

Rhinoceros auklets (*Cerorhinca monocerata*) and

Columbia black-tailed deer

(*Odocoileus hemionus columbianus*),

Managing a Wildlife Dilemma on Protection Island

David B. Falzetti

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Managing a Wildlife Dilemma on

Protection Island

By

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ABSTRACT

Rhinoceros auklets (*Cerorhinca monocerata*) and
Columbia black-tailed deer (*Odocoileus hemionus columbianus*),

Managing a Wildlife Dilemma on Protection Island

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Protection Island National Wildlife Refuge and associated Zella M. Schultz Seabird Sanctuary is vital seabird habitat and a nationally significant environmental resource. The Island supports more than 70% of the nesting seabirds in Puget Sound including; one of the last inland tufted puffin (*Fratercula cirrhata*) colonies; the largest glaucous-winged gull (*Larus glaucescens*) colony in Washington State; and one of North America's largest and most important rhinoceros auklet colonies. Two or three Columbia black-tailed deer, previously unknown on Protection Island, colonized in the early 1990's, probably by swimming some 2.25 km from the mainland. Non-native mammals are a primary cause of extinctions and ecosystem changes on islands around the world and a major threat to seabird populations. In 2010, Refuge managers elected to remove the deer based on concern for seabirds. This thesis explores information regarding ungulate impacts on biodiversity, particularly deer impacts in various habitats including San Juan and Gulf Island archipelagos and the Queen Charlotte Islands with the intention of characterizing potential deer impacts on Protection Island. Moreover, I wanted to inform the question of whether deer removal is warranted and satisfy the public's trust by incorporating the best available science in a clear rationale for the deer policy. In early 2011, the Island's deer density of 53 deer/km² (78 individuals) was very high compared to other habitats within their Pacific Northwest range. Deer impacts at high densities resemble those of domestic sheep and can significantly reduce island species diversity and alter vegetation structure and function. Anecdotal information and recent research shows deer on the Island damage burrows and facilitate erosion in auklet habitat, and likely reduce gull reproductive success. Furthermore, deer inhibit vegetation recovery from extensive domestic grazing (1874 to 1968) and can jeopardize future restorations. My findings validate concerns regarding deer impacts and support the removal decision.

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I. Introduction

A Seabird Sanctuary

A little over two kilometers off the northern shore of Washington's Olympic Peninsula in the Strait of Juan de Fuca lies a small steep sided island crowned with a picturesque plateau of gently rolling hills. Just shy of 1.5 km² (364 acres), this inland oasis is known as Protection Island (hereafter PI) and is critical nesting habitat for more than 70% of the seabirds in Puget Sound (USFWS 2002). In addition to being home to one of the Sound's last tufted puffin (*Fratercula cirrhata*) colonies (WDFW 2011b, USFWS 2010b) and home to the largest glaucous-winged gull (*Larus glaucescens*) colony in Washington State (PTMSC 2011), it is also home to one of North America's largest and most important breeding colonies of rhinoceros auklets (*Cerorhinca monocerata*), (hereafter RHAU), (Pearson et al. 2009). Protection Island is vital seabird habitat and a nationally significant environmental resource (USFWS 2010b).

Because RHAU breed in the Salish Sea, unlike many other species of seabirds that simply migrate through or overwinter, and because RHAU are a top-level piscivorous predator, they are particularly vulnerable to fluctuations in local conditions such as forage fish population declines (WDFW 2011c). Their relationship to the local environment combined with an extensive body of research on the population dating back to the 1950's (Richardson 1961, Wilson 1977, 1986, 1993, 2005, Thompson et al. 1985, Wilson and Manuwal 1986, Wahl and Speich 1994, Pearson et al. 2009, 2010), make PI's RHAU colony size a good candidate for measuring trends in the health of the Salish Sea (WDFW 2011c).

In 1975, the State of Washington recognized PI's importance to seabirds and established the Zella M. Schultz Seabird Sanctuary (hereafter Sanctuary) on the Island's southwestern tip (JC 1974, USFWS 1990). In 1982, the U.S. Congress followed suit with the remaining land and established the Protection Island National Wildlife Refuge (hereafter Refuge). Today the entire Island is co-

managed by the Washington Department of Fish and Wildlife (hereafter DFW) and the U.S. Fish and Wildlife Service (hereafter FWS) as a protected seabird nesting refuge and marine mammal reserve. In addition to bald eagles (*Haliaeetus leucocephalus*), great horned owls (*Bubo virginianus*) and numerous other terrestrial avians, six species of seabirds: rhinoceros auklets, tufted puffins, pigeon guillemots (*Cephus columba*), pelagic cormorants (*Phalacrocorax pelagicus*), double-crested cormorants (*Phalacrocorax auritus*), and glaucous-winged gulls, typically nest on the Island (USFWS 2010b).

