STRATEGIES FOR MANAGING ENDANGERED WATERBIRDS ON HAWAIIAN NATIONAL WILDLIFE REFUGES

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DEDICATION

Dr. Mark W. Sayre

A Professor

An Inspiration

A Friend

He will live on in all of us.

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I am especially grateful to my family for their encouragement and support over the years. Lastly, thanks to Doreen Ho for her patience, understanding and support of my efforts. ʻE na kanaka Hawaiʻi, imua a loaʻa ka naʻauao no ka malama honua.

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CHAPTER 1

INTRODUCTION

The wetland national wildlife refuges in Hawaii were established to preserve and enhance habitat for Hawaii's four endangered waterbirds: the Black-necked (Hawaiian) stilt (*Himantopus mexicanus knudseni*), American (Hawaiian) coot (*Fulica americana alai*), Common (Hawaiian) moorhen (*Gallinula chloropus sandvicensis*) and Hawaiian duck (*Anas wyvilliana*). Survival of these endemic waterbirds depends on a multi-faceted approach that will maximize their production and survival. At the core of a long-term conservation program is the need to secure, maintain, and enhance suitable habitat.

Since humans first arrived in Hawaii, waterbird numbers have declined. Extensive modifications of natural wetlands and coastal areas by Hawaiian and European cultures reduced the quantity and quality of waterbird habitats. Consumption of waterbirds by humans and introduction of exotic predators to the islands caused further declines or extinction of several waterbird species (Berger 1981).

Early ornithologists, such as Henshaw (1902), recognized hunting and predation by mongoose as limiting factors decimating waterbird populations. All waterbird hunting was stopped by 1939 (Shallenberger 1977). Surveys and notes published by Schwartz and Schwartz

(1949, 1951, and 1953) further exposed the need for management of declining waterbird populations. Efforts to protect and restore endemic waterbird populations began with establishment of the first state wetland sanctuary (Kanaha Pond, Maui) in 1952, and captive breeding of the Hawaiian duck in 1962. All five endemic waterbirds were listed in the Endangered Species Preservation Act of 1966, calling further attention to declining waterbird populations.

Effective habitat enhancement programs have become more important as unprotected wetlands continue to diminish in extent and quality. The U.S. Fish and Wildlife Service (USFWS) established several waterbird refuges during the 1970's and encouraged further research on Hawaiian waterbirds. Additional studies in the late 70's and early 80's focused on breeding biology of stilts, moorhens, and coots (Coleman 1981, Nagata 1983, Byrd and Zeillemaker 1981, Byrd et al. 1985) However, little was known about vegetation and water management for Hawaiian waterbirds and their responses to changes in habitat.

A waterbird study conducted by the University of Missouri for the USFWS from 1985 - 1987 contributed to our understanding of how type, availability, dispersion, and quality of wetlands influence waterbird concentrations and behaviors. This study also developed habitat management techniques to help control weedy vegetation and to enhance desired hydrophytes and macroinvertebrate populations for waterbirds. However, additional waterbird management-related questions remained or required more research. These were addressed as my objectives for the present study on the James M. Campbell National Wildlife Refuge, Kii Unit, Oahu, Hawaii, including:

1. Evaluate the accuracy of waterbird survey methods;
2. Assess the response of waterbirds to vegetation manipulation;
3. Monitor nesting, productivity, and mortality factors of waterbirds;
4. Determine the relation between invertebrate numbers, fish populations, and water salinities.

This thesis consists of three discrete manuscripts, written in style and format in which they will be submitted to respective journals, and an appendix. Authors and abstracts for the three manuscripts have been omitted in the thesis. Other variations from manuscript form include consecutive pagination throughout the thesis, placement of tables and figures within the body of the text, and a general literature cited for all chapters combined at the end of the thesis.

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CHAPTER 2

VARIATION IN HAWAIIAN WATERBIRD DETECTABILITY AND ITS RELATION TO ACCURACY OF WATERBIRD SURVEYS

Statewide waterbird counts were initiated in the 1960's to monitor population trends of Hawaii's four endangered waterbirds: the Black-necked (Hawaiian) stilt (*Himantopus mexicanus knudseni*), American (Hawaiian) coot (*Fulica americana alai*), Common (Hawaiian) moorhen (*Gallinula chloropus sandvicensis*), and Hawaiian duck (*Anas wyvilliana*). These semi-annual counts typically consist of short observations at known waterbird habitats in the state. Observers drive through an area and stop briefly (1 - 2 min.) to count waterbirds, or survey an area from a single high point. Results from these counts have been used to monitor waterbird population trends and to assess recovery plan objectives for these four endangered waterbirds (U.S. Fish and Wildlife Service (USFWS) 1985). However, these count data are highly variable between years and the accuracy of survey methods is not known. Timing of counts, level of rainfall, inter-island movements, and recruitment success during the year are thought to account for fluctuations in waterbird count data (Englis 1988) Additionally, observability of waterbirds on a wetland may vary because of differences in waterbird behavior and vegetative structure. Thus, the objectives of my study were to examine the variation in waterbird detectability and its relation to accuracy of waterbird surveys.

Study Area

The study took place on the Kii Unit (47ha) of James Campbell National Wildlife Refuge (NWR) located on Oahu's North Shore. The Kii Unit consists of seven impoundments (A - G) and several ditches which provide important habitat for Hawaiian waterbirds (Fig. 2.1). Impoundments vary in depth, size, water chemistry, and vegetative cover. Water is supplied to impoundments from ditches and wells by windmills and pumps. Impoundment water levels are maintained by stop-log control structures. Impoundment water levels are seasonally adjusted according to waterbird nesting phenology and vegetation management needs. However, in general, most impoundment water levels are maintained high from fall through winter to accommodate wintering waterfowl and to retard vegetative growth. Slow drawdowns are initiated in mid March, providing increased waterbird nesting and feeding habitat.

Methods

I selected four impoundments, B (4.1ha), E (1.8ha), G (4.0ha), and a portion of A (2.0ha), to survey because they represented a range of wetland vegetation structures (Table 2.1). Waterbirds were monitored biweekly from

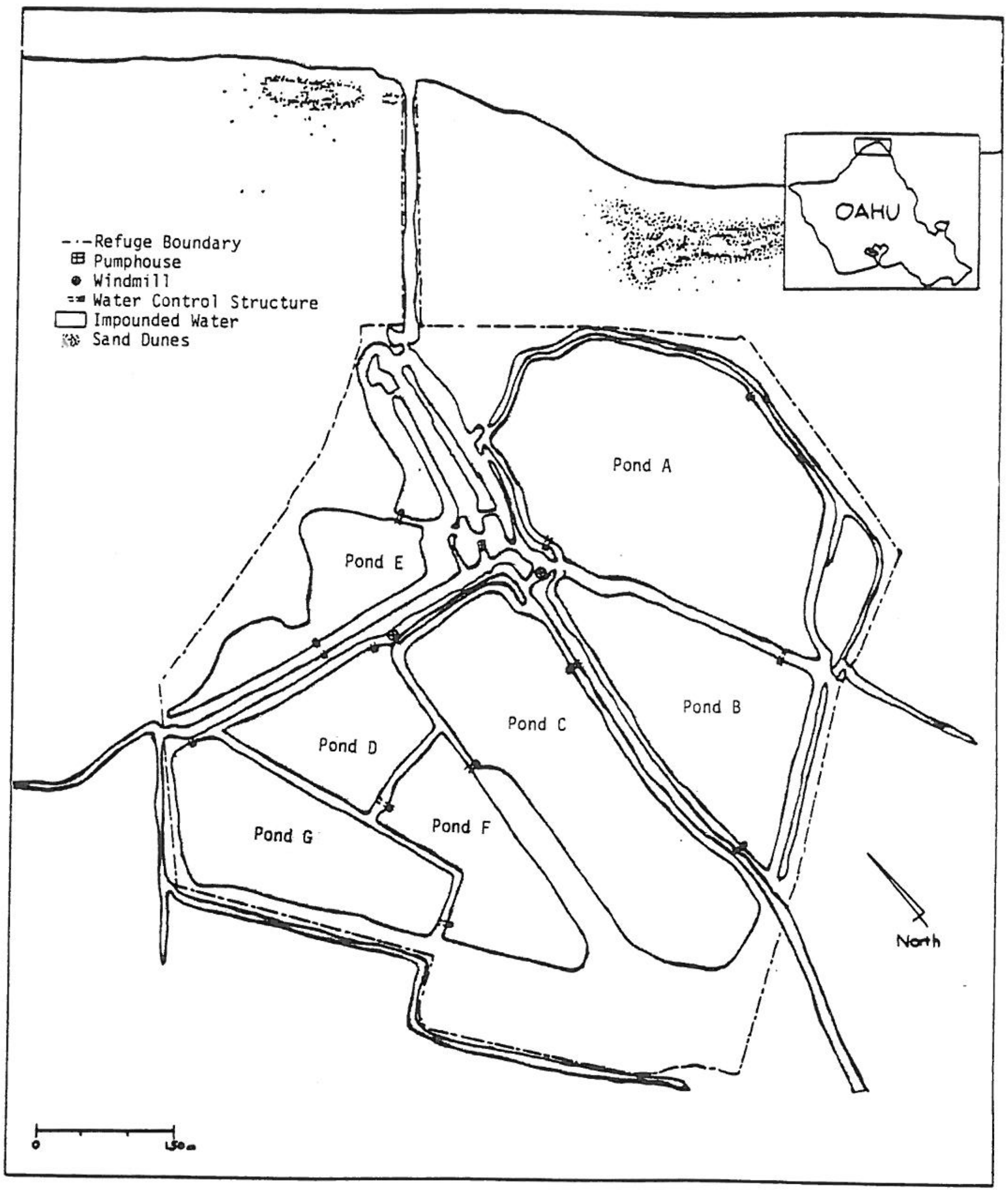

Fig. 2.1. Study area. Kii Unit, James Campbell National Wildlife Refuge, HI.

Table 2.1. Size and vegetative structure of four survey impoundments. 1987, Kii Unit, James Campbell National Wildlife Refuge, HI.

Impoundment	Hectares	% Cover	Major Veg.
A	2.0	91.0	Batis maritima
B	4.1	27.0	Brachiaria mutica Pluchea indica
E	1.8	74.0	*Pluchea* indica Batis maritima
G	4.0	22.0	Brachiaria mutica Paspalum distichum

April through August 1987 for 120-minute periods from stations overlooking each impoundment. Monitoring sessions were done either from a vehicle parked on a levee high point or from atop a 12m windmill. I was in position at least 5 minutes prior to beginning a count, and movement and noise was minimized to reduce bird disturbance.

Standardized cumulative counts began at 0800 and ended at 1000 hrs. During these counts, I scanned with 10 X 40 binoculars. Numbers of each species were totaled after the first two 5-minute intervals and then every 10 minutes until 120 minutes had elapsed. Counts were done under various weather conditions, ranging from clear and sunny to overcast and rainy. Waterbirds that flew over study areas without landing, and birds that flew from the study area upon my arrival or within the 5-minute wait period were not counted. Rarely did birds fly into the study area during the 120-minute survey period. However when this did occur, it was usually not possible on subsequent scans to differentiate these birds from those that were in the study area when observations began. Thus, these birds were included in counts if they remained on the study area. However, this happened infrequently (1 - 2 times) and it is believed the overall results were minimally affected.

I expressed cumulative number of birds by species for each time interval as a proportion of the total number of birds recorded during the 120-minute counts. Mean proportions of birds seen during each time interval (T5, T10, T20, - T120) were calculated by species for each impoundment. I determined the average time in minutes required to account for 80% of the total numbers seen of each species during a count. If this point was between two time intervals, I calculated it to the nearest 1-minute interval. I assumed 100% of the birds were seen by the end of each survey period. A two-way analysis of variance (ANOVA) was used to test for significant effects of impoundment, species, and impoundment by species on time required to observe 80% of the birds. Additionally, simple contrast tests were applied to determine which impoundments and species accounted for significant differences (Glass and Hopkins 1984). Vegetation cover maps compiled from aerial photographs, dated October 1986, were used to identify dominant and total vegetation cover.

