AN ABSTRACT OF THE THESIS OF

Amber Marie Johnson for the degree of Master of Science in Wildlife Sciences

Title: Food Abundance and Energetic Carrying Capacity for Wintering Waterfowl in
the Great Salt Lake Wetlands

Abstract approved:

________________________________________
Bruce D. Dugger

Abstract

Great Salt Lake (GSL), Utah, is surrounded by 191,884 hectares of wetlands. These
wetlands provide critical habitat for hundreds of thousands of breeding, migrating and
wintering waterfowl each year. As the human population around GSL increases,
government officials have proposed to divert more water from the Jordan, Ogden, and
Bear Rivers to meet municipal demands. Freshwater diversions from the rivers that
supply the wetlands will likely decrease the amount of wetland habitat available to
waterfowl. To justify current and future water allocation for wetlands around the
GSL, managers must quantify the value of these habitats to migratory wetland birds,
and one measure of habitat quality is food abundance. However, food abundance in
the GSL wetlands is unknown. In this study, I quantified food abundance in managed
and unmanaged wetland areas along the east side of the GSL. Managed wetlands
included state, federal, and private lands contained within levees and actively managed
as waterfowl habitat. I defined unmanaged wetlands as those shallow water areas
outside levees impacted by freshwater flows from the Jordan, Bear, and Ogden Rivers and from managed wetlands. Unmanaged wetlands are under no specific management regime and are under the greatest threat by proposed water diversions because they are currently not protected under Utah water laws. I used multi-stage and simple random sampling to quantify food abundance in managed and unmanaged wetland. Sampling in managed wetlands occurred during September 2005-2006 (n = 12) and unmanaged wetlands during 2006 (n = 3) before the majority of waterfowl arrive on fall migration. I estimated the biomass of seeds, tubers, invertebrates, and submerged aquatic vegetation (SAV). Mean tuber and invertebrate biomass was similar between managed and unmanaged wetlands in 2006 (ps > 0.09); however, leafy biomass was 37% higher in unmanaged wetlands (t = 2.47, p = 0.015) and seed biomass was 65% higher in managed wetlands (t = 5.06, p < 0.001). I then used biomass estimates to calculate energetic carrying capacity of both managed and unmanaged wetland habitats and determine the effects of possible water diversions from unmanaged wetlands on migratory waterfowl. I used information on daily bird energy needs, biomass estimates, and true metabolizable energy of each food type to estimate duck carrying capacity in habitats dominated by submerged aquatic vegetation. I separated ducks into three guilds based on food habits and feeding methods (dabblers, divers and grazers). The unit of comparison for energetic carrying capacity was Duck Use Day (DUD), defined as the amount of energy needed to support one duck for one day. Carrying capacity was higher in managed vs. unmanaged habitats for all three duck foraging guilds. If unmanaged wetlands decline by 80%, I estimated that duck carrying capacity would decrease 34% for grazers, 20% for divers, and 14% for
dabblers. Results from this research provide a quantitative value to both managed and unmanaged wetlands surrounding the GSL to wintering waterfowl and provide guidance for the future conservation, restoration and maintenance of hundreds of thousands of hectares of critical wetland habitat.
Food Abundance and Energetic Carrying Capacity for Wintering Waterfowl in the Great Salt Lake Wetlands

by

Amber Marie Johnson

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I understand that my thesis will become part of the permanent collection of Oregon State University. My signature below authorizes release of my thesis to any reader upon request.

______________________________
Amber Marie Johnson, Author
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Chapter 1

GENERAL INTRODUCTION

The Great Basin is a vast area, surrounded by the Rocky Mountains to the east and the Sierra Nevada and Cascade Mountain ranges to the west. It includes areas of Nevada, California, Oregon, and Utah. Due to low precipitation, the few wetlands located within this semi-arid region are ecologically significant (Post 1977; Bellrose 1980; Kadlec and Smith 1989; Baldassare and Bolen 1994; Warnock et al. 1998; Oring et al. 2000; Haig et al. 2002; Intermountain West Joint Venture 2005). Important wetland areas within the Great Basin include the Great Salt Lake, Utah, Ruby Lake National Wildlife Refuge (NWR) and Carson Sink, Nevada, Malheur NWR, Summer Lake and Abert Lake, Oregon (Kadlec and Smith 1989).

The Great Salt Lake (GSL) is the largest wetland complex in the Great Basin. Although it receives only an average of 38 cm of rainfall per year; it is surrounded by 247,000 ha of both managed and unmanaged wetlands (Aldrich and Paul 2002). Maintained by fresh water from the Jordan, Ogden, and Bear Rivers, these wetlands provide critical migratory waterbird habitat (Christiansen and Low 1970; Baldassarre and Bolen 1994; Aldrich and Paul 2002) supporting five million waterfowl and five hundred thousand shorebirds annually (Gorrell et al. 2005). Because GSL wetlands play an essential ecological role in North American and hemispheric bird life, conservation planning and habitat management is of great importance.

As the human population around GSL increases, government officials have proposed to divert more water from the Jordan, Ogden, and Bear Rivers to meet
municipal demands. Freshwater inflows are important as they influence wetland quantity and quality (Kadlec 1982; Aldrich and Paul 2002). Water law in Utah requires proof of beneficial use before granting water rights (Utah Department of Natural Resources 1999; Aldrich and Paul 2002) but Utah does not recognize wildlife as a beneficial use (Utah Department of Natural Resources 1999). Therefore, freshwater supply to GSL wetlands is not a priority (Aldrich and Paul 2002), and conservation groups are concerned that increased diversions will reduce the capacity of freshwater wetlands around the GSL to support wetland-dependent birds.

To justify current and future water allocation for wetlands around the Great Salt Lake, managers must quantify the value of these habitats to migratory wetland birds, and one measure of habitat quality is food abundance (Goss-Custard and Charman 1976; Smith et al. 1989; Sutherland and Allport 1994; Guthery 1999; Goss-Custard et al. 2003; Taylor and Smith 2005). Wetlands provide foods that fuel important life-history events, including molt, pairing, thermoregulation, establishing nutrient reserves for migration, and reproduction (Heitmeyer and Fredrickson 1981, Krapu 1981, Miller and Newton 1999; Taylor and Smith 2005). Due to increasing threats to the GSL wetland habitat, there is a need for research that determines the potential affects of habitat loss to migratory wetland birds.