Deer Colonize Protection Island

In 1991, two or three Columbia black-tailed deer (*Odocoileus hemionus columbianus*) (hereafter deer) colonized PI (USFWS 1991, 1992, Cowles and Hayward 2008) probably by swimming from the mainland which is just 2.25 km to the south as they are good swimmers (IHEA 2011). Since their arrival deer have been observed feeding and resting within the RHAU colony (Pearson and Hodum pers. corr. 2009) leading to concerns that a growing deer herd may be negatively impacting the Island's RHAU colony (USFWS 2010b). Despite being native to Washington State, deer can be considered non-native to PI based on the fact that there are no historical accounts of their presence on the Island dating back to 1792 (Vancouver 1798, Richardson 1961, Larsen 1982, Carson 1983, Clark 1995, USFWS 2010b).

Although PI lacks a readily available source of freshwater, Refuge staff provided the new arrivals with troughs of fresh well water which may have inadvertently facilitated colonization. Due to a lack of competitors, predators, and other controls such as hunting, the deer herd burgeoned. By early 2011, the number of deer on PI had swelled to no less than 78 animals (Falzetti pers. corr. 2011). Judging by the number of fawns observed (15) in fall/winter of 2009-2010 and number of deceased deer observed (13) during roughly the same time period (Davis pers. corr. 2010), the herd may not yet be at carrying capacity defined as the density at which mortality is equal to recruitment (McCullough 1984).

According to counts conducted in February of 2010 and March of 2011, deer density on PI is very high, 48 / km² (71 deer) and 53 deer / km² (78 deer) respectively (Davis pers. corr. 2010, Falzetti pers. corr. 2011). By comparison, black-tailed deer densities along Washington's Columbia River typically range from 4 to 12 deer / km² (USFWS 2010b) and Columbia black-tailed deer densities on the north Olympic Peninsula in a 1997 study ranged from 1.14 to 9.99 deer / km² (Ratti et al. 1999). Recent density estimates of Sitka black-tailed deer (*Odocoileus hemionus sitkensis*), which are closely related to Columbia black-tailed deer, ranged from 7 to 12 / km² in the temporal coastal rainforest of southeast Alaska (Brinkman et al. 2011).

Understanding and Mitigating Impacts of Deer

Non-native mammals are a primary cause of extinctions and ecosystem changes on islands around the world and a major threat to seabird populations (Donlan and Heneman 2007). In 2005, the California Current Marine Bird Conservation Plan identified habitat modifications such as changes to soils and vegetation caused by introduced mammalian herbivores as a serious threat to seabirds along the Pacific coast (Mills et al. 2005). For the most part, a small group of species including rats, rabbits, pigs, goats, and sheep are responsible for much of the damage to invaded insular ecosystems where indigenous species have evolved in their absence (Kenyon 1964, Lloyd et al. 1975, Laughrin et al 1994, McChesney and Tershy 1998, Ebbert and Byrd 2000, Courchamp et al. 2003, Campbell and Donlan 2004). However, there is growing evidence that ungulate invaders such as deer are also negatively impacting biological diversity on many islands (Gaston et al. 2008, Martin et al. 2011).

The relationship between deer and RHAU on PI is not well understood as there has been little deer research conducted on the Island (Hayward and Henson 2008). However, FWS managers believe that deer are damaging seabird burrows and subsequently increasing RHAU mortality (USFWS 2010b) and that deer could be impacting other species. In 2010, the FWS completed a 15 year comprehensive

conservation management plan (hereafter CCP) for the Refuge which includes removing all deer from PI in order to eliminate their impacts on seabird habitat (USFWS 2010b). The decision was based on a variety of observational evidence that suggests deer have compromised burrows, increased erosion, and altered the Island's vegetation (Cowles and Hayward 2008, Hodum and Pearson pers. corr. 2009). Recent research has begun to quantify RHAU burrow damage from deer (Balbag and Hodum unpublished) lending support to the decision outlined in the CCP.