Results

There was much variation in observability of waterbirds depending on species and impoundment. For all impoundments combined, moorhens were least observable with only 12% of the total seen during the first 5-minute interval (Fig. 2.2), whereas coots (69%) and stilts (58%) were most observable during the first 5-minute interval (Figs. 2.3 and 2.4). Hawaiian ducks typically flew from study impoundments when I arrived at a census station, therefore I have no comparable data for this species.

Eighty percent of total birds seen for each species for all impoundments combined was between 20 - 30 minutes for coots (Fig. 2.3) and stilts (Fig. 2.4), and between 70 - 80 minutes for moorhens (Fig. 2.2). Observation periods of at least 60 and 110 minutes were needed to always account for maximum numbers of stilts and coots, respectively. Further, based on my knowledge of moorhen territories from field observation, I believe a 120 minute observation period did not always account for 100% of moorhens on an impoundment. Hawaiian coots and stilts were not significantly different in time required to observe 80% of their numbers ($P > 0.05$, $F = 0.024$). However, time needed to observe 80% of stilts and coots were significantly less than for moorhens ($P < 0.01$, $F = 46.312$), ($P < 0.01$, $F = 48.477$), respectively.

Waterbird detectability also varied between impoundments (Table 2.2). Time required to observe 80% of waterbirds on an impoundment was greatest on the most heavily vegetated impoundments, A (91% cover) and E (74% cover), and least on impoundments B (27%) and G (22%) (Table 2.1). Although impoundment G had less vegetation cover than B, more time was required to observe 80% of waterbirds in G. This was probably due to the nearly contiguous pattern of vegetation dispersion around the perimeter of impoundment G that provided abundant cover for waterbirds. In contrast, vegetation in impoundment B was scattered in small clumps throughout the impoundment. Time required to observe 80% of the waterbirds differed significantly between impoundments ($P < 0.01$, F - ratio = 12.442). Time required to observe 80% between species was also significant ($P < 0.01$, F - ratio = 31.902). However, interaction of species and impoundments was not significant ($P > 0.05$, F - ratio = 0.139) (Fig. 2.5).

Discussion

Variation in Hawaiian waterbird detectability according to species and vegetative cover in a wetland indicates that current semi-annual waterbird surveys provide large underestimates of waterbird numbers. Waterbirds not seen within the first few minutes are missed during these short-duration counts. Stilts are undercounted the least because they do not usually evade observers and their contrasting colors allow for high detectability. Coots are easily counted when they are floating on open water but can be overlooked when they are in more vegetated wetlands. Common moorhens are missed by observers most frequently because of their secretive behaviors and their occurrence in thickly vegetated wetlands (USFWS 1985). Sattler and Bart (1984) found that raptor survey efficiency decreased with low bird density, because of reduced observer effort. This may be contributing to low counts for moorhens.

Observers may be overlooking less visible, secretive moorhens in thickly vegetated areas which contain more visible species such as stilts or coots. Scott and Ramsey (1981) found surveyors of forest birds underestimate the most common birds when counting all species present. Surveyors that divided counting responsibilities amongst the most common species obtained better estimates. Since moorhens are more difficult to see compared to other waterbird species in Hawaii, surveyors might consider counting in teams, splitting survey responsibilities of more common waterbirds, but always looking for and counting moorhens. Moorhen numbers could then be checked and compared amongst team members prior to moving on to the next survey point.

While current statewide semi-annual counts show general trends in waterbird populations, they do not provide accurate population estimates. However, it is probably not logistically practical for personnel to spend the time required to obtain accurate estimates. Information on population trends may be sufficient for monitoring general year to year population changes; however, accurate estimates of endangered waterbird population numbers are needed periodically to assess management needs and to guide recovery plan efforts.

Management Recommendations

Several changes to the semi-annual Hawaiian waterbird survey methodology need to be implemented to increase count accuracy and to better compare results between years. Training sessions need to be held prior to surveys to train new observers and re-familiarize experienced observers with survey methodology. Survey teams might consider dividing counting effort between species to potentially improve estimates of more common species. Survey duration at each observation point should be standardized (Englis 1988). Further, observation points must be clearly marked, mapped, and copies of maps provided to each survey team. Intensive counts, providing more accurate population estimates, should be done every two to three years to determine whether recovery plan goals are being met and maintained.

Additional research is required to improve our understanding of waterbird behaviors and detectability. Although preliminary playback census attempts using a recording of the common moorhen failed to elicit responses from moorhens on Oahu (Nagata 1983), playback calls have been used successfully to obtain indices of abundance for several rallid species in the continental U.S. (Brackney 1979, Marion et al. 1981, Manci and Rusch 1988). Additional research is needed to develop more efficient survey methods for this elusive species.

Similarly, there are no accurate survey data for Hawaiian ducks. The high sensitivity of Hawaiian ducks to disturbance by surveyors precludes accurate survey counts by current methods. Further, the similar appearance of Hawaiian ducks and mallards and the occurrence of feral mallards on some islands (Griffin et al. 1990), confounds species identification and accurate counts of Hawaiian ducks. Much additional data are also needed on waterbird movements between wetlands and other islands. These data, combined with more accurate waterbird surveys, should improve our understanding of waterbird ecology and provide tools for meeting recovery plan goals.

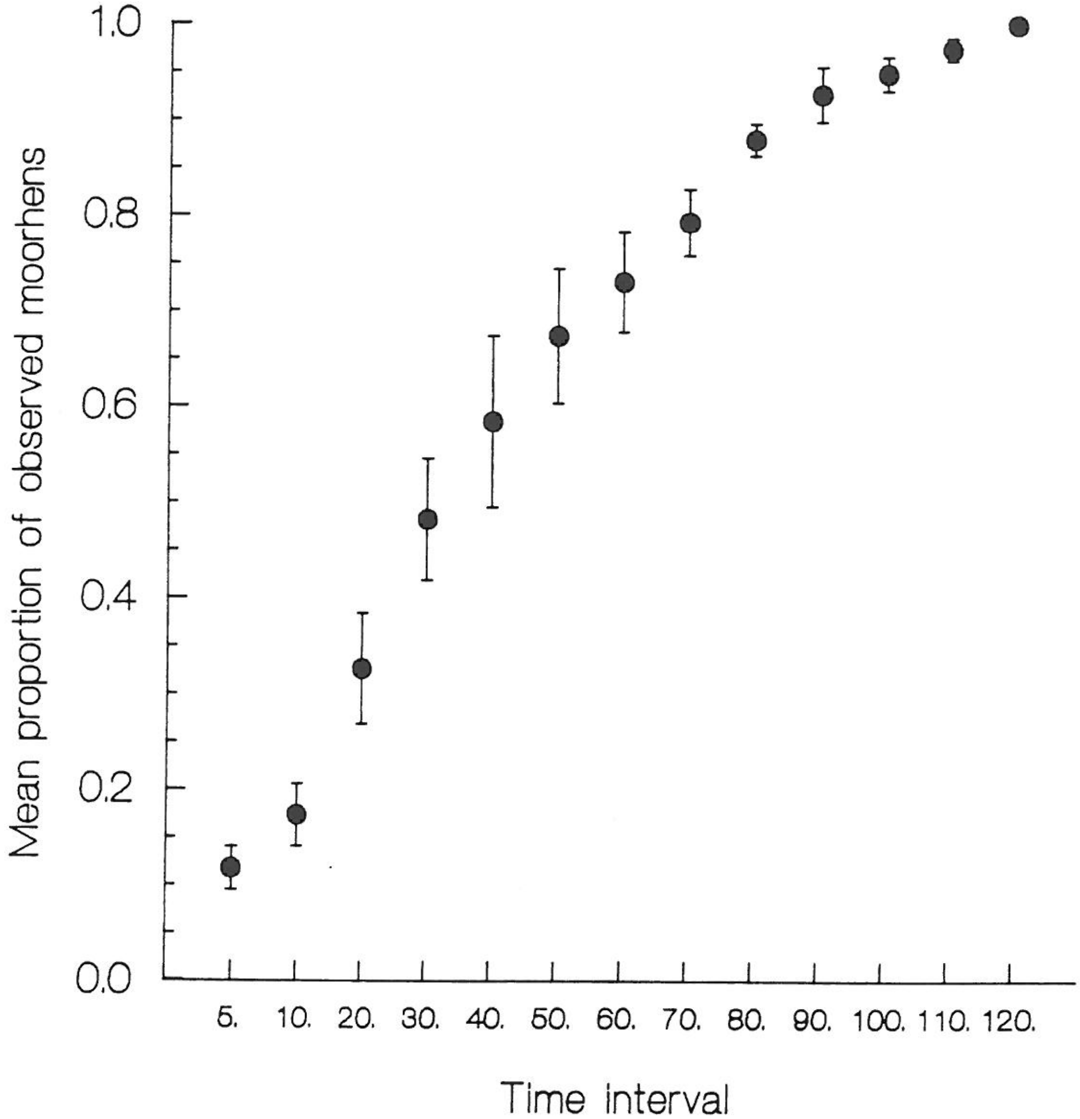

Fig. 2.2. Proportion of Hawaiian moorhens seen by time interval, impoundments combined. 1987, Kii Unit, James Campbell National Wildlife Refuge, HI.

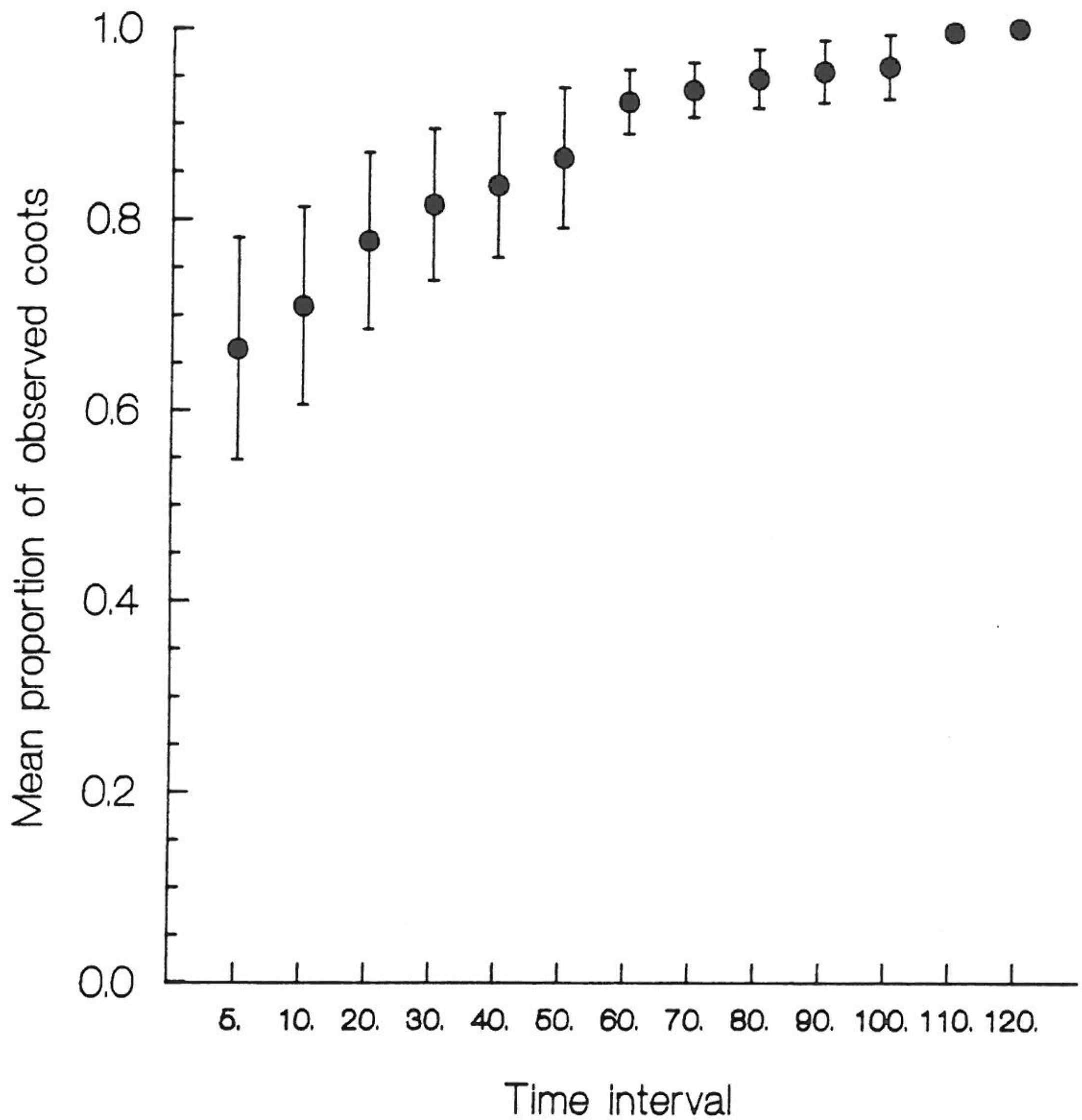

Fig. 2.3. Proportion of Hawaiian coots seen by time interval, impoundments combined. 1987, Kii Unit, James Campbell National Wildlife Refuge, HI.