Conservation planning efforts for non-breeding birds most commonly assume food is limiting population performance (Central Valley Joint Venture 2006; Intermountain West Joint Venture 2005). Thus, a meaningful and standardized measure of habitat quality is energy density, the number of kilocalories per unit of habitat (kcals/ha). This along with estimates of habitat abundance (ha) can provide a
measure of bioenergetic carrying capacity (Goss-Custard and Charman 1976; Goss-Custard et al. 2002). Estimates of carrying capacity can help guide conservation planning and provide one measure of the impact of human actions on habitat quality for migratory birds (Sutherland and Allport 1994; Goss-Custard 2003). In this study, I sampled foods in permanently flooded wetlands to estimate food abundance for waterfowl (Order ANSERIFORMES), estimate energy availability, and compare energy-based carrying capacity estimates between managed and unmanaged areas on the eastern side of the Great Salt Lake. My objectives were to 1) estimate and compare food resources available to migratory waterfowl in both managed and unmanaged wetlands around Great Salt Lake during fall and 2) estimate and compare carrying capacity for wintering waterfowl in both managed and unmanaged areas under current management and policy regulations.

**Study Area**

Located west of the Wasatch Mountains, GSL is the largest lake (4,403 km²) west of the Mississippi River (Collister and Schamel 2002), but is relatively shallow (mean depth = 4 m; maximum depth = <13 m; Arnow and Stephens 1990). Summer and winter temperatures can reach 38°C and -18°C, respectively (Aldrich and Paul 2002). GSL is bordered along its north and east shoreline by greater than 50,000 ha of managed (20,235 ha privately owned and 32,376 ha state or federally owned) and 197,089 ha of unmanaged wetlands (Aldrich and Paul 2002). Most wetlands are classified as lacustrine, littoral by Cowardin et al. (1979). These wetlands share the following characteristics: 1) situated in a topographic depression or dammed river channel; 2) lack trees, shrubs, persistent emergents, emergent mosses, or lichens with
greater than 30% aerial coverage; 3) total area exceeds 20 acres; and 4) are found from shore to a depth of 6.6 feet. All the wetlands I sampled were further classified as submerged aquatic bed, being dominated by rooted vascular plants, mainly sago pondweed (*Potamogeton pectinatus*) and wigeon grass (*Ruppia maritima*). GSL is a closed basin and water loss occurs only via evapotranspiration. The accumulation of salts during evapotranspiration has created alkaline conditions throughout the system (Kadlec and Smith 1983). Water salinities vary geographically depending on distance from freshwater inflows via the Bear, Ogden, and Jordon River systems and range from highly saline (330-gml⁻¹) to moderately saline (120-gml⁻¹; Post 1977, Kadlec 1982).

Acquisition of GSL wetlands for preservation and private ownership has a long history in Utah. Organized hunting groups first acquired land in the 1890’s for recreational waterfowl hunting; this continued for 50 years (Nelson 1966). The state first acquired important GSL wetlands in 1932 and continued making purchases through the 1990s. All state lands are designated as Waterfowl Management Areas (WMA) and managed by Utah’s Division of Wildlife Resources (UDWR). Currently, UDWR owns and manages eight WMAs that total 32,376 ha (Aldrich and Paul 2002; Figure 1.1). Bear River Migratory Bird Refuge is the only federally managed area on the eastern shore of the GSL, established by Congress on April 23, 1928 (Utah Department of Natural Resources 1999). All private duck clubs and state owned areas manage wetland habitats for migratory waterbirds, particularly waterfowl. In managed wetland impoundments, managers emphasize production of submerged
aquatic vegetation, specifically sago pondweed and wigeon grass (Kadlec and Smith 1983).

Unmanaged wetlands of interest in this study were those occurring west of levees that separate managed areas from the lake and east of Antelope Island (Fig. 1.1). The majority of unmanaged wetlands are regulated under “state management authority” which gives UDWR the ability to create and manage for possible future wildlife areas (Utah Code Ann Sec, 23-21-5). The extent and composition of unmanaged wetland habitats varies greatly with annual changes in lake level. From 1850 to present, the lake has fluctuated in depth more than 6 m, with the lowest elevation recorded at 1277m in 1963 and the highest elevation recorded at 1284 m in 1866 and again in 1986 (University of Utah, http://greatsaltlake.utah.edu/descrption/greatsaltlake/)

Approximately 35 species of waterfowl use GSL wetlands during migration and winter, commonly occurring species include tundra swans (Cygnus columbianus), lesser snow goose (Chen caerulescens), green-winged teal (Anas crecca), mallard (A. platyrhynchos), northern pintail (A. acuta), cinnamon teal (A. cyanoptera), northern shoveler (A. clypeata), gadwall (A. strepera), American wigeon (A. americana), canvasback (Aythya valisineria), redhead (A. americana), ring-necked duck (A. collaris), lesser scaup (A. affinis), common goldeneye (Bucephala clangula), bufflehead (B. albeola), common merganser (Mergus merganser), and ruddy duck (Oxyura jamaicensis; Aldrich and Paul 2002). Additionally, over 35 species of shorebird use the GSL and surrounding wetlands (Aldrich and Paul 2002).
Figure 1.1 Image of the Great Salt Lake, Utah showing the location of managed (white circles) and unmanaged (black square) wetlands I sampled during late summer 2005 and 2006.
Chapter 2

FOOD ABUNANCE AND ENERGETIC CARRYING CAPACITY FOR WINTERING WATERFOWL IN THE GREAT SALT LAKE WETLANDS

Abstract

Great Salt Lake (GSL), Utah, is surrounded by 191,884 hectares of wetlands. These wetlands provide critical habitat for hundreds of thousands of breeding, migrating and wintering waterfowl each year. I used multi-stage and simple random sampling to quantify food availability in managed and unmanaged wetland areas along the east side of the GSL. Sampling in managed wetlands occurred during September 2005-2006 ($n = 12$) and unmanaged wetlands during 2006 ($n = 3$) before the majority of waterfowl arrive on fall migration. Sampling focused on estimating biomass of seeds, invertebrates, and submerged aquatic vegetation (SAV). Mean tuber and invertebrate biomass was similar between managed and unmanaged wetlands in 2006 ($ps > 0.09$); however, leafy biomass was 37% higher in unmanaged wetlands ($t = 2.47, p = 0.015$) and seed biomass was 65% higher in managed wetlands ($t = 5.06, p < 0.001$). I used biomass estimates to calculate to determine energetic carrying capacity of both managed and unmanaged wetland habitats and determine the effects of possible water diversions from unmanaged wetlands on migratory waterfowl. Carrying capacity was higher in managed vs. unmanaged habitats for all three duck foraging guilds (divers, dabblers, and grazers). If unmanaged wetlands decline by 80%, I estimated that duck carrying capacity would decrease 34% for grazers, 20% for divers, and 14% for dabblers. Results from this research will provide guidance for the future conservation, restoration and maintenance of hundreds of thousands of hectares of critical wetland habitat.
Introduction

The Great Salt Lake (GSL) and its associated wetlands are the most important in the Great Basin, supporting over five million wintering waterfowl (Order Anseriformes) annually (Smith et al. 1989, Gorrell et al. 2005). Increase in demand for freshwater in Salt Lake City and surrounding suburban areas may threaten the fresh water supply needed to maintain GSL wetlands. There is concern that increased freshwater diversions will have an impact on the abundance and quality of GSL wetlands for migrating waterfowl. To justify current and future water allocation for wetlands around the GSL, biologists and managers must quantify the value of these habitats for migratory waterfowl.