Prior to the Refuge's establishment, PI suffered extensive damage due to development and overgrazing by domestic sheep and cattle from which it has not fully recovered (Richardson 1961, Larsen 1982, Cowles and Hayward 2008). Although development has been halted and the domestic ungulates are long gone, deer may be slowing the Island's recovery from years of high grazing pressure. Conservation efforts elsewhere have increasingly focused on removing invading ungulates as an effective strategy in restoring island native plant and/or animal communities (Ebbert and Byrd 2000, Donlan and Heneman 2007, Gaston et al. 2008).

Justification for the Deer Removal Decision

In developing their justification for deer removal, FWS relied on the Refuge's enabling legislation which states that the Island was set aside as a sanctuary and research facility for seabirds and marine mammals. There is no mention of deer (USFWS 1985). In fact, the original Refuge proposal presented to the public, including the property owners who were being asked to divest of their interests, specifically mentioned keeping the Island free of "conflicting animal usage" (Larsen 1982, pg. 13).

The FWS also considered that the purpose of the associated Zella M. Schultz Seabird Sanctuary managed by DFW is "... for the preservation of unique and endangered species of wildlife . . . including in particular the Rhinoceros Auklet

breeding colony. It is further intended that scientific, educational and other compatible uses may be undertaken on the property.” (JC 1974). The purpose of the Sanctuary includes “compatible uses” language and preserving “unique and endangered species” which seemingly precludes deer.

The FWS concluded that deer have a high potential to negatively impact Refuge biodiversity in general, and seabirds in particular, especially RHAU, despite the lack of strong empirical evidence demonstrating that conclusion at the time of their decision. Furthermore, FWS noted that the deer population in Western Washington was robust (WDFW 2006) and therefore conserving the PI herd was inconsequential to the overall regional population. Lastly, FWS considered the fact that there were no other large RHAU colonies located in the inland waters of the Pacific Northwest where the species is buffered from severe ocean weather and fluctuating climate conditions including El Niño events.

During the draft CCP’s public comment period FWS was criticized for their decision to remove the deer herd without a complete, or at least a better understanding of deer impacts on the Refuge (USFWS 2010a). Because FWS is a public agency charged with managing public lands it must justify management decisions, especially where there is a lack of definitive research guiding such actions. There was an obvious concern from those that commented that decisions affecting wildlife resources are made without a complete understanding of the implications of those choices including the potential for unintended consequences.

This thesis explores information on ungulate impacts in mainland habitats and on island biodiversity with the intention of characterizing the potential for deer impacts on PI. Specifically, I was interested in how deer might affect breeding RHAU. Through review and analysis, I hope to inform the question of whether complete deer removal is warranted and to address a controversial Refuge management planning issue in the absence of complete scientific information. I conducted an exhaustive literature search on the impacts of ungulates on island species including seabirds and considered recent unpublished research on deer

damage to RHAU burrows on PI and the personal observations and accounts of Island residents, visiting scientists and staff. I then synthesized this information in an effort to provide a comprehensive account of the impacts of deer on PI. My intention is to document and discuss what is known and thereby begin to illuminate information gaps in the complex relationship between deer and RHAU.

Beyond Protection Island

Conservation biology as it informs public lands management is not an exact science. Rather, it includes an evolving experience-based practice of defining issues and goals, evaluating available evidence, formulating logical responses, and predicting and measuring outcomes. Using all available science in developing management actions and thoroughly assessing outcomes will maximize our ability to inform critical information gaps and develop better management strategies through time. Conservation biology functions within an adaptive management paradigm that evolves according to best available science and gives consideration to returns on investment.

In the year 2000, the U.S. Government spent \$635 million on invasive species (Donlan and Heneman 2007). Endeavoring to understand each unique situation in an effort to make the best management decisions is a critical element in satisfying the public's trust and in maximizing returns on current and future investments. Moreover, utilizing an adaptive management approach on islands provides unique cost effective opportunities to develop and assess advanced conservation strategies in somewhat isolated environments which can provide foundations for larger and more complex restoration efforts on our public lands.