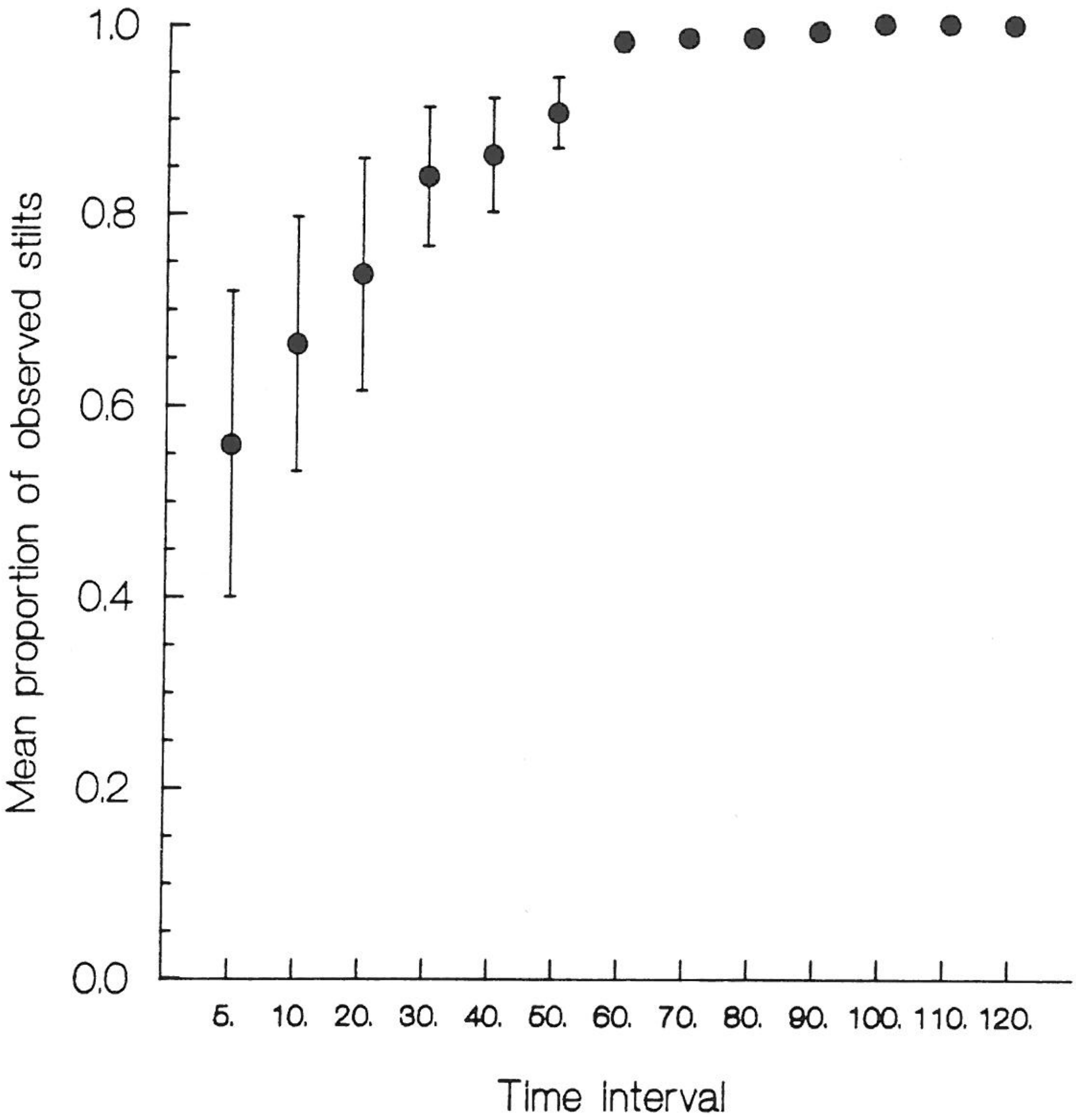

Fig. 2.4. Proportion of Hawaiian stilts seen by time interval, impoundments combined. 1987, Kii Unit, James Campbell National Wildlife Refuge, HI.

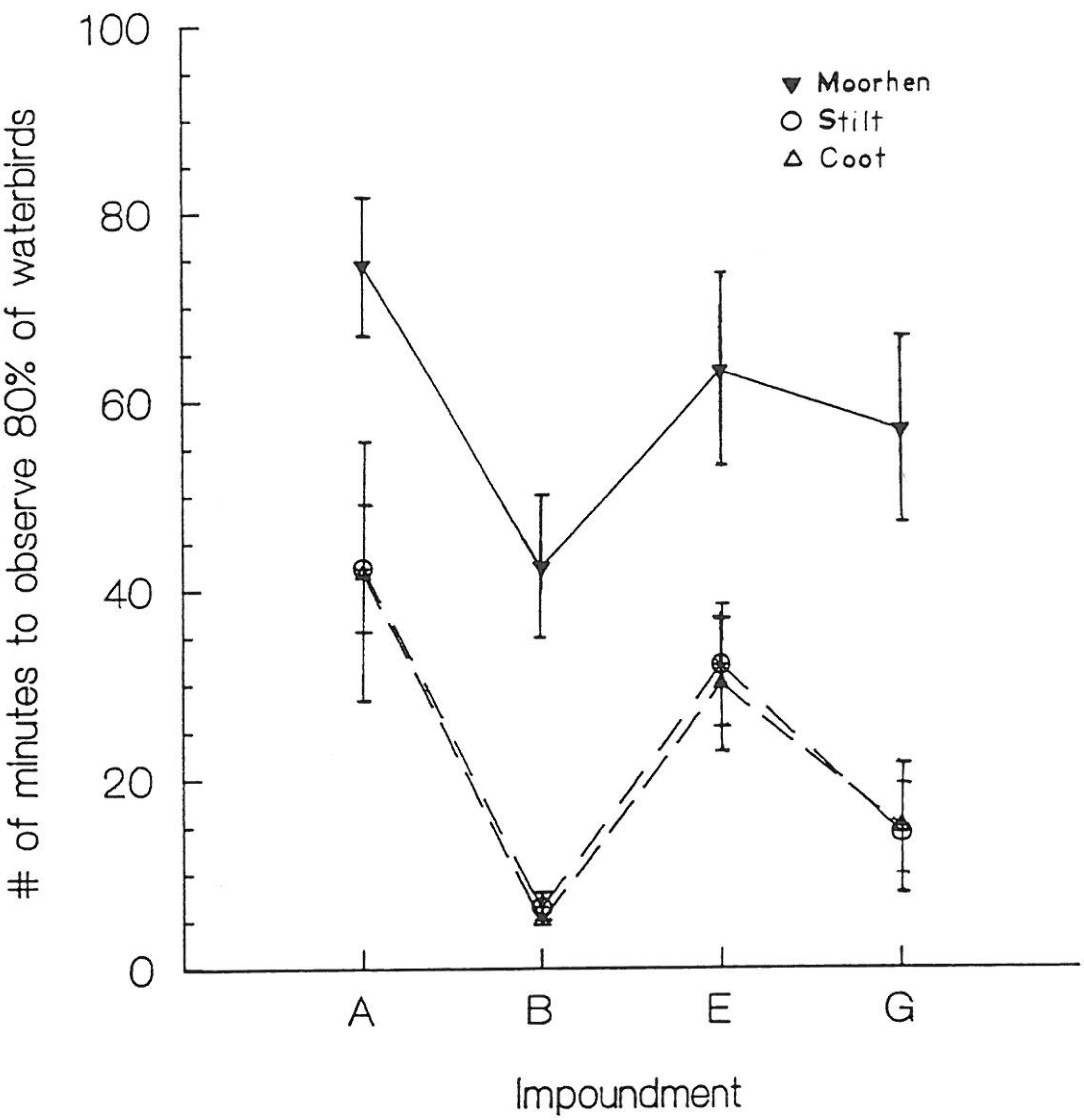

Fig. 2.5. Minutes to observe 80% of waterbirds by impoundment. 1987, Kii Unit, James Campbell National Wildlife Refuge, HI.

Table 2.2. Average minutes required to observe 80% of Hawaiian stilts (HS), Hawaiian coots (HC), and Hawaiian moorhens (HM) by impoundment. 1987, Kii Unit, James Campbell National Wildlife Refuge, HI.

		Impoundments			
		A	B	E	G
HS	Mean	42	6	32	14
	N	11	11	7	11
	SE	6.75	1.17	6.52	6.89
HC	Mean	42	5	30	15
	N	7	11	11	11
	SE	13.64	0	7.01	4.77
HM	Mean	74	42	64	57
	N	10	9	11	11
	SE	7.36	7.56	10.12	9.86

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CHAPTER 3

RESPONSE OF HAWAIIAN WATERBIRDS TO AN EXPERIMENTALLY MANIPULATED IMPOUNDMENT

The long history of modifications to Hawaii's wetlands by Hawaiian and European cultures has left only remnants of former natural wetlands. These wetlands are critical to the survival of Hawaii's four endangered waterbirds, the Black-necked (Hawaiian) stilt (Himantopus mexicanus knudseni), American (Hawaiian) coot (Fulica americana alai), Common (Hawaiian) moorhen (Gallinula chloropus sandvicensis), and Hawaiian duck (Anas wyvilliana). Further, man-made or altered sites, such as fish ponds, silting basins, reservoirs, and aquaculture facilities, have replaced natural wetlands and are now of primary importance to waterbirds (Griffin et al. 1990).

Continued urbanization of lowland areas, particularly on Oahu, is accelerating the conversion or alteration of remaining wetlands. Further, most of these wetlands are degraded by introduced plants. In the last 200 years, more than 4600 plant species have been introduced into Hawaii and over 600 have become naturalized (Smith 1985). Approximately 55% of the plants found in Hawaiian wetlands are exotic (Stemmermann 1981). Exposed to Hawaii's year-around growing conditions and lack of frost at lower elevations, these exotics often replace native species and form monotypic stands. Introduced plants such as California grass (Brachiaria mutica), cattail (Typha angustata), pickle weed (Batis maritima), Indian marsh fleabane (Pluchea indica), and American mangrove (Rhizopora mangle) present serious problems in many Hawaiian wetlands by reducing the interspersion of open water and vegetated areas (Stemmermann 1981). Reduced interspersion adversely affects the suitability of these wetland habitats for Hawaiian waterbirds.