Wetlands around GSL can be divided into managed and unmanaged. Managed wetlands are surrounded by levees and their total acreage is relatively fixed (50,000 ha). The acreage in unmanaged wetlands varies considerably with lake level, but may be as much as 197,000 ha (Aldrich and Paul 2002). The relative quality of managed vs. unmanaged wetlands for migratory waterfowl is unclear. A useful measure of habitat quality is food abundance (Goss-Custard and Charman 1976; Smith et al. 1989; Sutherland and Allport 1994; Guthery 1999; Goss-Custard et al. 2003; Taylor and Smith 2005), because food provides energy for important life-history events including molt, pair formation, thermoregulation, and premigrational fattening (Baldassarre and Bolden 1994). With estimates of food abundance, the nutritional value of each food, and knowledge about daily bird energy demands, planners can estimate energetic carrying capacity (Sutherland and Allport 1994; Goss-Custard et al. 2003). Estimates of carrying capacity are useful for monitoring current or potential changes to critical
habitats and the resulting effects on bird use (Sutherland and Allport 1994; Percival et al. 1998; Sutherland 1998; Stillman et al. 2005).

In this study, I sampled managed and unmanaged wetlands around the Great Salt Lake to estimate the abundance of submerged aquatic vegetation (SAV) tubers, leafy vegetation, seeds and invertebrates to compare abundance between managed and unmanaged habitats. I then used a bioenergetic approach to estimate wetland carrying capacity for non-breeding waterfowl, compare carrying capacity between managed and unmanaged submerged aquatic vegetation, compare carrying capacity estimates between habitat types, and quantify the impact of decreasing wetland acres available to waterfowl during the non-breeding season.

Methods

Measuring food abundance

Survey sampling design

I sampled six managed and three unmanaged wetland areas in and adjacent to the Great Salt Lake, Utah (Fig. 1.1; Table 2.1). Managed wetlands included state, federal, and private lands contained within levees and actively managed as waterfowl habitat. Managed wetlands are under specific management regimes (e.g., manipulation of fresh water inflow) to support and improve the growth of SAVI defined unmanaged wetlands as those shallow water areas outside levees impacted by freshwater flows from the Jordan, Bear, and Ogden Rivers and from managed wetlands (Aldrich and Paul 2002). On each area, I sampled lacustrine littoral wetlands dominated by submerged aquatic vegetation (Cowardin et al. 1979) because this habitat type provides important waterfowl food (Martin and Uhler 1939; Kuehn and Holmes 1963;
Baldassarre and Bolen 1994) and because this habitat was a management priority on public and private lands.

I used multi-stage sampling (MSS) to estimate food abundance in managed wetlands during September 2005 and 2006 (Schaeffer et al. 1986; Thompson 1992; Stafford et al. 2006) and simple random sampling (SRS) to estimate food abundance in unmanaged wetlands during September 2006 (Thompson 1992). Multi-stage sampling provided estimates representative of all managed wetlands and considered logistical and statistical efficiency (Schaeffer et al. 1986; Stafford et al. 2006), this was particularly important as I dealt with multiple landowner access issues in managed habitats. The three stages land owners, impoundments within landowners, and samples within each impoundment (Stafford et al. 2006). My objective was to make inference to all managed wetlands around GSL; thus, I sampled two of eight WMA’s, one federal refuge, and three private duck clubs (Table 2.1). I randomly selected two impoundments at each site and collected 15 samples from each impoundment. Samples were collected along transects oriented parallel or diagonal to the longest side of each impoundment. I randomly selected the first sampling point by traveling a distance determined using a random number table; subsequent sampling points were placed at fixed intervals with the distance between points varying among units to assure each transect stretched the full length of each impoundment (Anderson and Smith 1993; Adair et al. 1994).

In 2006, I collected 15 samples along each of five transects in Farmington Bay \(n = 75\) and Willard Spur \(n = 75\) and 2 transects in Ogden Bay \(n = 30\). I oriented
transects perpendicular to the levee that separated the lake from managed habitats (Stafford et al. 2006).

To improve sampling efficiency in 2006, I used means and standard deviations from 2005 and conducted a power analysis that explored samples sizes necessary to achieve acceptable power to detect differences in biomass of each food type between managed and unmanaged impoundments (see Appendix I). In response to the power analysis we increased our sampling effort in unmanaged areas by 4 times (original sample size was \( n = 45 \), revised sample size \( n = 180 \)). In addition, I conducted a variance components analysis to determine if variation was greater among landowners or between managed impoundments within a landowner. Variance components analysis indicated the largest source of variation varied depending on the food type; however, variation among landowners tended to be greater than between managed impoundments within a landowner. As I desired some duplicate sampling per landowner, and I only sampled two impoundments per landowner in 2005, I did not chose to spread my fixed number of samples among more landowners.

*Sampling procedures*

At each sampling location, I collected samples in a way that allowed me to generate food-type specific estimates of biomass (seeds, tubers, invertebrates, SAV leafy vegetation) in different portions of the wetland (benthos, water column). Collection of all samples occurred within a 61 cm diameter aluminum stovepipe-sampling frame pressed firmly into the soil (Taylor and Smith 2005). First, I collected above ground SAV leafy vegetation using a double-sided rake (Adair et al. 1994). SAV leafy vegetation was then refrigerated until final processing occurred in the lab. After
removing all above ground leafy vegetation, I collected a 9.6 cm diameter benthic core sample within the stovepipe-sampling frame (Swanson 1983). Core diameter was fixed; however, depth varied to include all benthic material occurring above the hardpan (Engelhardt 1997). After collecting the benthic core, I used a hand pump (5 cm diameter inlet-outlet hose, 60 L per minute pumping capacity) to remove all water contained by a PVC-pipe sampler (22 cm) placed inside the larger 61 cm diameter sampler. Water was filtered through a sweep net (1 mm) and all seeds or invertebrates retained were stored in jars (8 oz. Low Form, plastic screw-top) and preserved in 10% formalin solution for future processing.