II. Protection Island – The Study Area

Figure II.1, Protection Island, Washington



(USFWS)

Protection Island History

In 1592, Greek mariner Apostolos Valerianos sailing for Spain under the name Juan de Fuca claimed he may have discovered the entrance to the fabled Strait of Anián, the northwest passage connecting the Pacific Ocean with the Gulf of Saint Lawrence (Enchanted Learning 2011). While exploring the west coast of North America 200 years later, Captain George Vancouver found a large passage in an otherwise rugged and unbroken coastline that resembled de Fuca’s account although it was much further north than de Fuca had placed it. While sailing deep into this inlet which he called the “supposed strait of Fuca”, Vancouver encountered a small unique island protecting the mouth of a long open bay. Vancouver’s description of the island’s landscape was; “. . . almost as enchantingly beautiful as the most elegantly finished pleasure grounds in Europe”. He immediately recognized the bay’s potential as a strategic military

harbor and named it in honor of his ship Discovery. He also recognized that the formidable sentinel standing guard at its entrance would make an ideal position for harbor defenses and appropriately dedicated it Protection Island (hereafter PI) (Vancouver 1798).

Encompassing 1.47 km² (364 acres), PI is located just 2.25 km north of Diamond Point and approximately 13 km west of Port Townsend in Jefferson County, Washington. It is a crescent-shaped island flanked by low sandy spits pointing east and west. Both spits, Kanem to the west and Violet to the east, do not exceed 12 m of elevation. The Island was shaped primarily by glacial forces during the Pleistocene epoch (USFWS 2010b) and is comprised of a high central plateau with low undulating hills flanked by cliffs averaging approximately 45 m high (maximum 62 m (Larsen 1982)) along the northern shore and steep bluffs upwards of 30 m high along the southern shore. The high plateau accounts for approximately 70% of the Island while the spits, bluffs and cliffs comprise the remainder. About 80% of the high plateau is grassland and the rest is mixed coniferous forest (USFWS 2010b). Lying in the rain-shadow of the Olympic Mountains, PI typically receives around 40 cm of rain annually (CTI 2011) making it one of the driest places in Western Washington (OSU 2006).

Farming, Grazing, and Development

Protection Island has a colorful history that includes over 100 years of farming and livestock grazing, more than 70 years of avian research, a stint as a World War II artillery battery including a bombing range, at least two catastrophic fires in the 1940's and 50's, and a failed proposal to turn the Island into a leper colony (Clark 1995). In 1968, the Island was purchased for \$435,000 by the newly created Protection Island Company and in 1969 Jefferson County approved subdividing 4 of 5 platted units into 831 small lots averaging 9,000 square feet (Larsen 1982). The full plan intended 1,098 lots (Wilson 1977, USFWS 1992).

An intensive period of development for a summer home resort followed. During

the late 1960's and 1970's developers created an active airstrip, installed extensive infrastructure, dredged a marina destroying the wetland at the base of Violet Spit, and aggressively marketed the project (USFWS 1987). By the early 1980's there were 14 houses and 27 house trailers on PI (Hirsch 1981) and at least 528 separate property owners (USFWS 1987) had a stake in the Island's future.

The Fight to Save the Birds

Despite extensive habitat alteration by humans, the Island continued to be an important breeding site for seabirds and the focus of concerted efforts by conservationists to protect nesting colonies (Palmer 2000). At the same time, supplying enough fresh water to satisfy the growing human population was an ongoing problem for developers. The Island's only well which was not certified for human consumption was woefully inadequate. In 1975, Jefferson County enacted a building moratorium on PI because the developers lacked a reliable freshwater supply and abruptly halted issuing building permits (Clark 1995). In 1980, a 100 m well was drilled but no fresh groundwater was encountered (Larsen 1982).

In 1972, the 5th un-platted unit measuring roughly 0.2 km² (48 acres) or about 13% of PI was purchased by The Nature Conservancy (hereafter TNC) with the help of the Interagency Committee for Outdoor Education (Hirsch 1981, USFWS 1992). In 1974, the TNC parcel including the marine mammal haul out areas on Kanem Spit and the western bluffs which contained about 30% of the seabird nests on the Island at the time (Larsen 1982), was turned over to the Washington Department of Game (hereafter WDG) "for the consideration of \$252,570.00" (JC 1974). WDG dedicated it as the Zella M. Schultz Seabird Sanctuary on May 30, 1975 (Hirsch 1981). It was the agency's first non-game wildlife preserve (Palmer 2000). The Sanctuary was named in honor of a local seabird biologist and one of the Island's greatest wildlife advocates. For many years Zella Schultz aggressively fought to save PI from development but sadly passed away from a childhood disease the very month TNC purchased the west end of the Island

(Palmer 2000).