Continental studies indicate that maximum levels of avian use and production occur when emergent macrophytes and open water cover roughly equal areas in an interspersed pattern (Weller and Spatcher 1965, Weller and Fredrickson 1974). Highest densities of dabbling duck pairs were associated with 50:50 cover:water ratios (Kaminski and Prince 1981, Murkin et al. 1982). Moist - soil management techniques developed on a Hawaiian National Wildlife Refuge (NWR) during 1985 - 1986 demonstrated that many aggressive exotic plants can be controlled and interspersion of water and vegetation increased (Griffin et al. 1990). My study objective was to determine Hawaiian waterbird responses to vegetation manipulations in order to optimize waterbird use of an artificial wetland.

Study Area

The study was conducted on the Kii Unit of James Campbell National Wildlife Refuge on the North Shore of Oahu. The Unit, consisting of seven impoundments, is managed as a complex for production of different foods and covers to attract various waterbird species (Fig. 2.1). The primary 2.0 ha study section was located in the southern end of impoundment A. Prior to vegetation manipulation, this area consisted of approximately 75% pickle weed.

Methods

The study was conducted in three phases. Phase 1 (April - July 1987) consisted of pre-manipulation observations of waterbird use. Phase 2 (August 1987 - March 1988) entailed dewatering the impoundment, vegetation manipulation, and reflooding. Phase 3 (April - July 1988) involved of post-manipulation observations of waterbird use.

Vegetation Manipulation

In late August 1987, dewatering of the study area was initiated in preparation for vegetation manipulation. Dewatering was by evaporation and 75% of the area was dry by late October 1987. This area was then mowed by tractor with a bush hog until a 50:50 ratio of vegetation to open water interface was achieved. The mowing pattern was irregular with numerous vegetation islands (3 - 30m diameter), and vegetation strips (1 - 3m wide by 3 - 20m long). The eastern quarter of the study area was too wet to mow.

Waterbird Use

I evaluated waterbird use of the study area pre- (phase 1) and post- (phase 3) manipulation. Criteria for measuring use consisted of biweekly waterbird counts and systematic waterbird nest searches every three weeks. Status of nests were checked at least weekly. Surveys of waterbird numbers were conducted for 30 minutes using binoculars from a vehicle parked at a single observation station. I typically arrived at a survey station at least 5 minutes prior to starting a count to allow birds to settle. Counts were done under various weather conditions ranging from clear and sunny to overcast and rainy. Each cumulative count began at 0800 with total numbers of observed birds being recorded at 0805, 0810, 0820, and 0830. Hawaiian ducks were not included in these counts because they often flushed off the study area when we arrived at the survey station. Refuge-wide waterbird surveys were conducted monthly. These counts consisted of driving and stopping frequently (1 - 2 minutes) around impoundments while scanning vegetation for waterbirds. We used two-sample t-tests (Sokal and

Rohlf 1981) to examine differences in waterbird surveys for both refuge-wide and 30 minute counts.

Results and Discussion

There were differences in waterbird use of the study area between phases 1 and 3. Only two nests were found in the study area prior to vegetation manipulation, while 18 nests were found post-manipulation (Table 3.1). Additionally, all four waterbird species nested on the study area post-manipulation, whereas only Hawaiian stilts nested in the area prior to vegetation manipulation. Stilt nests also increased substantially post-manipulation in response to increased mudflat habitats created by mowing.

Numbers of waterbirds using the study section of impoundment A also increased between phases 1 and 3. Hawaiian coot and stilt numbers were significantly higher ($P < 0.01$) post-manipulation (Table 3.2). Although mean number of moorhens increased over 60% in phase 3, this difference was not significant ($P > 0.10$).

This increased use of the study area by these three waterbird species was not a function of increased waterbird use of Kii Unit as a whole. Monthly waterbird surveys conducted on similar dates in both years showed no significant increases in stilt and moorhen numbers, $P > 0.05$, and $P > 0.05$, respectively. Further, coot numbers significantly decreased for refuge-wide surveys between phases 1 and 3 ($P < 0.05$), (Table 3.3). This decrease in coot use of the refuge as a whole, combined with a significant increase in observed coots in the manipulated study section of impoundment A further illustrates the increased use of the manipulated section by waterbirds.

These findings support the hypothesis that monotypic stands of vegetation in Hawaiian wetlands decreases waterbird use and production. The results also support conclusions of other studies that maximum levels of avian use and production occur when emergent macrophytes and open water are in equal proportions in an interspersed pattern (Weller and Spatcher 1965, Weller and Fredrickson 1974, and Murkin et al. 1982). Average distances of Hawaiian stilt (120cm), coot (143cm), moorhen (212cm), and duck (217cm) nests to open water measured on Kii Unit from 1987 - 1988 indicate that Hawaiian waterbirds are unlikely to nest in interiors of large monotypic vegetation stands (Table 3.4).

Management Recommendations

The primary recovery objective of the Hawaiian waterbirds recovery plan is to maintain minimum populations of at least 2,000 each of Hawaiian stilts, coots, moorhens, and ducks for three consecutive years in habitats and island distributions as of 1976 (U.S. Fish and Wildlife Service (USFWS) 1985). A critical step in should be done every three to four years to assess recovery plan goals.

Table 3.1. Numbers of waterbird nests on the study section in response to increased interspersion of vegetation and water . 1987 - 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

	Phase 1 (pre-manipulation)	Phase 3 (post-manipulation)
Hawaiian coot	0	5
Hawaiian moorhen	0	3
Hawaiian stilt	2	8
Hawaiian duck	0	2

Table 3.2. Mean numbers of Hawaiian coots (HC), Hawaiian moorhens (HM), and Hawaiian stilts (HS) counted on study area pre (phase 1) - and post (phase 3) -manipulation of vegetation. 1987 - 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

Phase	Species	Mean*	S.E.	P
1	HC	1.63	0.73	
3		6.25	1.24	<0.01
1	HM	2.13	0.88	
3		3.50	0.98	>0.10
1	HS	2.75	0.62	
3		13.50	2.78	<0.01

* based on 8 counts each year.

Table 3.3. Refuge-wide monthly waterbird counts for Hawaiian coots (HC), Hawaiian moorhens (HM), and Hawaiian stilts (HS). 1987 - 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

Year	Species	Mean	S.E.	P
1987	HC	256.50	38.90	
1988		150.70	16.91	<0.05
1987	HM	11.00	0.71	
1988		14.00	5.52	>0.10
1987	HS	87.75	6.52	
1988		96.50	10.87	>0.10

Table 3.4. Mean distance (cm) from nest center to nearest open water for Hawaiian stilts (HS), coots (HC), moorhens (HM), and ducks (HD). 1987 - 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

Species	N	Mean	S.E.
HS	79	120	12.21
HC	60	143	26.89
HM	35	212	38.35
HD	51	217	26.44

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Chapter 4

NEST SUCCESS AND BROOD SURVIVAL OF ENDANGERED HAWAIIAN WATERBIRDS

Increased levels of recruitment and survival are paramount to the long-term viability of Hawaii's four endangered waterbirds: the Black-necked (Hawaiian) stilt (*Himantopus mexicanus knudseni*), American (Hawaiian) coot (*Fulica americana alai*), Common (Hawaiian) moorhen (*Chloropus gallinula sandvicensus*), and Hawaiian duck (*Anas wyvilliana*). While the need to secure, maintain, and enhance suitable habitats are at the core of a long-term conservation program, proximate factors limiting waterbird productivity and survival need to be identified and managed. Habitat loss and predation are considered primary causes of declines in waterbird numbers (Shallenberger 1977). Diseases, such as Salmonella and Botulism, have also been identified as potential mortality factors (Coleman 1981). Although several studies have been conducted on Hawaiian waterbirds in recent years (Byrd and Zeillemaker 1981, Byrd et al. 1985, Coleman 1981, Nagata 1983), specific factors limiting waterbird recruitment and survival have not been well quantified. Thus, the objectives of this study were to

1) determine nesting chronology of Hawaiian waterbirds,
2) monitor nesting success and cause of nest failure, and

3) determine cause of chick loss and fledging success for Hawaiian waterbirds.

Study Area

The study took place on the Kii Unit (47ha) of James Campbell National Wildlife Refuge (NWR) located on Oahu's North Shore. The Unit is comprised of seven impoundments with water control capabilities (Fig. 2.1). Predator trapping is done sporadically throughout the year. Trapping levels increase in March and continue at high levels through August at about 160 trap nights per month. Vegetation around impoundment edges is mowed regularly to reduce predator cover on levees.

Methods

Complete nest searches of the study area were done every 3 weeks from January 1985 - December 1986, April - August 1987, and January - August 1988. The three weeks between nest searches was short enough so that eggs in nests initiated a day after a search would not hatch before the next search. Each nest was mapped and visited 1 - 2 times per week until all eggs hatched or I determined nest fate. I recorded nest fate and causes of eggs loss. If at least one egg hatched, the nest was considered successful. Thus, successful nests could also have an egg loss code if some eggs did not hatch. If all eggs hatched successfully, I recorded no loss code. I identified egg predators by studying characteristics of the depredated eggs or nests. Predation sign by rodents was similar to small Indian mongoose (*Herpestes auropunctatus*) with eggs cracked open and contents licked out, except scrapes and punctures made by rodent incisors were square in appearance. Eggs with punctures approximately 10mm on one side and 5mm on the other, or eggs cracked open with contents not licked clean, were attributed to avian predators.

Data from the 1985 - 1986 study period were used to document nesting chronology for coots, moorhens, and Hawaiian ducks. Data from 1985 - 1988 were used to assess stilt nesting chronology because an investigator was always present during the stilt nesting season. I described nesting chronology by calculating a cumulative nest count for each month by nest initiation date. Nest initiation dates were calculated by back-dating based on 1 egg being laid each day and average incubation periods: stilts, 24 days; coots, 23 days; moorhens, 23 days (Shallenberger 1977); and ducks, 28 days (Swedberg 1967). I summarized mean clutch sizes, mean number of young hatched per nest, hatching success, nest fates, and egg loss codes by species to identify levels of production and to identify egg and chick mortality factors.

In 1988, I attempted to observe selected broods twice a week for at least 30 minutes. Hawaiian ducks left nests shortly after their eggs hatched so data on broods could not be collected for this species. I selected broods based on my ability to observe them without causing undue stress to the group. If alarm calls were given for more than 15 minutes, observation was abandoned. I used several methods to approach broods depending on where they were located. Some observations were done from the top of a windmill using 10x40 binoculars, a vehicle parked on a levee, or a slow moving tractor, all of which birds seemed accustomed to seeing routinely as "normal" refuge activities. I used blinds in areas which allowed a close approach and vantage point. Some coot and moorhen broods located in the middle of impoundments were observed successfully from under camouflage netting. I moved slowly and stopped frequently, while mostly submerged under the water surface. I also wore camouflage clothing and a hat whenever approaching a brood from levees or within impoundments. When I saw less chicks in a brood, immediate areas were searched for signs of mortality. I examined dead chicks for external injuries such as broken bones, bites, and peck punctures. Chicks were summarized by age at death or disappearance, with hatch date based on day 1. I calculated the proportion of fledged birds per pair for each species.

Results

On Kii Unit, 657 nests were located for all species of which 642 had eggs. Numbers of nests and placement on Kii differed between species and impoundments (Table 4.1). Stilts and Hawaiian ducks accounted for 37.3% and 27.2% of total nests, respectively. Coots and moorhens nested at lower levels, 21.8% and 13.7%, respectively. Stilts initiated most nests on impoundments A, B, C, and F as they contained the most mudflat areas. Hawaiian ducks usually nested in impoundments A, B, and C. Moorhens nested most frequently in ditches and in impoundments A, E, and G. The majority of coot nests were initiated on impoundment G and in Unit ditches.

Chronology

The stilt nesting season was short and intense compared to other Hawaiian waterbird nesting periods. For all years, nesting began in March, peaked in May, and ended in July (Fig. 4.1). I recorded one nest initiated in September. Coot nesting was highest from February - May and was lowest from June - October (Fig. 4.2). Peak nesting for moorhens was in March and October with low levels through August and September (Fig. 4.3). Hawaiian ducks initiated most of their nests in January, February, and October (Fig. 4.4). Lowest nesting levels for all species occurred in July and August.