Sample processing
I initially processed soil cores each sampling day by rinsing material through 2 mm and 500 um sieves. I froze remaining soil, tubers, and seeds in resealable bags until we returned to Oregon State University where each sample was washed through a series of screens (sizes included 2 mm, 355 um, and 500 um) to separate SAV roots and tubers from seeds and invertebrates. Each food type was dried at a constant temperature (60°C for >24 hrs for leafy vegetation and invertebrates; 60°C for > 48 hrs for seeds and tubers) and weighed (nearest 0.01 g for leafy vegetation, tubers, and invertebrates; 0.0001g for seeds).

SAV leafy vegetation samples were processed by placing samples in a shallow tub and gently washing the floating vegetation to remove any seeds or invertebrates, then the remaining water was passed through 2 mm and 355 um sieves to capture any remaining seeds and invertebrates. All seeds and invertebrates retained in the sieves
were sorted by hand, dried and weighed as above. Processing was the same for samples collected during both years.

In 2006, I randomly selected one invertebrate sample per impoundment in managed areas (n = 12) and four samples in each of the three sample regions (n = 12) in unmanaged habitats to characterize invertebrate community composition. All invertebrates were identified to family using Merritt and Cummins (1996).

Statistical analysis

To generate a single biomass estimate for each food type at each sampling location, I first standardized the food-specific biomass estimates (benthic core, SAV sample, water column sample) to kg/ha. I then summed estimates for each food type across sampling gear and calculated mean and standard error for SAV leafy vegetation, SAV tubers, SAV seeds, and invertebrates using equations appropriate for multi-stage sampling and simple random sampling (SAS PROCSURVEY MEANS and PROCMEANS; SAS Institute 1999). To estimate variances for the MSS, PROC SURVEYMEANS uses Taylor series linearization (Seber 1982; SAS Institute 1999; Stafford et al. 2006). Each MSS sample was weighted using selection probabilities corresponding to the three stages of sampling (Schaeffer et al. 1986; Stafford et al. 2006). I estimated mean biomass for each food type for each sampling procedure (benthic core, SAV sample, water column sample) and then summed values across all sampling procedures for an overall mean. All means are reported ± SE.

Prior to conducting analyses, I evaluated data for normality by plotting residuals (Ramsey and Schafer 2002). I detected no patterns in residual plots, so I used t-tests to compare food abundance between years 2005 and 2006 in managed
wetlands and food abundance by management type during 2006 (managed verses unmanaged) (PROC TTEST, SAS Institute 1999).

I used digitized vegetation maps for three of my sampling sites (Ambassador Duck Club, Chesapeake Duck Club, and Farmington Bay WMA) to estimate the coverage of habitats important to waterfowl and compare the contribution of open water habitats to total area in managed impoundments (Fig. 2.1). Maps were created by Ducks Unlimited from 2006 photo imagery. Digitized maps included multiple habitat types based on predominant vegetation and percent cover of that vegetation (Fig. 2.1). Although ten habitat types were mapped, I used the four habitat types dominated by vegetation known as important to foraging waterfowl; Open water, Alkali Bulrush 75%+, Alkali Bulrush 51-75%, and mud/playa (Martin et al. 1951).

To determine the relative contribution of open water habitats to total energy production in managed impoundments, I estimated energy (kcal/ha) for each habitat type by multiplying food biomass by an estimate of its true metabolizable energy (TME; Sibbald 1976) and the total hectares of each habitat type. I used biomass estimates for food resources in open water habitat from my sampling (Table 2.3). Biomass estimates for all other habitat types came from Dugger et al. 2007. For open water habitats, I included the foods sago pondweed leafy vegetation, seeds, tubers, and invertebrates in my estimate of total energy; for all other habitats available foods included only seeds and invertebrates. I used TME values from the literature including 2.00 kcal/g for leafy vegetation (Petrie et al. 1998), 1.42 kcal/g for sago pondweed seeds (Anderson and Low 1976), 4.02 kcal/g for sago pondweed tubers (Nolet et al. 2006), 2.5 kcal/g for invertebrates (Puol 1975), and 0.65 kca
alkali bulrush seeds (Dugger et al. 2007). There is no estimate of TME for salt grass seeds; therefore, I used the average TME values of multiple grass species (2.5 kcal/g; Kaminski et al. 2003).

I calculated relative abundance of individual invertebrate families for both managed and unmanaged sites. To determine relative abundance for invertebrates by family, I divided the number of individuals from a given family by the total number of individuals. Randomly selected samples were separated and recorded by wetland type (managed or unmanaged).

Bioenergetic carrying capacity

I used information on daily bird energy needs and the amount (biomass) and nutritional quality of each food type to estimate duck carrying capacity in submerged aquatic habitats (i.e., the habitat dominated by submergent aquatic vegetation). I focused on ducks as geese rely heavily on surrounding agricultural fields for food and swans were the focus of earlier work (Engelhardt 1997). I separated ducks into three guilds based on food habits and feeding methods (dabblers, divers and grazers).

Dabbling ducks included green-winged teal (Anas crecca), mallard (Anas platyrhynchos), northern pintail (Anas acuta), northern shoveler (Anas clypeata), American wigeon (Anas americana), and cinnamon teal (Anas cyanoptera). Diving ducks include Redheads (Aythya americana), canvasbacks (A, valisineria), and ring-necked ducks (A. collaris). Grazers included coots (Fulica americana) and gadwall (Anas strepera). I assumed that ducks met 100% of their energy needs from wetland foods, were ideal free foragers with equal ability to obtain food resources, and suffered no travel costs for movements among foraging patches.
The unit of comparison was the Duck Use Day (DUD), defined as the amount of energy needed to support one duck for one day and calculated as:

$$DUD = \frac{\text{Biomass} \times \text{TME} \times \text{Acreage}}{\text{DER by foraging guild}}$$

where biomass equaled food density (kg/ha), TME equaled true metabolizable energy of each food (kcal/kg), acreage was the amount of each habitat type (ha), and DER was the daily energy requirement for a representative bird in each guild. I used food biomass estimates from my sampling of managed and unmanaged habitats. TME estimates for each food was obtained from published sources (sago pondweed seed 1.45 kcal/g, Anderson and Low 1976; sago pondweed leafy vegetation 2.00 kcal/g Petrie et al. 1998; sago pondweed tubers 4.03 kcal/g, Nolet et al. 2006; invertebrates 2.50, Purol 1975). Managed wetland acreage came from Aldrich and Paul (2002) while the source for the amount of unmanaged acreage was Martinson (2007). I only included acreage values for unmanaged habitats in Bear River Bay and Willard Spur. Sampling occurred in other unmanaged wetlands; however Bear River Bay and Willard Spur are historically the only portions of the GSL that have enough freshwater input to support submergent hydrophytes (Aldrich and Paul 2002).