On October 15, 1982, President Ronald Reagan followed suit and signed the Protection Island National Wildlife Refuge Act proclaiming the remaining acreage a National Wildlife Refuge and giving FWS authority to purchase the land. It was the only federal wildlife refuge established during his administration (Palmer 2000). By July of 1985, just 195 of the original 831 lots had been acquired by FWS from willing sellers (Thompson et al. 1985). On April 11, 1986, FWS filed a “Declaration of Taking” with the U.S. District Court for the remaining PI properties. An expensive and lengthy legal battle ensued (USFWS 1987).

In March of 1987, a U.S. Magistrate Judge signed an “Order for Possession” and gave lot owners 60 days “notice to vacate” thereby condemning their lands. He also set a trial date for early June. At issue was compensation as property owners were demanding ten times the assessed values. FWS spent approximately \$404,000 on independent engineering and assessment. The result was that all lots were appraised at \$2,000, although most had originally been sold for 3 – 6 times that amount when they were part of the summer home resort development. Assessors concluded their worth was substantially diminished because without water they could only be used for camping. The trial lasted three weeks and included a day-long trip to PI for the judge, jury, and lawyers for both parties. In the end property owners were awarded 3 – 5 times the appraised value (USFWS 1987).

Two months after the trial FWS moved two volunteer caretakers onto the island and in December 1987, established a Refuge Manager position. On August 26th, 1988, the Protection Island National Wildlife Refuge was officially dedicated (USFWS 1992). In the end, all but 1 of the 528 property owners surrendered their rights to live on PI although most were justly compensated and FWS promised to forever preserve the Island for the study and advancement of the Island’s endemic wildlife, especially seabirds.

Today the Island is co-managed as a protected seabird nesting site and marine mammal reserve by the Washington Department of Fish and Wildlife (formerly the WDG) and the U.S. Fish and Wildlife Service. It is without a doubt the single most important RHAU colony in Washington State (USFWS 2010b) and remains one of the most important, and probably the most expensive seabird island in the Salish Sea. As mentioned previously, the State of Washington spent \$252,570 to acquire the Zella M. Schultz Seabird Sanctuary. FWS spent more than \$4 million to purchase the rest of the Island including about \$404,000 to assess the property and \$3,624,095 to buy lots. Approximately 1.42 acres were donated to the Refuge (USFWS 2009) including 6 lots acquired by TNC (USFWS 1987). In addition to the cost to purchase the land, FWS and DFW have expended funds to manage the Refuge and Sanctuary. It would be safe to conclude that well in excess of \$5 million in public funding has been spent thus far to protect seabird colonies and other endemic wildlife on PI (JC 1974, USFWS 1985, 87, 90, 91, 2009).

A Common Vision for the Island

The Nature Conservancy was specific about how the property could be used when they transferred the land that would become the Zella M. Schultz Seabird Sanctuary to the WDG. “This deed is granted with the intent of securing this property as a permanent sanctuary for the preservation of the unique and endangered species of wildlife which exist on said property, including in particular the Rhinoceros Auklet breeding colony. It is further intended that scientific, educational, and other compatible uses may be undertaken on the property” (JC 1974).

The Refuge’s enabling legislation was equally specific. The Protection Island National Wildlife Refuge Act, Public Law 97 – 333, Oct 15, 1982 (96 Stat. 1623), stated; “The purposes of the refuge are to provide habitat for a broad diversity of bird species, with particular emphasis on protecting the nesting habitat of the bald eagle, tufted puffin, rhinoceros auklet, pigeon guillemot, and pelagic cormorant; to protect the hauling-out area of harbor seals; and to provide for scientific

research and wildlife-oriented public education and interpretation (96 Stat. 1623)” and apply to all portions of Protection Island NWR (USFWS 2010b) (See *Appendix* for additional text).

The Sanctuary, the Refuge, and the Deer

There can be little doubt as to the reasons for establishing the Sanctuary and the Refuge. There is also little doubt concerning the trust responsibilities of the managing agencies. Both FWS and WDG founding documents focus specifically on seabirds and scientific research and contain language addressing compatible uses. Conspicuously absent is any mention of terrestrial mammals. In addition, the Sanctuary’s purpose is specific to the “unique and endangered species of wildlife which” existed on the Island at that time; deer did not. Today, PI is an important seabird research station that is closed to public visitation to reduce wildlife disturbance and limit the potential for introductions of exotic species.