Clutch Size

Stilts had a mean clutch size of 3.4 eggs which was lower than the value noted by Coleman (1981) (Table 4.2). Mean number of eggs produced per clutch was similar for coots and moorhens. Average clutch size for coots was the same as that recorded by Byrd et al. (1985) for Hawaiian coots, and was lower than the average 8.2 eggs from 17 North American studies (Byrd et al. 1985). My mean clutch size for moorhens was less than the five year average of 5.6 eggs per clutch obtained in a Hawaiian moorhen study (Byrd and Zeillemaker 1981), and less than the average of 8.4 eggs per clutch from seven North American studies (Byrd and Zeillemaker 1981). Hawaiian duck clutch sizes, like other waterbirds in this study, were comparatively lower than clutch sizes of 8 - 8.4 reported in other studies (Schwartz and Schwartz 1953, Swedberg 1967). However, these clutch sizes for Hawaiian ducks were within ranges reported from continental studies on mallards (6.8 - 9.6 eggs per clutch) (Bellrose 1976), the presumed ancestral stock of Hawaiian ducks. Average number of chicks per clutch produced by stilts was no different than 3.6 chicks per clutch reported by Coleman (1981). Coots, moorhens, and Hawaiian ducks hatched similar numbers of young per clutch (3.2 - 3.5) in this study (Table 4.2).

Hatching Success

Hatching success on an egg basis (number of eggs hatched/number of eggs laid) was highest for moorhens and coots and was lowest for Hawaiian ducks (Table 4.3). Hatching success on a nest basis was again highest for moorhens and coots (Table 4.4). Additionally, moorhen hatching success (76.7%) was higher than mean hatching success (61.5%) of six North American studies reported by Byrd and Zeillemaker (1981). However, coot hatching success (67.1%) was lower than mean hatching success (92%) of 12 North American studies (Byrd et al. 1985).

Most nest failures for all species were attributed to predation (Table 4.4). Stilts lost a higher percentage of their nests to flooding compared to the other three waterbird species. Hawaiian ducks abandoned 15% of their nests. This agrees with 10 - 15% desertion reported for mallards in the continental U.S. (Bellrose 1976). Moorhens showed the lowest rate of nest abandonment (1.1%). All waterbird species had relatively high proportions of egg loss from unhatched eggs. Hawaiian duck nests had the highest percentage of unhatched eggs, whereas moorhen nests had the lowest percentage of unhatched eggs.

Unidentified predators were responsible for most stilt, coot, and moorhen egg losses (Table 4.5). Dogs accounted for the highest loss on Hawaiian duck eggs in comparison to other species. Stilts and moorhens had the highest losses to bird predators, 19% and 20%, respectively. Moorhens and coots lost higher proportions of their eggs to mongooses than other species. Only stilts were affected by rat predation.

Stilt and coot chicks were most susceptible to mortality factors during the first two weeks of life (Table 4.6). Hawaiian moorhen data were too few to categorize age of chick disappearance. Causes of chick and adult mortality were difficult to collect. However, dogs and mongooses accounted for high proportions of losses (Table 4.7). The highest percentages of dead moorhen and coot chicks were found in nests with no evidence of injuries. A high percentage of chick and adult stilt deaths was also attributed to unknown causes. Of the known predators, dogs affected stilt chicks and adults, and adult coots most severely. However, owls caused a notable level of mortality on coots. A Short-eared owl (*Asio flammeus*) was observed eating the single adult coot lost to owl predation. The proportion of fledged birds in the observed brood sample was 0.243, 0.276, and 0.418 for stilts, coots, and moorhens, respectively (Table 4.8).

Discussion

Timing of Hawaiian waterbird nesting in intensively managed wetlands, such as the Kii Unit, depends on management of water levels. Nest failure can result when water levels are maintained too high or too low. Nests are easily flooded if water levels are allowed to rise unchecked, causing loss of nesting and feeding substrate. However, if water levels are too low, access to nests and chicks becomes easier for predators such as mongooses, cats, and rodents. Further, drying mudflats become unsuitable habitat for invertebrate populations, which are believed to be an important food source for waterbirds. Inattentiveness by refuge staff to water level manipulations and the poor condition of some water structures contributed to overall nest loss by flooding.

Stilts lost 7% of their eggs to mongooses. This level of mongoose predation seems low considering stilts nest on mudflats often contiguous with impoundment levees. The aggressive mobbing of predators by stilts may deter potential predation. Only stilts lost eggs to rat predation. These losses occurred at nests that were located on contiguous mudflats supporting dense stands of barnyard grass (Echinochloa crus-galli), and makai (Scirpus maritimus). Rats fed on makai rhizomes and Barnyard grass seeds located around nests. Eggs were often removed from nests and taken a few meters away where ends were bitten open and egg contents licked out.

Stilts lost 19% of their failed eggs to avian predators. Bird predators such as black-crowned night heron (Nycticorax n. hoactli), cattle egret (Bubulcus ibis), ruddy turnstone (Arenaria interpres), and common mynah (Acridotheres tristis) have been implicated as egg predators (Berger 1981). Ruddy turnstones and black-crowned night herons were often seen near nesting stilts. Although Coleman (1981) recorded no response from incubating stilts to herons nearby, I witnessed aggression between these two species.

Rate of nest abandonment by Hawaiian ducks coincides with the highest value of nest desertion for mallards (Bellrose 1976). I believe both genetic dilution through hybridization with feral mallards and disturbance caused by predators (dogs) could be influencing this high nest abandonment rate. Dogs often enter refuges in groups and run through impoundments. Egg loss results from flushing hens and rolling eggs out of nests, or completely smashing them. Further, dogs are a serious predator of both adult and young waterbirds. Coots are especially vulnerable on land as they are less agile than other waterbirds. Nesting islands maintained for Hawaiian ducks are often surrounded by impoundment water which usually deters other mammalian predators such as mongooses. However, in early fall and late summer when impoundments are often dry for vegetation manipulation, duck nests are vulnerable to mammalian predation.

It was difficult to find dead or missing chicks, and determine cause of death. However, relatively high proportions of coot and moorhen chicks were found dead in nests. Most of these birds found at nests had no evidence of injury.

Both nonbreeding coots and moorhens from previous broods are known to help raise younger siblings (Gibbons 1987, Eden 1987, Wood 1974). Additionally, moorhens and coots are known to split broods between parents for foraging. I believe the difficulty in both following moorhen broods through time, and recovering missing chicks is reflected in this behavior and is compounded by the species' secretive nature.

It is important to note that the results reported herein apply only to an area where habitat is intensively managed for positive waterbird recruitment and maintenance. Unmanaged areas probably have much lower waterbird nesting activity, lower productivity, and higher mortality rates.

Management Recommendations

The potential for increased waterbird productivity can be realized through timing of management actions. Managers in Hawaii need to control predators, such as mongooses, dogs, and cats on a year-around basis by trapping. Trapping levels should intensify during periods of waterbird nesting and whenever predator sign is discovered. Further, moats should be dug and maintained around impoundment parameters to deter entry by predators and to provide refuge to invertebrates when impoundments are dried out in preparation for vegetation management activities.

Vegetation control is important to reduce predator cover and to produce favorable mixtures of plant substrate for waterbird nesting and feeding. This can be accomplished through herbicide application, controlled burning, seeding, and water level manipulation.

Water level management greatly affects waterbird nesting success. Managers must maintain water levels low enough to provide habitat for waterbird nesting and feeding, and high enough to deter predators. Water structures must be maintained in good condition to facilitate water level management. Further, managers need to be aware of nest locations to avoid inadvertently flooding nests when changing water levels. Strategies for controlling high populations of black-crowned night herons and cattle egrets on the aquafarms adjacent to the refuge should be established. Waterbird nesting and recruitment should be monitored every two to three years to measure progress of management and recovery plan goals.

Table 4.1. Number of nests initiated by Hawaiian stilts (HS), Hawaiian ducks (HD), Hawaiian coots (HC), and Hawaiian moorhens (HM) by impoundment. 1985-1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

Impoundment	HS(n=245)	HD (n=179)	HC (n=143)	HM (n=90)
A	40	50	29	15
B	50	45	2	3
C	42	39	8	10
D	8	2	3	2
E	4	14	22	16
ED*	1	--	5	3
F	60	12	2	3
G	27	14	37	17
X**	13	3	35	21

* = outlet ditch for impoundment E
** = all other ditches combined

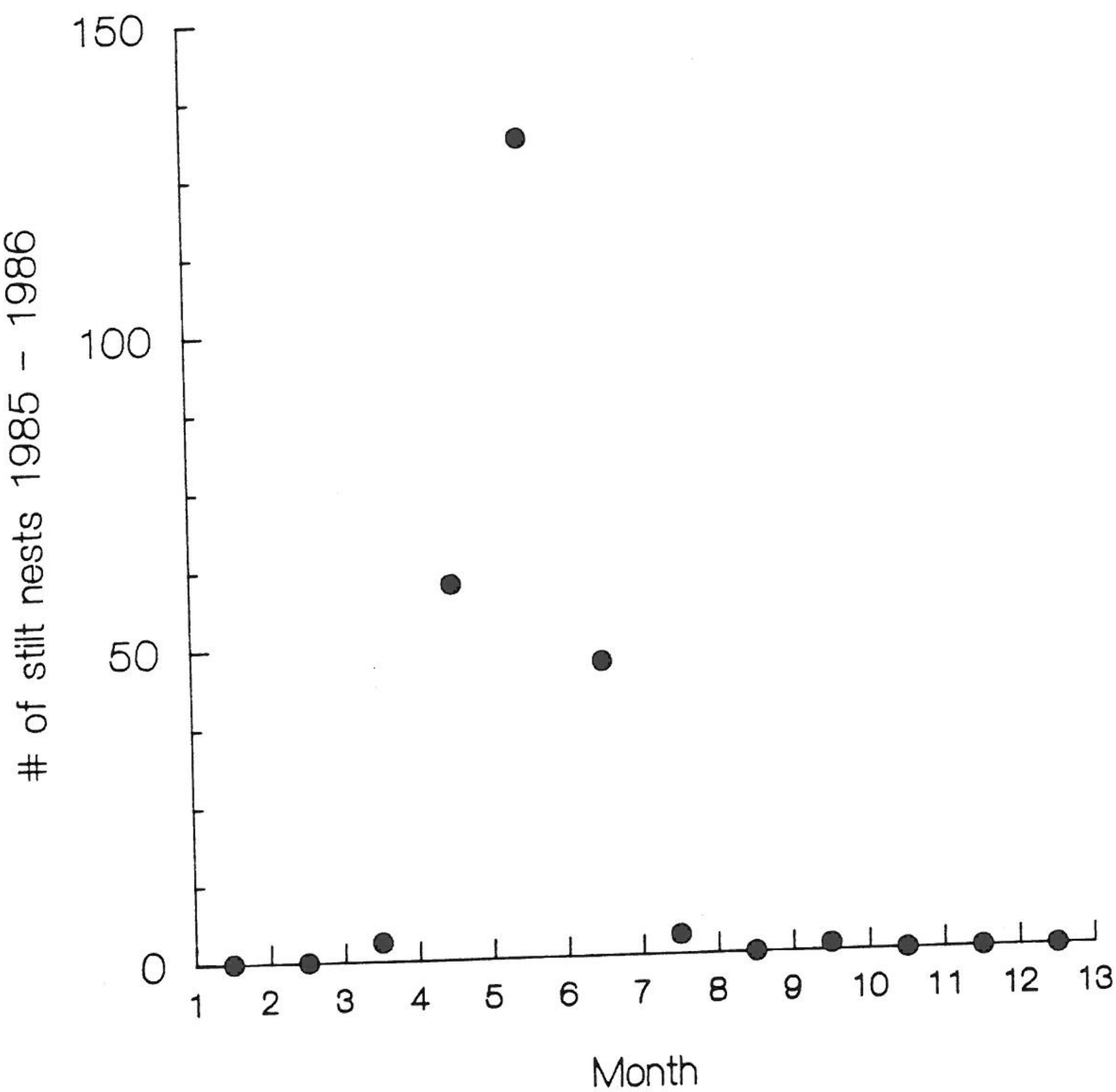

Fig. 4.1. Number of stilt nests initiated by month. 1985 - 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