**Daily energy requirements**

I estimated daily energy requirement as 3 x basal metabolic rate (BMR) where BMR was calculated as:

$$\text{BMR (kJ/day)} = 433 \times (\text{body mass in kg})^{0.74}$$

I calculated a representative body mass for each foraging guild for each month as a weighted mean based on the relative abundance of each species in that interval (Table
2.2). Species composition data came from monthly surveys conducted on lands managed by the state of Utah during 2004-05 and 2005-06.

I used information on food habits from the literature to determine which food or combination of foods a guild could use to satisfy its daily energy need. Dabblers met their energy needs by consuming aquatic invertebrates and seeds (Austin and Miller 1995; Johnson 1995; Drilling et al. 2002). Gadwalls were included in both the dabbling and grazing foraging guilds; I assumed they relied exclusively on leafy plant material (i.e. grazers) in September and October. Leafy plant material was then assumed to be senescent after November 1 and no longer available to gadwall. After this date, gadwalls were allowed to exploit the same foods as other dabbler species. American coots, the other grazer, were constrained to meeting their energy needs by consuming stems and leafy parts of sage pondweed and wigeon grass (LeSchack et al. 1997; Mowbray 1999). Since I assumed leafy plant material senesced after 1 November, coot food supplies were essentially eliminated after that date. Divers were allowed to forage on roots and tubers to meet their energy needs (Hohman and Ederhardt 1998; Woodin and Michot 2002).

*Carrying capacity*

I calculated carrying capacity for each foraging guild for both managed and unmanaged wetlands, first assuming that both habitat types were flooded in their entirety (50,000 ha for managed habitat vs. 23,058 ha in unmanaged habitats). This resulted in a single estimate of total DUD for each management type. I expressed DUD as weekly waterfowl abundance by distributing total DUD across calendar dates using information on duck migration chronology from bimonthly aerial survey data.
from the Klamath Basin, CA and OR (unpubl. data). Lastly, I calculated how loss of
unmanaged habitats would impact DUDs by decreasing habitat acreage in unmanaged
habitats from 100 to 20% in 20% increments. Currently, unmanaged wetlands are the
only habitat type under threat to water loss (subsequently habitat loss).
Figure 2.1. Digitized map of habitat types found at Ambassador Duck Club, adjacent to the Great Salt Lake, Utah, 2006. Each color represents a different habitat type based on predominant vegetation and percent cover per acre. The four habitat types compared in this study were, open water (dark blue), Alkali Bulrush 75%+ (dark pink), Alkali Bulrush 51-75% (light blue), and mud/playa (light pink).
Table 2.1. Properties sampled to estimate food abundance in permanent managed and unmanaged wetlands in and around the Great Salt Lake during September 2005 and 2006.

<table>
<thead>
<tr>
<th>Site Name</th>
<th>Management Type</th>
</tr>
</thead>
<tbody>
<tr>
<td>1) Bear River Migratory Bird Refuge</td>
<td>Managed</td>
</tr>
<tr>
<td>2) Farmington WMA</td>
<td>Managed</td>
</tr>
<tr>
<td>3) Ogden Bay WMA</td>
<td>Managed</td>
</tr>
<tr>
<td>4) Ambassador Duck Club</td>
<td>Managed</td>
</tr>
<tr>
<td>5) Chesapeake Duck Club</td>
<td>Managed</td>
</tr>
<tr>
<td>6) New State Duck Club</td>
<td>Managed</td>
</tr>
<tr>
<td>7) Farmington Outflow Area</td>
<td>Unmanaged</td>
</tr>
<tr>
<td>8) Ogden Outflow Area</td>
<td>Unmanaged</td>
</tr>
<tr>
<td>9) Willard Spur Outflow Area</td>
<td>Unmanaged</td>
</tr>
</tbody>
</table>
Table 2.2. Daily energy requirements (DER; kcal/day) for a representative bird from each of three duck foraging guilds by month for two fall-spring seasons at the Great Salt Lake, Utah.

<table>
<thead>
<tr>
<th>Month</th>
<th>Dabbling Duck</th>
<th>Diving Duck</th>
<th>Grazer</th>
</tr>
</thead>
<tbody>
<tr>
<td>August</td>
<td>254 236</td>
<td>316 314</td>
<td>208 208</td>
</tr>
<tr>
<td>September</td>
<td>233 218</td>
<td>296 310</td>
<td>250 242</td>
</tr>
<tr>
<td>October</td>
<td>254 273</td>
<td>305 307</td>
<td>238 242</td>
</tr>
<tr>
<td>November</td>
<td>254 282</td>
<td>307 314</td>
<td>208 208</td>
</tr>
<tr>
<td>December</td>
<td>262 259</td>
<td>312 250</td>
<td>208 208</td>
</tr>
<tr>
<td>January</td>
<td>287 278</td>
<td>269 235</td>
<td>208 208</td>
</tr>
<tr>
<td>February</td>
<td>254 215</td>
<td>257 287</td>
<td>208 208</td>
</tr>
<tr>
<td>March</td>
<td>210 238</td>
<td>329 323</td>
<td>208 208</td>
</tr>
<tr>
<td>April</td>
<td>197 215</td>
<td>307 303</td>
<td>208 208</td>
</tr>
</tbody>
</table>
Results

Environmental conditions

Temperatures fluctuated around normal from June through September of both years in the GSL region (Western Regional Climatic Centers; Fig. 2.6). Temperatures during the early spring of 2005 (March-April) were warmer when compared to 2006 and the mean temperature from 1948-2007. Spring 2006 was wetter than normal from March through June, and below average in July and September (Fig. 2.6) Precipitation fluctuated noticeably around normal during 2005 with no detectable seasonal patterns.

Food abundance

I collected samples from 12 managed impoundments during 31 August to 10 September 2005 (n = 540) and 30 August to 9 September 2006 and samples from unmanaged impoundments during 30 August to 9 September 2006 (n = 540).

Coefficients of variation for mean estimates of food density ranged from 10 to 40 percent for estimates by sampling type and 13 to 21% when each food was lumped across sampling gear. Over half (55%) of CVs for means estimated by sampling type were ≤ 20 % and three out of four CV’s for overall all estimated means were ≤20%.

Mean biomass differed between years in managed impoundments for three of four foods. Invertebrate biomass was 10 times higher in 2006 than in 2005 (t =7.52, p < 0.001), whereas SAV leafy vegetation biomass was higher in 2005 by 30% (t = 3.2, p = 0.002) and tubers were four times higher in 2005 (t = 7.61, p < 0.001; Table 2.3). Mean tuber and invertebrate biomass was similar between managed and unmanaged wetlands in 2006 (ps > 0.094); however, leafy biomass was 37% higher in unmanaged
wetlands ($t = 2.47, p = 0.015$) and seed biomass was 65% higher in managed wetlands ($t = 5.06, p < 0.001$).

