival and reproduction rates for surviving individuals (Knowlton, 1972). Furthermore, the removal of territorial individuals may attract individuals from nearby uncontrolled areas (Knowlton, 1972). However, studies have also indicated that predator control can be effective in improving waterfowl nesting success when done intensively in a small area (e.g. < 300 km$^2$) (Duebbert and Lokemoen, 1980; Pieron and Rohwer, 2010). Details on the intensity of predator control and the methods for removal that were used on Fish Springs NWR have been lost.

We found weak support for an intra-seasonal decline in nesting success. The West Desert of Utah that encompasses Fish Springs is characterized by a bimodal precipitation pattern (i.e., spring and fall), with a transition from water accumulation to water deficit in mid-June even in spring-fed areas of the Refuge. After spring rains cease, seasonal pools across the West Desert decrease in number and size, and much of the vegetation senesces. For vertebrates, especially those too large to find refuge in microhabitats such as burrows, the need for water grows as the summer progresses and temperatures increase into July (Noy-Meir, 1974). With the decline of seasonal resources, Fish Springs becomes an important source of water and green vegetation for wildlife throughout the area. Coyotes may migrate to the Refuge, drawn by the need for

water and density of prey (Moorcroft et al., 2006). Thus, the number of coyotes on the Refuge—and the likelihood that waterfowl nests will be depredated by foraging coyotes—may increase through the nesting season. Emery et al. (2005) found a decline in nesting success during the breeding season within managed nesting cover in the Canadian PPR, with concurrent increases in nesting success in unmanaged cover types. They posited that the different patterns observed during their study could have resulted from differences in prey availability between the cover types and the resultant attraction to predators; small mammal abundance is generally greater in managed cover types later in the summer than in unmanaged types (Pasitschniak-Arts and Messier 1998). Fish Springs NWR may similarly attract increasing numbers of nest predators through the breeding season as water and forage resources become less available in the surrounding landscape.

Furthermore, the negative relationship observed between nesting success and initiation date may be a reflection of changes in coyote food requirements. Reproductive female coyotes have an increased caloric requirement throughout pup rearing (April- August) (Laundré and Hernández, 2003), which coincides with waterfowl nesting on the Refuge and may cause female coyotes to increase the time they spend hunting. Evidence has also shown that coyotes focus hunting effort on the most efficient prey items available (largest energetic return with least effort) (Laundré and Hernández, 2003; Till, 1992) while provisioning young, further supporting the hypothesis that coyotes would travel to areas where larger prey—such as lagomorphs or waterfowl— will be concentrated. The increase in nest predation toward the end of the season also

coincides with increases in juvenile coyote mobility and foraging, as offspring begin to actively travel with their parents to learn foraging skills (Laundré and Hernández, 2003).

Nesting success was negatively related to distance to road, although support for these results were·also weak.  These findings are counter to much of the current available research on waterfowl nesting success, which has indicated nesting success generally increases with distance to habitat edge (Horn et al., 2005). The results observed in this study may be the product of the unique habitat created on the Refuge. Other studies exploring waterfowl nesting success near roads have been done where roadsides are strip habitats, often hemmed in by agricultural fields or development (Bergin et al., 2000). It is possible that the gravel roads of the Refuge create updrafts and wind turbulence that reduces olfactory detection, providing greater protection to nests in close proximity (Dritz, 2010). Also, although traffic on the Refuge is very light, coyotes have been shown to avoid open areas near roads during daylight hours (Roy and Dorrance, 1985), which could reduce foraging near roads. This avoidance may have been enhanced by predator control that was being carried out during and prior to 1993. However, these results should be considered cautiously due to the imprecision of hand-drawn maps and the resultant relativity of nest placement in the digitizing process.

The final model within the top model set considered indicated nesting success declined as distance to water increased.  A study in North Dakota had similar results, finding that mallards nesting in marshes had higher success than those nesting in upland habitats (Krapu et al., 1979). Higher nesting success with increasing proximity to water could be the effect of higher and denser vegetative cover near water. Nesting success is

positively influenced by vegetation density at the nest site (Schranck, 1972). Dense cover may provide several benefits to nesting waterfowl. Cover may create a microclimate that alters ambient temperature, a factor which influences attendance on the nest (Caldwell and Cornwell, 1975; Stokes and Boersma, 1998). With higher temperatures, more frequent female movements to and from the nest for incubation rests may attract predator attention and increase the likelihood that predators will locate nests. Due to the dryness of the environment at Fish Springs NWR, females lacking the protection of this vegetation-created microhabitat may also show an increased need to leave the nest to re-hydrate.

Higher and denser cover near water can also provide visual and olfactory concealment, reducing the risk of depredation from both avian and mammalian predators (Borgo, 2008; Dritz, 2010). In addition to reducing detection, studies have also noted that denser cover can slow or decrease predator searches and efficiency, decreasing the likelihood of discovery during the foraging activities of mammalian predators (Bouffard et al, 1987; Schranck, 1972). This would seem especially true of Fish Springs NWR, where predators that are known to prefer hunting in wetland areas— such as raccoons (*Procyon lotor*)—are not known to occur.