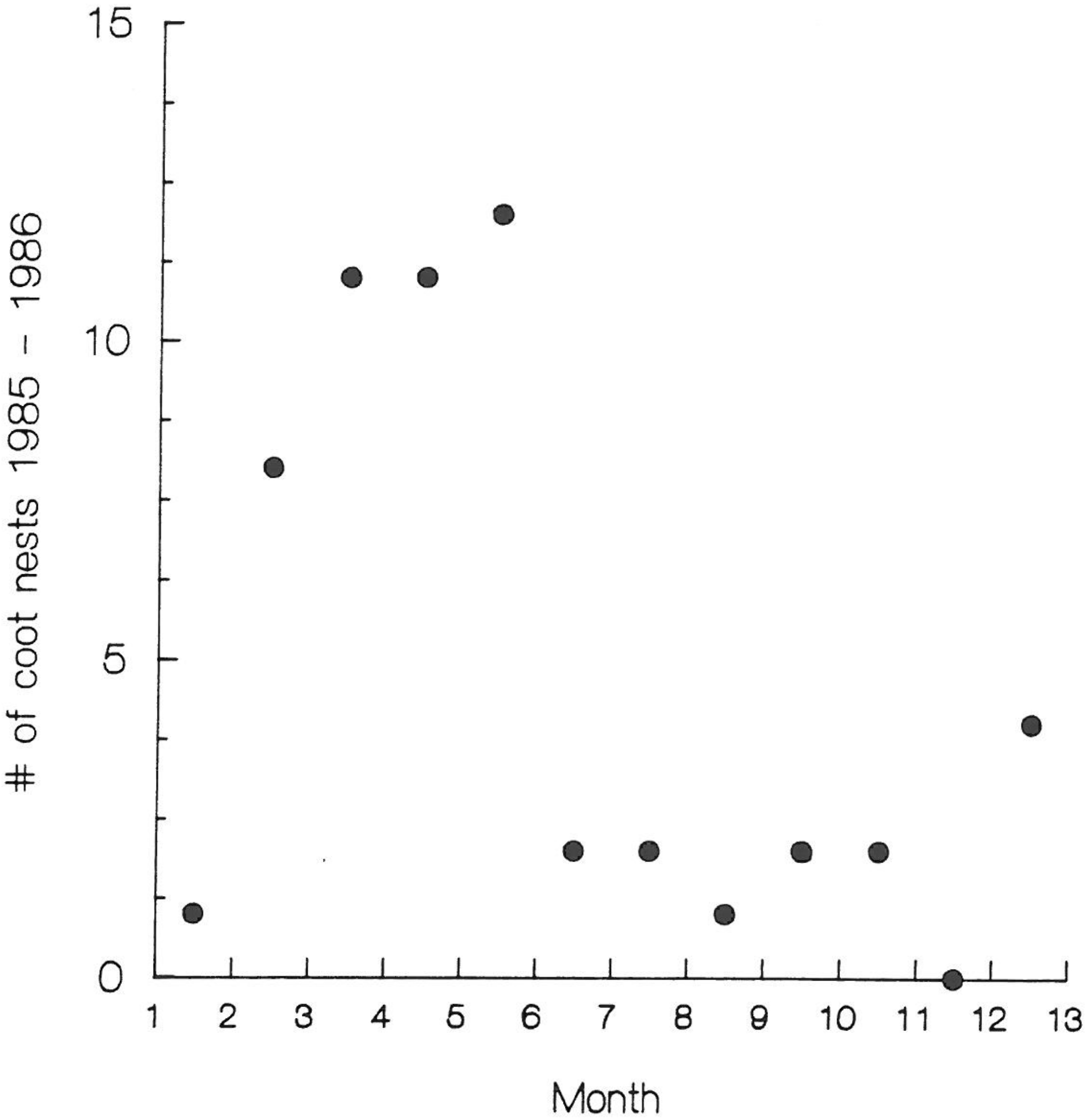

Fig. 4.2. Number of coot nests initiated by month. 1985 - 1986, Kii Unit, James Campbell National Wildlife Refuge, HI.

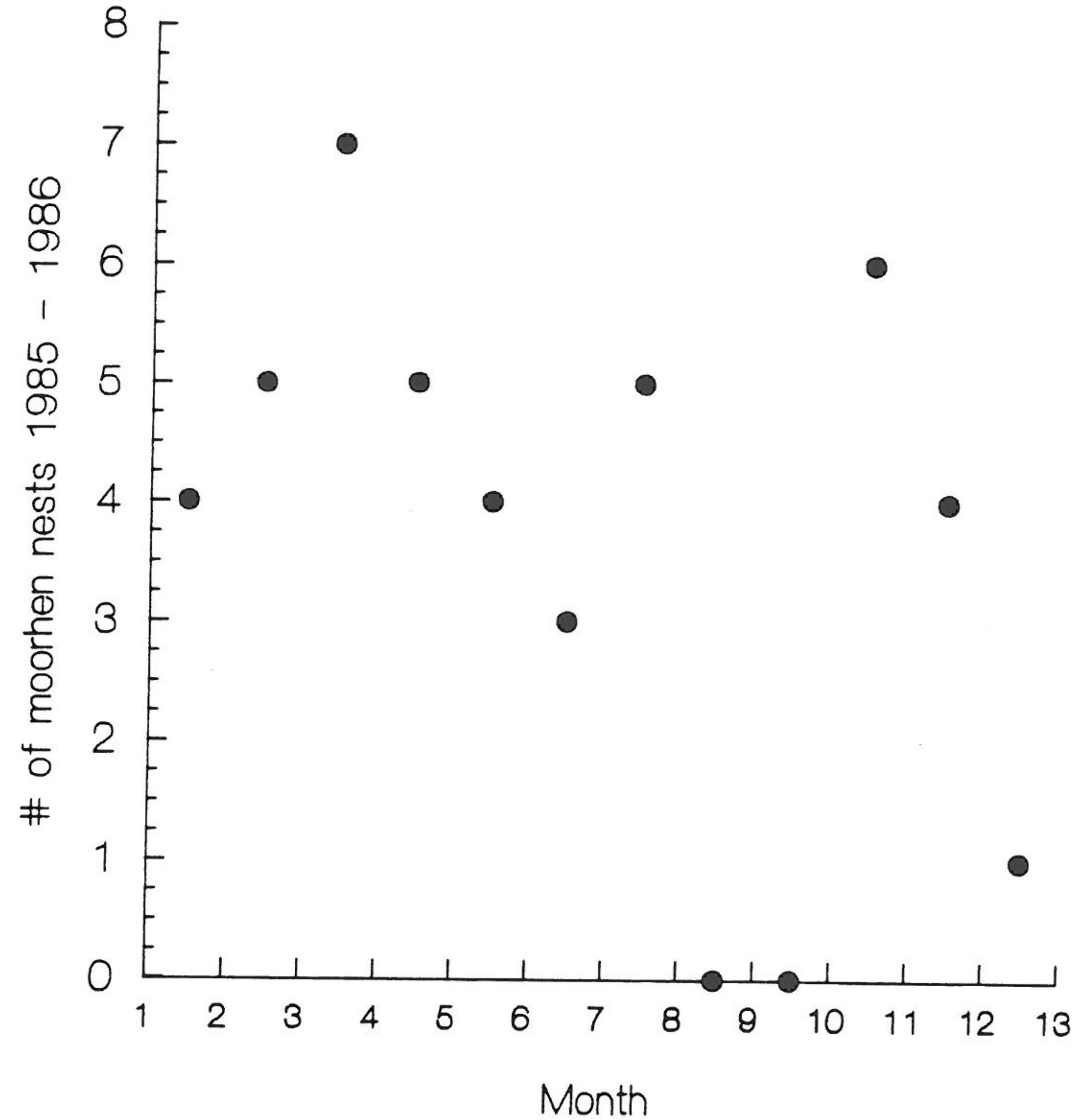

Fig. 4.3. Number of moorhen nests initiated by month. 1985 - 1986, Kii Unit, James Campbell National Wildlife Refuge, HI.

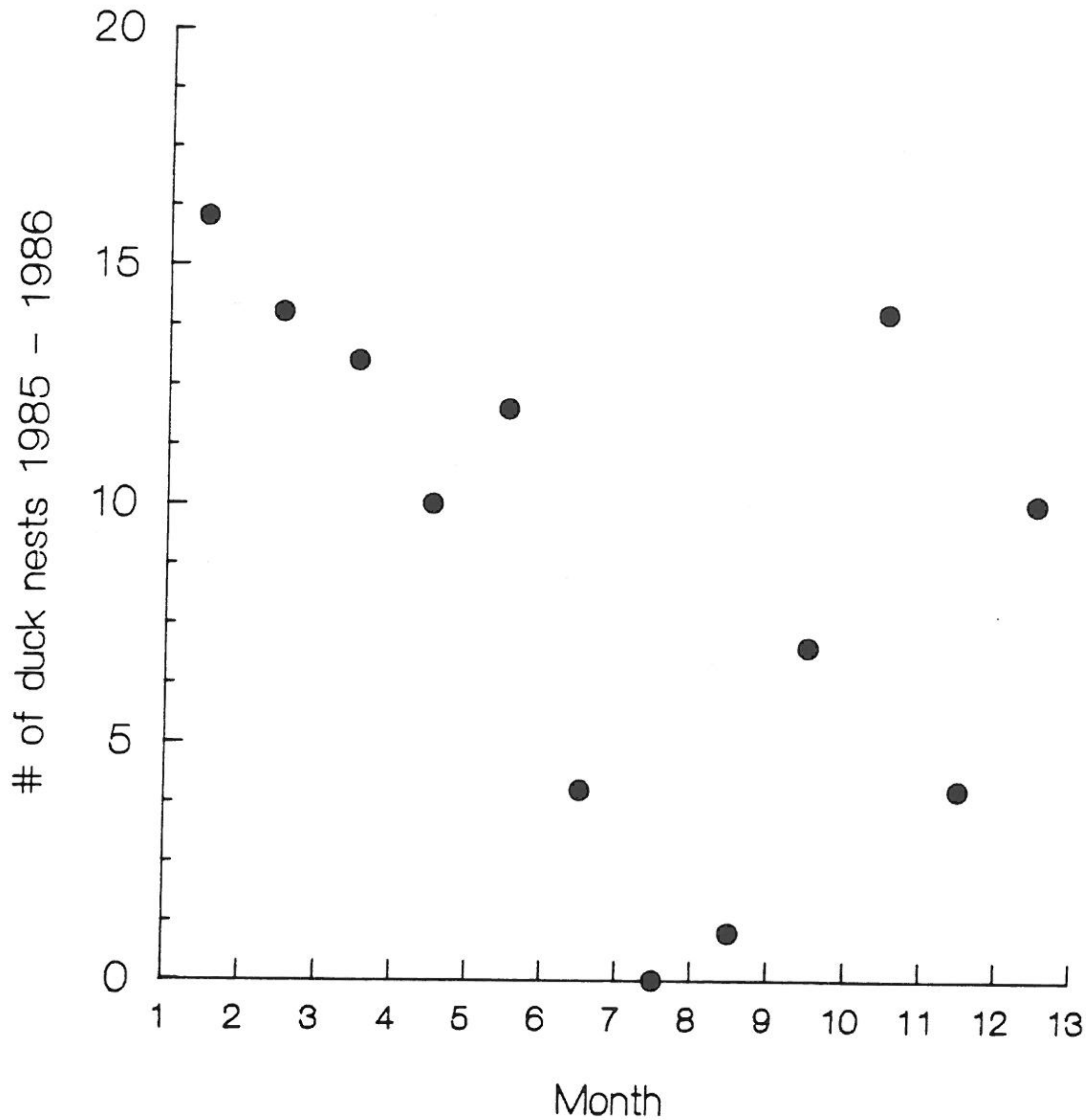

Fig. 4.4. Number of duck nests initiated by month. 1985 - 1986, Kii Unit, James Campbell National Wildlife Refuge, HI.