Although sampling was not designed expressly to compare food production by ownership (public versus private), I conducted an a posteriori analysis to test this relationship. In 2005, biomass was independent of ownership for SAV leafy vegetation, invertebrates, and tubers ($t = 0.77, p > 0.44$). However, mean seed biomass was 32% higher in private than public managed wetlands ($t = 2.67, p = 0.009$). In 2006, biomass was independent of ownership for seeds, invertebrates, and tubers ($t = 1.87, ps > 0.06$), but mean SAV leafy vegetation biomass was 44% higher in publicly managed wetlands 2006 ($t = 3.13, p < 0.002$).

Open water was the dominate habitat type in managed wetlands comprising 77% of total area (Fig. 2.2), and open water provided 98% of total energy available on these same areas (Fig. 2.2). Physidae were the dominant invertebrate group, comprising 40% in managed and 53% in unmanaged wetlands (Fig. 2.3), followed by Planorbidae, Chironomidae and Arthemiidae. Together, the snails (Physidae and Planorbidae comprised 70% of all invertebrates. Managed wetlands had higher family richness ($n = 14$) than unmanaged wetlands ($n = 10$).

**Bioenergetic carrying capacity**

I used 2005 data for invertebrate biomass when estimating carrying capacity because I felt the 2006 estimate was biased high. Specifically, the 2006 estimates included the shells for all gastropods. Snails were the most abundant invertebrate in my samples, and while shells have no nutritional value, they account for a considerable percent of total dry weight.
Total energy varied by food type thus bioenergetic carrying capacity differed by duck guild. In managed wetlands, DUDs were higher in 2005 than 2006 for diving ducks and grazers, but similar for dabblers (Fig. 2.4). Managed wetlands during 2005 supported the highest number of waterfowl in the diver foraging guild followed by grazers then dabblers. In 2006, carrying capacity was higher in managed vs. unmanaged habitats for all three foraging guilds (Fig. 2.4). Managed wetlands supported the highest number of waterfowl in the grazing foraging guild followed by diving than dabbling when compared to unmanaged wetlands.

I estimated managed permanent wetlands around GSL could support 90,000 divers and 32,000 dabblers during the peak of migration in November (Fig. 2.5). I did not create abundance graphs for grazers (Gadwalls and Coots) as they were only a distinct foraging guild during the months of September and October. Unmanaged wetlands could only support 10% of the diving duck population when compared to managed wetlands (Fig 2.5). I did not create a graph for dabbling ducks in unmanaged wetlands because biomass estimates for seeds in unmanaged habitats were below published foraging thresholds (Reinecke et al. 1989). Reducing unmanaged habitat acreage decreased carrying capacity for all three foraging guilds (Fig. 2.6). At an 80% loss of unmanaged wetlands, DUD decreased 34% for grazers, 20% for divers, and 14% for dabblers.
Figure 2.2 Average monthly temperatures (top) and monthly precipitation (bottom) for March-September 2005 and 2006 compared to the long-term mean (1948-2007). Data was collected at four weather stations surrounding the eastern shore of the Great Salt Lake and included Salt Lake City, Farmington, Ogden, and Bear River Bay, UT (Western Regional Climatic Center).
Table 2.3. Mean biomass [kg/ha (SE)] of key wetland food types by sampling procedure during September in managed and unmanaged wetland sampling regions at Great Salt Lake, Utah, 2005-06.

<table>
<thead>
<tr>
<th>Sample Type</th>
<th>2005 Managed</th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>SAV</td>
<td>Seed</td>
<td>Invertebrate</td>
<td>Tuber</td>
<td>SAV</td>
<td>Seed</td>
<td>Invertebrate</td>
<td>Tuber</td>
<td>SAV</td>
<td>Seed</td>
<td>Invertebrate</td>
<td>Tuber</td>
</tr>
<tr>
<td>Water Column</td>
<td>0(0)</td>
<td>0.8(0.3)</td>
<td>4.0(0.7)</td>
<td>3.1(1.0)</td>
<td>0(0)</td>
<td>4.3(0.8)</td>
<td>4.6(0.7)</td>
<td>0.6(0.2)</td>
<td>0(0)</td>
<td>2.2(0.3)</td>
<td>11.8(1.9)</td>
<td>0.7(0.3)</td>
</tr>
<tr>
<td>Benthic</td>
<td>0(0)</td>
<td>66.6(10.9)</td>
<td>2.1(0.8)</td>
<td>495.4(97.2)</td>
<td>0(0)</td>
<td>64.6(10.9)</td>
<td>69.5(16.5)</td>
<td>122.3(27.4)</td>
<td>0(0)</td>
<td>20.5(2.4)</td>
<td>62.1(34.2)</td>
<td>81.7(12.1)</td>
</tr>
<tr>
<td>SAV</td>
<td>500.3(68.0)</td>
<td>3.1(0.6)</td>
<td>0.5(0.1)</td>
<td>0(0)</td>
<td>354.5(72.2)</td>
<td>1.6(0.6)</td>
<td>0.7(0.2)</td>
<td>0.1(0.1)</td>
<td>561.4(72.3)</td>
<td>2.0(0.5)</td>
<td>5.6(1.6)</td>
<td>0(0)</td>
</tr>
<tr>
<td>Total</td>
<td>500.3(68.0)</td>
<td>70.5(11.8)</td>
<td>6.6(1.6)</td>
<td>498.5(98.2)</td>
<td>354.5(72.2)</td>
<td>70.5(12.3)</td>
<td>74.8(17.4)</td>
<td>123.0(27.7)</td>
<td>561.4(72.3)</td>
<td>24.7(3.2)</td>
<td>79.5(37.7)</td>
<td>82.4(12.4)</td>
</tr>
</tbody>
</table>
Figure 2.3. Percent composition (top) of four habitat types on three managed wetlands (Ambassador Duck Club, Chesapeake Duck Club, and Farmington Bay WMA) and the relative contribution of each habitat type to total energy (kcal/ha) available to migrating waterfowl (bottom) around the Great Salt Lake, 2006.
Figure 2.4. Relative abundance of invertebrates keyed out to family in unmanaged and managed GSL wetlands during September 2006.
Figure 2.5. Comparison of bioenergetic carrying capacity (Duck Use Days) in managed wetlands around the Great Salt Lake for three foraging guilds of ducks between years (top) and between managed and unmanaged wetlands in 2006 (bottom).
Figure 2.6. Carrying capacity based on food energy supplies during 2005-06 and the migration chronology data from nearby Klamath Basin for diving and dabbling ducks in managed wetlands (top) and diving ducks in unmanaged wetlands (bottom).
Figure 2.7. Duck use days for three foraging guilds of ducks (dabbling, diving, and grazing) based on food energy supplies in managed and unmanaged wetlands around the Great Salt Lake. Total habitat includes both managed and unmanaged wetland total hectares (50,000 ha for managed habitat vs. 23,058 ha in unmanaged habitats), sequential x-axis points represents how loss of unmanaged habitats would impact DUDs by decreasing habitat acreage in unmanaged habitats from 100 to 20% in 20% increments.
Discussion

Biomass estimates for all food types were likely biased high as food samples were not taken to a constant weight. Constant weight is achieved when weight is within 0.3 mg of an original estimate after one hour of heating and cooling (Moore and Johnson 1967). I weighed samples at both a constant temperature and at constant time intervals but did not determine a constant weight. Because I did not use the constant weight procedure, it is likely that some sample still contained moisture and resulted in higher biomass estimates.