Although considerable uncertainty was evident in model selection during this study, results suggests that nest initiation date, distance to road and distance to water are worthwhile hypotheses to consider in future studies of waterfowl nesting success on Fish Springs National Wildlife Refuge. Additional data could provide a clearer understanding of their importance as drivers of nesting success on this refuge. Ultimately, this

information could assist refuge managers in the future as they consider habitat improvements and predator control in their attempts to improve waterfowl production on the Refuge.

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## Appendix
*R code used to analyze the 1993 Fish Springs National Wildlife Refuge duck nesting data.*

```
#### RMark analysis of Fish Springs nesting success 1993 & 2012 ####
## JM Warren; 5/9/2012
## Will need to update for 2012 at the end of the field season.

##Open RODBC package to link to the database
library(RODBC)

##Link to the .mdb file; this will need to be updated for each person running the models
channel<-odbcConnectAccess("C:\\Documents and Settings\\jewarren\\Zone
Biologist\\Refuges and Offices\\Fish Springs\\Nesting study\\Fish Springs Nest
Data.mdb")

##Extract the Access tables and put into a data frame.

fsnest<-sqlFetch(channel,"NestIntervalDataRMark")
names(fsnest)
close(channel)

##The earliest nest found was on May 11th 1993 (131).
##The last day a nest was viable was August 10th, that is the last day July 30th 1993
(207).
##This results in 76 'occasions' for the nest survival models.
```

##Both of these dates need to be updated for 2012 if earlier or later dates occur.

##Drop nests that do not have a distance to road value
fsnest<-na.omit(fsnest)

##Treat Year as a factor
fsnest$NestYear<-as.factor(fsnest$NestYear)

##Group nests based on species (gadwall, pintail, and other)
fsnest$spp<-"gadw"
fsnest$spp[fsnest$NestSpecies=="NOPI"]<-"nopi"
fsnest$spp[fsnest$NestSpecies %in%
c("MALL","Teal","TEAL","CANV","BWTE","NOSH","REDH")]<-"other"
fsnest$spp<-as.factor(fsnest$spp)

##Log-transform distance to road and distance to water to test psuedo-threshold forms of each
fsnest$lDistH2O<-log(fsnest$DistH2O+0.5)
fsnest$lDistRd<-log(fsnest$DistRd)

##Examine a summary of the dataset and its structure
summary(fsnest)
str(fsnest)

##Process the data
fsnest.process<-process.data(fsnest, model="Nest", begin.time=1, nocc=81,
groups=c("NestYear","spp"))
fsnest.ddl<-make.design.data(fsnest.process)

#Figure of nest initiation date summary by year
summary(fsnest$Init[fsnest$NestYear==1993])
summary(fsnest$Init[fsnest$NestYear==2012])
boxplot(fsnest$Init~fsnest$NestYear, xlab="Year", ylab="Julian Date", cex.axis=1.5,
cex.lab=1.5, ylim=c(140,200))

#Figure of clutch size summary by year
summary(fsnest$CS[fsnest$NestYear==1993])
summary(fsnest$CS[fsnest$NestYear==2012])
boxplot(fsnest$CS~fsnest$NestYear, xlab="Year", ylab="Clutch Size", cex.axis=1.5,
cex.lab=1.5, ylim=c(0,12))

##A function for evaluating a set of competing models
run.fsnest <- function()
{
#1. Constant daily survival rate (DSR)
null<-mark(fsnest.process, fsnest.ddl)

```
#2. DSR varies by year
#year<-mark(fsnest.process, fsnest.ddl,
model.parameters=list(S=list(formula=~NestYear)))

#3. DSR varies by time
time.trend<-mark(fsnest.process,
fsnest.ddl,model.parameters=list(S=list(formula=~Time)))

#4. DSR varies by time each year (a year and time interaction)
#time.year<-mark(fsnest.process, fsnest.ddl,
model.parameters=list(S=list(formula=~Time*NestYear)))

#5. DSR varies by species (gadw, nopi, other)
spp<-mark(fsnest.process, fsnest.ddl, model.parameters=list(S=list(formula=~spp)))

#6. DSR varies with distance to water
dist.water<-mark(fsnest.process, fsnest.ddl,
model.parameters=list(S=list(formula=~DistH2O)))

#7. DSR varies by distance to road, psuedo-threshold functional form
dist.water.pt<-mark(fsnest.process, fsnest.ddl,
model.parameters=list(S=list(formula=~lDistH2O)))

#8. DSR varies with distance to road
dist.road<-mark(fsnest.process, fsnest.ddl,
model.parameters=list(S=list(formula=~DistRd)))

#9. DSR varies by distance to road, psuedo-threshold functional form
dist.road.pt<-mark(fsnest.process, fsnest.ddl,
model.parameters=list(S=list(formula=~lDistRd)))

#Return model table and list
return(collect.models() )
}

fsnest.results<-run.fsnest() #Runs the models above

#View the model results in a table
fsnest.results
```