Table 4.2. Clutch and brood size for Hawaiian stilts (HS), Hawaiian coots (HC), Hawaiian moorhens (HM), Hawaiian ducks (HD), and mallards (MD). 1985 - 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

Species Study	Clutch size Mean	SE	N	Chicks/clutch Mean	SE	N
HS Present	3.4	0.06	243	1.9	0.12	242
Coleman 1981	3.6		73	2		73
HC Present	4.9	0.31	138	3.2	0.22	136
Byrd and Zeillemaker 1981	4.9		33			
HM Present	4.9	0.13	87	3.5	0.24	87
Byrd et al. 1985	5.6		64			
HD Present	7.3	0.16	174	3.5	0.26	174
Schwartz and Schwartz 1953	8.0					
Swedberg 1967	8.4		11			
MD Bellrose 1976	6.8 - 9.6		>100			

Table 4.3. Percent hatching success for Hawaiian stilts (HS), Hawaiian coots (HC), Hawaiian moorhens (HM), Hawaiian ducks (HD). 1985 - 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

Species	Total nests	Total eggs	Total eggs hatched	Hatching success
HS	243	833	453	54
HC	138	678	432	64
HM	87	430	303	70
HD	174	1275	606	47

Table 4.4. Number of nests by fate for Hawaiian stilts (HS), Hawaiian ducks (HD), Hawaiian coots (HC), and Hawaiian moorhens (HM). 1985 - 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

NESTFATE	HS (n=245)	HD (n=179)	HC (n=143)	HM (n=90)
Successful	137	107	96	69
Abandoned	20	28	6	1
Flooded	22	9	7	4
Depredated	61	32	29	16
Unknown	2	1	1	--
Nest fell apart	3	2	4	--

Table 4.5. Causes of egg loss by number of nests for Hawaiian stilts (HS), Hawaiian ducks (HD), Hawaiian coots (HC), and Hawaiian moorhens (HM). 1985 - 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

EGGLOSS	HS (n=146)	HD (n=142)	HC (n=74)	HM (n=34)
Unhatched eggs	46	80	23	4
Died while hatching	1	1	1	--
Collected for study	--	3	--	--
Predator unknown	34	9	19	10
Predator dog	1	21	1	1
Predator mongoose	10	9	9	5
Predator bird	28	2	6	7
Predator rat	5	--	--	--
Flooding	15	9	7	5
Eggs out of nest	3	7	7	2
Nest fell apart	3	1	1	--

Table 4.6. Age at death of Hawaiian stilt (HS), and Hawaiian coot (HC) chicks. 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

Age days	HS # dead (n=24)	HC # dead (n=49)
1 - 10	11	32
11 - 20	6	6
21 - 30	4	5
31 - 40	3	6

Table 4.7. Causes of death for Hawaiian stilts (HS), Hawaiian coots (HC), Hawaiian moorhens (HM), and Hawaiian ducks (HD). 1987 - 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

Species	Cause of death	# Adults	# Chicks
HS	Hatching complications		2
	Unknown (Found in nest)		2
	Predator (Bird Unkn)		1
	Predator (Dog)	1	9
	Unknown	2	12
HC	Unknown (Found in nest)		6
	Predator (Bird Owl)	1	2
	Predator (Bird Unkn)		1
	Predator (Dog)	4	
HM	Unknown (Found in nest)		3
	Predator (Bird)		1
	Predator (Mongoose or bird)		1
	Predator (Dog)	1	
HD	Unknown		1
	Predator (Mongoose)	1	

Table 4.8. Number of fledglings of Hawaiian stilts, Hawaiian coots, and Hawaiian moorhen from brood observations. 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

Species	Total chicks	Total fledged
Hawaiian stilt	37	9
Hawaiian coot	127	35
Hawaiian moorhen	67	28

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APPENDIX

RELATIONSHIP BETWEEN INVERTEBRATE NUMBERS, FISH POPULATIONS, AND WATER SALINITIES

This study segment was originally planned as a complete study. However, logistical constraints and environmental changes prevented me from accomplishing a portion of the original goals. Rotenone applied by Refuge staff in 1987 to eliminate fish, may have effected invertebrate declines for some taxa in 1988 (Davies and Shelton 1983). Further, fish were not completely eliminated in the treated impoundment. Therefore, this chapter is in appendix form, and presents limited data from this study segment.

Diverse populations of macroinvertebrates regularly occur in moist-soil impoundments, and comprise the major or seasonal food source for many wetland birds (Fredrickson and Taylor 1982). Although moist-soil management techniques were developed in temperate areas of the continental U.S., the ecological and management principles of moist-soil management are also of great value to the tropical wetlands of the Hawaiian Islands.

Waterbird research done by the University of Missouri for the U.S. Fish and Wildlife Service (USFWS) from 1985 - 1986 addressed habitat management techniques to enhance desired hydrophytes and macroinvertebrate populations for waterbirds. Sampling of invertebrates associated with a variety of plant species provided important baseline data on the seasonal phenology of invertebrates in relation to different plants. However, most of this sampling occurred prior to the infusion of relatively high salinity waters into the refuge pump ditch in late 1986. This occurred when an adjacent aquafarm operation converted from freshwater to saltwater culture.

Little is known about the response of invertebrates to changes in salinity. Further, fish in refuge impoundments may be competing with waterbirds for invertebrates as a food source. Information on changes in invertebrate numbers with changes in salinity and fish populations will provide for increased management of factors potentially limiting invertebrates for waterbirds. Thus, the objective of this study was to investigate the relationship between invertebrate numbers, fish populations, and water salinities before and after removing fish and implementing a new freshwater source to refuge impoundments.

Study Area

The study took place at the Kii Unit (47ha) of James Campbell National Wildlife Refuge (NWR) located on Oahu's North Shore. The Unit consists of seven impoundments and several ditches (Fig. 2.1). Water is pumped into impoundments from ditches and wells by windmills and pumps. Impoundment water levels are maintained by stop-log control structures. Impoundment water regimes vary between ponds depending on waterbird nesting phenology and vegetation control. However, in general, most impoundments are full from fall through early winter, after which slow drawdowns are initiated to provide increased nesting habitat for waterbirds.

Methods

The study was done in two phases. Phase 1 involved a comparison of invertebrate abundance and diversity between years after a change in salinity took place between 1987 and 1988. Phase 2 involved the effects of fish on invertebrate populations between impoundments G and C in 1988.

I sampled invertebrates in impoundments A, B, C, and G from April - August 1987, and C, G, and F from January - June 1988 using a sweep net and a core sampler. The core sampler was made from a 10.16 cm wide by 8.89 cm long piece of PVC pipe with window screen (7 squares per cm) taped over the top. The sampler's bottom edge was sharpened to facilitate sampling. Core samples were placed in standard #30 mesh screen and hand sorted in the field. I used a modified sweep net (Usinger 1956) with a Nitex insert (Reid 1983) and sampled a distance of 1 m through the water column.

Five core samples in water hyssop (_Bacopa monnieri_), and five nectonic sweep samples through knottgrass (_Paspalum distichum_) were taken in each impoundment once a month during 1987. In 1988, five core samples were taken biweekly in undisced, disced (impoundment G and C), and water hyssop substrates (impoundment C and F). In addition, five sweep samples were taken biweekly in open water and knottgrass areas (impoundment G and C). Invertebrates were identified and counted in the field according to taxa. I recorded salinity with a refractometer (ppt) during each sampling period.

USFWS personnel dewatered impoundments G and C during August and September 1987. Rotenone was applied to remaining puddles and wet edges of impoundment G to remove fish. Fish were allowed to survive in puddles located in impoundment C. Portions of both impoundments were disced while drained to improve nutrient cycling capabilities of impoundment soil when reflooding occurred.

Impoundment C was reflooded in October 1987 by pumping water directly into it and allowing additional fish to enter. Impoundment G was reflooded by slow percolation through dikes of adjacent impoundments to prevent fish entry.

Two baited minnow traps were set in each impoundment to monitor for presence of mosquito fish (_Gambusia_ sp.) and Tilapia during 1988. Dry cat food was contained in a piece of window screen and suspended within traps using monofilament line. The traps were anchored to the impoundment bottom by stones. Traps were set every two weeks for 48 hours beginning January 1988. This frequency of fish sampling continued until equal numbers of fish were caught in each impoundment for three consecutive trapping periods. As in impoundment G, the southern half of impoundment F lacked fish. However, a filamentous algae bloom in F prevented representative invertebrate sampling except for water hyssop cores. I continued all types of invertebrate sampling in impoundments C, G, and water hyssop core sampling in impoundment F.

Results

Salinity

Mean salinity levels for all impoundments combined varied little within years but differed dramatically between years (Fig. 5.1). Mean salinity levels in 1987 (mean = 16.9) decreased by 70% in 1988 (mean = 6.2). Salinity levels within impoundments varied little within years except for impoundment E which received direct input from saline ditches and was influenced by ocean surges (Tables 5.1, 5.2).

Fish

By early March, the fish sampling scheme was abandoned because more mosquito fish were caught in impoundment G (fishless), than impoundment C. Although mosquito fish were present in minnow traps shortly after trapping began, Tilapia were never caught in traps or by other sampling methods used in the study. However, I saw Tilapia regularly in open water areas of impoundment C. Conversely, Tilapia were not seen in impoundment G.

Water Hyssop Cores (1987 vs. 1988)

I observed differences in invertebrate diversity and numbers in water hyssop core samples between 1987 and 1988 (Table 5.3). Despite the reduction in salinity levels in 1988, diversity of invertebrate families per core was higher in 1987 (mean = 2.3 families/core) than in 1988 (mean = 1.6 families/core). Further, mean number of invertebrates in core samples of water hyssop also declined significantly in 1988 ($P < 0.01$). Lymnidae snails declined most dramatically from 66 to 0.4 snails per core. Hydrophillids and Dytiscid beetles also decreased in 1988 while Tubificids and Stratiomyids disappeared from core samples completely. In contrast, Chironomids were absent in 1987 but were abundant in 1988.

Knottgrass Sweeps (1987 vs. 1988)

Although invertebrate diversity in knottgrass sweep samples was higher, total numbers of invertebrates per sweep sample dropped by 85% from 1987 to 1988 (Table 5.4). As in core samples, Lymnidae snails decreased dramatically in 1988. Tubificids and Stratiomyids disappeared completely. In contrast, both Chironomid and Corixid numbers increased per sweep sample in 1988 from zero individuals in 1987 to 26 and 16, respectively.

Disced and Undisced Cores (1988 G vs. C)

In 1988, invertebrate numbers in disced core samples were higher than undisced samples in impoundments G and C; however, there were no differences in invertebrate diversity (Tables 5.5 and 5.6). Further, impoundment G had higher invertebrate numbers per sample than impoundment C in both disced and undisced substrates. Numbers of chironomids were similar in disced versus undisced core samples.

Open Water Sweeps (1988 G vs. C)

There were notable differences in invertebrate populations in open water sweep samples between impoundments G and C. Diversity of invertebrate families per sample was only slightly lower in impoundment C than in G. Numbers of individuals per sweep sample were higher in impoundment G (Table 5.7). Corixids and

Chironomids were much more abundant in impoundment G. Although there were relatively low numbers of Lymnidae snails, damselflies, and mosquito fish in impoundment C, these taxa were completely absent from open water sweeps in impoundment G.