Food abundance estimates for both managed and unmanaged Great Salt Lake wetlands were predominantly higher than previous studies in similar wetland systems (Winslow 2003; Cox and Kadlec 1995; Dugger, unpubl. data). At Great Salt Lake, Cox and Kadlec (1995) estimated food abundance in impounded freshwater wetlands as 8.7 kg/ha for seeds, 65.3 kg/ha for tubers, and 6.7 kg/ha for invertebrates. No studies have estimated SAV vegetation biomass in the Great Salt Lake wetlands. However, estimates from similar habitats for Lower Klamath and Tule Lake National Wildlife Refuges, CA, ranged from $188 \pm 26$ to $226 \pm 53$ kg/ha (Dugger, unpubl. data) and an estimate for coastal wetlands in Louisiana and Texas was $101 \pm 55.7$ kg/ha (Winslow 2003).

The likely explanation for differences in biomass estimates among studies is natural variation. Cox and Kadlec (1995) sampled during 1989-99 just after a major flood inundated marshes with salt water and subsequently destroyed the submerged aquatic plant community (Foote 1991). This could account for lower biomass levels for sago pondweed seeds and tubers. Sago pondweed biomass decreases at water
depths > 46 cm and is completely excluded at depths >100-120 cm (Robel 1962; Wersal et al. 2006); mean depths at my sampling locations were 36 cm in managed and 27 cm in unmanaged. Biomass estimates based on samples not taken from the Great Salt Lake could vary due to differences in physical characteristics of the wetlands, geographic setting, and climate.

Between-year differences in biomass in my study could have resulted from differences in site specific management activities such as timing and duration of flooding. For example, one of the impoundments at Big Bear Migratory Bird Refuge sampled during 2005 had been reflooded that spring after having one year of remaining dry while other impoundments had been flooded for multiple years. Water level manipulation (including both timing of inundation and depth) is a common tool used by waterfowl managers to increase sago production. Multiple studies have found that drying an impoundment for one or more seasons increases growth after reflooding due to reduced competition from slowly recovering macrophytes and the removal of carp who influence water clarity (Harris and Marshall 1963; Lutz 1960; Van der Valk and Davis 1978). Differences may also reflect variation in foraging pressure by waterfowl. If foraging pressure varied between years (e.g., Greer et al. 2007), fewer seeds and tubers would be available to recruit the following spring, and this might influence fall production.

Variation in both the physical and environmental conditions of Great Salt Lake wetlands between years could also account for different biomass estimates. In 60 studies evaluating biomass estimates for sago pondweed, the most common limiting factor to growth was turbidity (Kantrud 1990). Causes of turbidity include
phytoplankton, erosion, resuspended sediments from the bottom (frequently stirred up by bottom feeders like carp), waste discharge, and algal growth (Perry and Vanderklein 1996). Recognizing that turbidity and nutrients can impact sago pond weed production raises concerns about how development of Southern and Eastern shores, primarily industrial, urban, agricultural, and recreational tourism threatens the lake by increasing runoff that contributes to turbidity (Kantrud 1990; Nelson and Booth 2002). Plans for a major highway that would bisect many GSL wetlands to connect Salt Lake and Davis Counties is also a threat, as is the possibility of using the lake as a dumping site for contaminated soil (arsenic and lead) from a super fund site (Great Salt Lake Planning Team 2000).

Based on data from the Western Climatic Center, temperatures and precipitation levels varied between sampling years. Spring weather conditions have the most influence on sago pondweed biomass because spring is the time of year when sago pondweed germinates and begins to grow rapidly (Kantrud 1990). Spring temperatures in 2005 were warmer than 2006. Warmer temperatures during spring can promote greater germination and increase vegetative growth for sago pondweed (Kantrud 1990). Precipitation also varied between sampling years, being higher during spring 2006. As previously mentioned, greater water depths can decrease growth of sago pondweed.

Yearly variation in algal growth likely contributed to observed differences in food biomass estimates between years. Algal mats suppress sago growth by shading and crushing (Ozimek et al. 1986; Kantrud 1990) and may lower sago’s resistance to burrowing invertebrates (Prejs 1986a, 1987). Similarly nektonic phytoplankton can
negatively impact sago production by increasing turbidity (Crum and Bachman 1973; Andersen 1976; Fbin and Barko 1985). Research is needed to determine if algal mats negatively effects sago growth and if so what are the factors that increase algal growth in the Great Salt Lake wetlands, then effective control and reduction of algae and phytoplankton growth can occur in the Great Salt Lake wetlands to improve submerged macrophyte growth.

Food abundance did vary between managed and unmanaged habitats for SAV leafy vegetation and seeds. SAV leafy vegetation was 37% higher in unmanaged wetlands, while seeds were 65% higher in managed habitats. This variation is likely the result of different physical characteristics between management types affecting the development and reproduction of sago pondweed. Haag (1983) found highest seed production at the shallowest sites in an Alberta Lake however numbers were not significantly related to depth. Instead he noted that seed production at such sites was often limited by light penetration into the water column and increased wave action. Managed wetlands in the GSL are surrounded by levees that provide protection from waste discharge and urban runoff that enters the lake and from high winds. The observed higher seed production in managed wetlands and higher leafy vegetation in unmanaged wetlands could have resulted from lower turbidity and greater protection from wave action by constructed levees.

Food biomass estimates provide a measure of food production; however, to understand energetic carrying capacity for ducks estimates of production need to be adjusted to consider food availability. My estimates of food biomass do not consider availability; thus, my estimates are relatively high. Factors that influence availability
vary by food type, duck species, and habitat (water column vs. benthic) (Cox and Kadlec 1995). For example, sago pondweed tubers have the highest energetic value of sampled food types but are usually only available to diving ducks. The maximum sediment depths that divers can extract tubers is 10cm; Anderson and Low 1976; Engelhardt 1997). In that case, adjusting my estimates of tuber abundance available to diving ducks would require knowing the vertical distribution of tubers in the sediment. Waterfowl managers can use this information to improve availability of tubers to diving ducks and ducks within other foraging guilds by lowering water levels during fall when other food resources have been reduced. More research to understand food availability to waterfowl is needed.

One objective of this study was to estimate and compare energetic carrying capacity of managed and unmanaged wetlands. Carrying capacity was higher in managed wetlands for all three foraging guilds. This is not unexpected as the total lands in managed wetlands exceed unmanaged wetlands and managers focus on waterfowl food production (Smith et al. 1989). However, unmanaged wetlands do have a value to waterfowl (total DUD’s for unmanaged wetlands in 2006 was 137,899,480 ducks) and provide important foraging habitat for wintering and migrating waterfowl. Unlike managed wetlands, unmanaged wetlands have the potential to increase in acreage if access to freshwater were to increase, as these areas are not defined by constructed levees.

Differences in the relative abundance of specific foods between managed and unmanaged wetlands indicate each meets the needs of ducks differently. The larger amount of leafy vegetation produced in unmanaged wetlands is relatively more
beneficial to American wigeon (*A. Americana*) and gadwall (*A. strepera*), species that rely heavily on aquatic vegetation in the fall (LeSchack et al. 1997; Mowbray 1999). While the larger seed and invertebrate biomass in managed wetlands favors Green-winged teal, mallard, and northern pintail primarily eat aquatic invertebrates and seeds (Austin and Miller 1995; Johnson 1995; Drilling et al. 2002).

Habitat loss in unmanaged wetlands reduced carrying capacity for all foraging guilds (Fig. 2.6). However, losses in DUDs were not proportional to area as food density was lower in unmanaged compared to managed habitats. For example, a loss of 80% of unmanaged habitat resulted in a 14% loss in dabbler, a 19% loss in diver, and a 34% loss in grazer DUDs, respectively. If food during the non-breeding season is limiting waterfowl population performance, this may have negative effects on duck survival and reproductive success (Heitmeyer and Fredrickson 1981; Kaminski and Gluesing 1987; Raveling and Hietmeyer 1989). Currently, unmanaged wetlands in the Great Salt Lake are unprotected and are under potential risk from increasing development and municipal demands.

My results quantify the value of unmanaged wetlands for waterfowl. Waterfowl managers can use my estimates to provide one measure of carrying capacity for Great Salt Lake wetlands for waterfowl. If the Intermountain West Joint Venture establishes waterfowl population objectives for the Great Salt Lake under the North American Waterfowl Management Plan, estimates of food abundance can be used to determine the acreage needed to meet the energetic needs of those populations. Future work is needed to better understand the value of unmanaged wetlands for both waterfowl and other waterbirds (e.g. American avocets and black-necked slits).
utilizing the area. Over five hundred thousand shorebirds use Great Salt Lake wetlands each year (Gorrell et al. 2005) and currently there is little information available about food abundance and availability to shorebirds. The water level of the GSL varies annually, increasing or decreasing the amount of available foraging habitat (Aldrich and Paul 2002). Work is needed to model how lake level variation influences the abundance and quality of unmanaged habitats in the lake.

Managed wetlands also need to be considered in future management and policy decisions. Over 65% of managed wetlands are in private ownership and most are exclusively used for recreational waterfowl hunting (Aldrich and Paul 2002). I found no consistent differences in food abundance between private vs. public lands indicating both provide valuable foraging habitat for ducks. If membership fails to support the maintenance costs for duck clubs, owners may be forced to sell their land or water rights. Water rights could be sold to increase development around the Great Salt Lake and valuable waterfowl habitat would be lost. Local government needs to consider alternatives to private duck clubs, such as, developing wildlife viewing areas or refuges. Future policy decisions need to establish an infrastructure that provides adequate protection of both managed and unmanaged wetlands.
Literature Cited


Appendix I

Power analysis to determine sample sizes for biomass estimates in the GSL wetlands during 2006

Power vs N1 by M2 with M1=0.0 S1=1175.9 S2=1175.9
Alpha=0.05 N2=N1 2-Sided T Tes

Figure 1. Variation in power (as sampling effort increases from 1-1000) to detect mean differences in total SAV biomass between managed and unmanaged wetlands around the GSL. Power calculations were based on N1=N2 (sampling effort in managed wetlands = sampling effort in unmanaged wetlands), α = 0.05 and used mean (1,216 kg/ha) and standard deviation (1,175.9) values from 2005. Each curve represents a difference between the two means (calculated % difference), red = a difference of 1(0.08%), blue = a difference of 251 (20%), green = a difference of 501 (41%), yellow = a difference of 751 (62%), and light blue = a difference of 1001 (82%).
Figure 2. Variation in power (as sampling effort increases from 1-1000) to detect mean differences in total invertebrate biomass between managed and unmanaged wetlands around the GSL. Power calculations were based on N1=N2 (sampling effort in managed wetlands = sampling effort in unmanaged wetlands), $\alpha = 0.05$ and used mean (14.5 kg/ha) and standard deviation (22.2) values from 2005. Each curve represents a difference between the two means (calculated % difference), red = a difference of 1 (7 %), blue = a difference of 4 (28%), green = a difference of 7 (48%), yellow = a difference of 10 (69%), and light blue = a difference of 13 (90%).
Figure 3. Variation in power (as sampling effort increases from 1-1000) to detect mean differences in total seed biomass between managed and unmanaged wetlands around the GSL. Power calculations were based on N1=N2 (sampling effort in managed wetlands = sampling effort in unmanaged wetlands), α = 0.05 and used mean (104.1 kg/ha) and standard deviation (95.1) values from 2005. Each curve represents a difference between the two means (calculated % difference), red = a difference of 1(.1 %), blue = a difference of 21 (20%), green = a difference of 41 (39%), yellow = a difference of 61 (59%), and light blue = a difference of 81 (78%).
Figure 4. Variation in power (as sampling effort increases from 1-1000) to detect mean differences in total tuber biomass between managed and unmanaged wetlands around the GSL. Power calculations were based on N1=N2 (sampling effort in managed wetlands = sampling effort in unmanaged wetlands), \( \alpha = 0.05 \) and used mean (351.8 kg/ha) and standard deviation (423.5) values from 2005. Each curve represents a difference between the two means (calculated % difference), red = a difference of 1(3 %), blue = a difference of 76 (22%), green = a difference of 151 (43%), yellow = a difference of 226 (64%), and light blue = a difference of 301 (86%).
Appendix II

Figure 1. Duck population estimates based on surveys conducted on selected managed wetlands in the Great Salt Lake by Utah’s Division of Wildlife Resources for 2004-05 and 2005-06 for diving ducks (top) and dabbling ducks (bottom).