Knottgrass Sweeps (1988 G vs. C)

Diversity of invertebrates in knottgrass sweep samples was only slightly greater in impoundment G than in C. However, numbers of invertebrates, specifically Chironomids, Corixids, and Lymnidae snails in impoundment G were at least twice that of C (Table 5.8). Hydrophillid and dragonfly numbers per core were only slightly higher in G than in C. Conversely, damselfly and Amphipod numbers per core in C were greater than numbers found in G. Equal numbers of mosquito fish were caught in impoundment G and C.

Discussion

Differences between years and impoundments in invertebrate numbers and diversity probably resulted from a variety of three factors, including changes in salinity, fish levels, and recolonization rates of invertebrates. The absence in 1987 and presence in 1988 of certain invertebrate families was due to a decrease in salinity, especially Chironomids in all types of samples, and Corixids in sweep samples. Further, decreases of

Lymnaea numbers in all samples between 1987 and 1988 was probably a function of fish presence, poison, and slow recolonization rates.

Salinity

The salinity decrease in 1988 was a result of installing a freshwater well that pumped water directly into impoundments B, C, F, and G. Prior to late 1987, existing windmills pumped high saline water into impoundments from ditches. High salinity variation in impoundment E during 1988 was a result of it receiving direct input of high saline water from ditches and the ocean. Small variation in salinity levels in impoundments in 1988 were caused by breakdown of the freshwater windmill pumping and periodic use of other windmills pumping saline water from ditches to maintain water levels in impoundments.

Fish

Rotenone application in impoundment G was effective in controlling Tilapia but failed to eliminate mosquito fish adequately. This could reflect difficulty in Rotenone application amongst thick vegetation in remaining puddles; a potential refuge for small fish.

Water Hyssop Core Samples (1987 vs. 1988)

Species diversity and numbers of invertebrates per water hyssop core sample decreased in the two sample years. The slow recolonization rate after Rotenone application and impoundment draw-down were the main reasons for decreased invertebrate numbers in benthic core samples. Lymnidae have the ability to aestivate over short dry periods as long as soil is muddy (Pennak 1978). However, Rotenone was applied to impoundment G when nearly dry, which eliminated invertebrates and Tilapia. Impoundment F was thoroughly dried before reflooding, and C was drawn down until only puddles remained. These factors probably decreased potential for rapid recolonization by certain invertebrate taxa (i.e. Lymnaea and Tubificids).

Knottgrass Sweeps (1987 vs. 1988)

Knottgrass sweep samples were characterized by higher invertebrate species diversity in 1988 than in 1987 but with less invertebrate numbers recovered per sweep sample. This difference reflects the ability of only a few species to tolerate higher saline conditions and, as a result, these species increase in numbers due to a lack of interspecies competition.

Disced and Undisced Cores (1988 G vs. C)

Disced substrates contained the highest invertebrate numbers when compared to undisced substrates. Discing provides loose soil, which is a more suitable substrate for benthic invertebrates. Further, the higher invertebrate numbers in both disced and undisced core samples in impoundment G compared to C reflects absence of Tilapia in G.

Open Water Sweeps (1988 G vs. C)

Invertebrates in the water-column, especially Corixidae and Chironomidae, responded quickly to reflooding. This rapid colonization could be a function of decreased salinity and absence of predators (Tilapia) removed by Rotenone. Rotenone application was successful at eliminating Tilapia and I believe this allowed for higher numbers of invertebrates per sweep sample in impoundment G.

Knottgrass Sweeps (1988 G vs. C)

Sweep samples through knottgrass in impoundments G and C showed differences per sample in invertebrate numbers. Higher numbers of invertebrates, Chironomids, Corixids, and Lymnids in samples from impoundment G could reflect absence of fish, specifically Tilapia.

Although my data reflect important differences between years, impoundments, and taxa, with changes in salinity, soil texture, and fish presence, it is still not known upon which invertebrates waterbirds feed. Despite this, resource managers need to be aware of the effects of their water manipulations and wetland management on invertebrates in general.

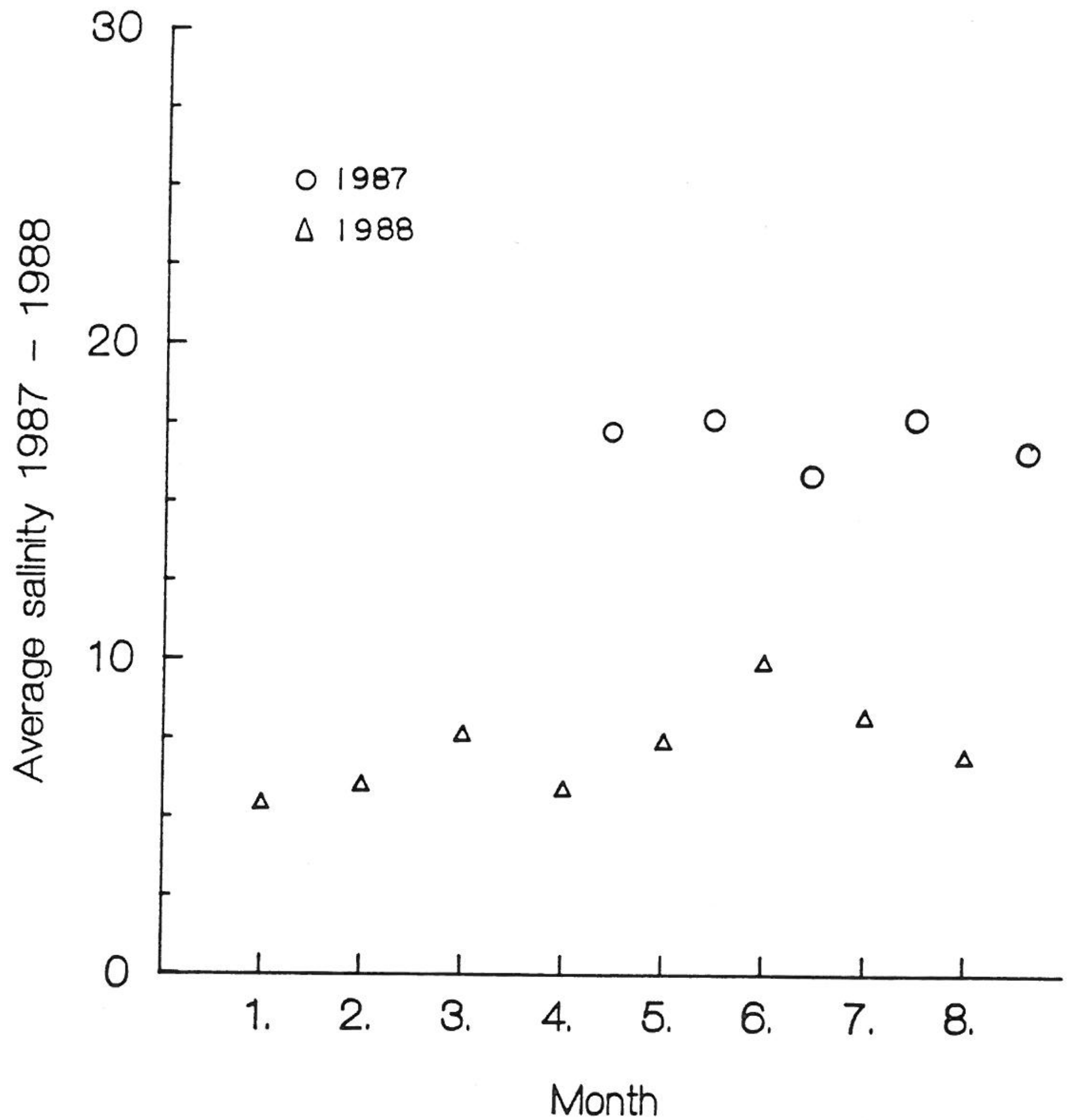

Fig. 5.1. Mean salinity levels for all impoundments combined. 1987 - 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

Table 5.1. Average monthly salinity by impoundment. 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

	Month							
Impoundment	1	2	3	4	5	6	7	8
A	5.0	6.2	7.2	5.0	4.2	4.7	11.7	8.0
B	4.5	7.5	4.7	1.7	2.2	2.7	1.2	0.0
C	4.5	3.0	3.0	2.7	6.2	6.3	4.0	2.0
D	6.5	7.5	9.7	10.0	10.2	11.3	9.0	5.0
E	5.3	7.5	16.2	9.2	16.8	29.0	29.7	26.0
F	3.7	2.5	2.0	2.0	1.4	2.0	1.2	2.0
G	8.0	7.5	10.0	10.0	10.4	10.3	6.2	5.0

Table 5.2. Average monthly salinity by impoundment. 1987, Kii Unit, James Campbell National Wildlife Refuge, HI.

	Month				
Impoundment	4	5	6	7	8
A	16.0		15.0	18.0	19.0
B		15.0	13.5	15.0	16.0
C	18.0	18.0	17.0	16.5	14.0
G		20.0	18.0	20.5	16.0

Table 5.3. Core sample contents taken from water hyssop. 1987 - 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

	1987 (n=85)		1988 (n=80)	
	Mean	SE	Mean	SE
Families	2.3	0.12	1.6	0.12
Number	71.3	5.93	28.6	3.22
Lymnidae	66.7	5.22	0.4	0.10
Hydrophillid	0.9	0.19	0.03	0.03
Dytiscid	0.4	0.08	0.01	0.01
Amphipod	1.3	0.50	1.1	0.45
Damselfly	0.4	0.10	0.2	0.06
Fish	0.2	0.06	0.03	0.03
Dragonfly	0.01	0.01	--	--
Chironomid	--	--	24.2	3.21
Tubificid	0.50	0.13	--	--
Stratiomyid	0.01	0.01	--	--

Table 5.4. Sweep sample contents taken in knottgrass. 1987 - 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

	1987 (n=85)		1988 (n=80)	
	Mean	SE	Mean	SE
Families	2.8	0.12	4.3	0.18
Number	553.4	63.15	83.8	11.44
Lymnidae	537.3	62.76	12.1	5.07
Damselfly	2.0	0.29	17.1	2.15
Amphipod	6.8	1.96	3.7	1.19
Hydrophillid	1.7	0.57	2.1	0.29
Dytiscid	0.5	0.12	0.4	0.09
Dragonfly	0.02	0.02	2.4	0.41
Chironomid	--	--	26.2	6.83
Corixid	--	--	16.5	7.32
Tubificid	0.01	0.01	--	--
Stratiomyid	0.10	0.05	--	--

Table 5.5. Disced core sample contents taken from impoundments G and C. 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

	G (n=40)		C (n=40)	
	Mean	SE	Mean	SE
Families	1.1	0.05	1.0	0.03
Number	134.9	14.12	44.6	4.56
Chironomid	104.4	14.14	39.9	5.04
Amphipod	0.5	0.25	---	---
Corixid	---	---	0.8	0.32

Table 5.6. Undisced core sample contents taken from impoundments G and C. 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

	G (n=35)		C (n=41)	
	Mean	SE	Mean	SE
Families	1.1	0.06	1.0	0.00
Number	104.0	11.56	39.2	4.73
Chironomid	91.1	12.92	35.7	5.13
Amphipod	0.4	0.23	0.1	0.08

Table 5.8. Sample contents of knottgrass sweeps within impoundments G and C. 1988, Kii Unit, James Campbell National Wildlife Refuge, HI.

	G (n=40)		C (n=40)	
	Mean	SE	Mean	SE
Families	4.5	0.26	4.2	0.25
Number	113.0	20.87	54.6	7.12
Damselfly	14.2	2.80	20.0	3.23
Chironomid	38.9	13.20	13.5	2.57
Fish	4.8	0.87	4.8	1.82
Hydrophillid	2.3	0.43	1.8	0.40
Dragonfly	3.0	0.73	1.9	0.38
Corixid	32.6	14.27	0.5	0.24
Lymnidae	18.9	9.37	5.3	3.75
Dytiscid	0.5	0.11	0.4	0.15
Amphipod	2.7	1.55	4.8	1.83

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