Nonetheless deer managed to reach, and subsequently colonize the Island and have now reached a density that threatens to compromise the Island’s mission. However, deer may have an important scientific role to play. Studying and understanding deer impacts on seabirds and on biodiversity in general, and how seabirds and other species respond to their removal could satisfy both the Sanctuary’s, and the Refuge’s scientific purposes. Such research could prove valuable in restoring biodiversity in other island systems and be beneficial to future PI restoration efforts as outlined in the CCP (USFWS 2010b).

III. Methods

Protection Island Deer Counts, February 2010 and March 2011

The first effort to quantify the number of deer on PI was a survey conducted on February 8th, 2010. That effort was followed with a second survey on March 5th, 2011. Direct counts are possible due to the Island's small size (1.473 km²), numerous excellent viewpoints, and relative open vegetation. This direct method likely yielded fairly accurate counts because many of the animals could be seen at one time avoiding double counting of individuals. However, it is possible that some deer were missed (Davis pers. corr. 2010) (Falzetti pers. corr. 2011).

Census Method, 2010

Figure III.1, Protection Island Map 1



On February 8th, 2010, a visual count of deer on PI was made by a two person team on foot and using ATV's. Counters stopped at a series of strategic view points and counted animals (See Figure III.1, Protection Island Map 1). Care was

taken not to re-count deer that moved during the census. Wooded areas were surveyed from the edges including short entries under the canopy in an effort to flush deer hidden by vegetation. Areas that deer cannot access were excluded from the survey. It is estimated that at least 90% of the island was surveyed during the 65 minute count. Visibility was very good and the survey was conducted just before dusk when deer are typically active.

A total of 71 deer were counted in this evening survey. In the summer and fall of 2009, Island caretakers counted 15 new fawns. Between October 2009 and February 2010, 13 deer were found dead on the island including three fawns (Davis pers. corr. 2010). However, there was no indication as to whether all of the deceased deer had perished in the same year.

Census Method, 2011

Figure III.2, Protection Island Map 2



On March 5th, 2011, a second survey of deer was made by another two person

team on foot and using ATV's. Unlike the 2010 count, the process in 2011 began with the team viewing the steep sides of the island from a boat and from the marina and shop area to insure that no deer were on the slopes below the upper plateau. There were no deer in these areas. On the plateau the route was nearly identical to the route taken in 2010, however, the order and number of counting locations was modified in an effort to reduce the possibility of double counting individuals (See *Figure III.2*, Protection Island Map 2). In 2010 deer were counted from 13 locations, in 2011 deer were counted from just 8 locations. In the second count the island was divided into two halves with a line running north/south transecting the water tower and the research cabin road.

Deer were first counted in the central north plateau and woods by both counters. Next, counters moved from the western highlands east to the water tower each confirming the other's counts. Once at the water tower one counter climbed the tower to act as a spotter while the other proceeded to the southeast turnaround and scanned Violet Spit to insure no deer had moved off the plateau to the east. From the water tower the spotter had a nearly complete view of the north/south transect and a clear view of the original groups on the north and south central plateau. He was able to confirm the original count and that no individuals had moved out of that area.

Next, the counter on the ground swept the eastern portion from south to north including the outer forest road, in effect herding or flushing the deer out of the woods towards the transect line. When that individual reached the northern terminus of the transect line, both counters confirmed the group of deer that had moved out of the forest in the northern half of the island, one from the tower to the south and the other from the ground to the north. At the same time the spotter was able to count all the deer flushed out of the forest on the southern half of the line on the ridge adjacent to the tower. Finally the counter on the ground moved back south along the inner forest road to confirm no deer remained in the forest and to count any stragglers of which there were none.

At one time the spotter on the tower could see 72 deer. The counter on the ground could see 4 additional deer behind the ridge east of the tower that the spotter could not see for a total of 76 deer. A sum of all counted groups from the 8 locations resulted in a total of 78 deer. As in 2010, the count took just over an hour, about 70 minutes. However, that does not include the initial survey from a boat and the marina. Care was taken not to re-count deer that moved during the census. Improving slightly on the 2010 method, it is estimated that at least 95% of the island was surveyed during the count excluding areas that deer cannot easily access. Visibility was at least five miles at the start and increasing with an overcast sky. In addition, 4 dead deer were observed during the count and the Island caretaker confirmed there were three additional dead deer on the Island at other locations for a total of 7 known deceased deer. Once again, it was difficult to estimate when the deer perished and several appeared to be dead for more than a year (Falzetti pers. corr. 2011).

Results, Protection Island Deer Density

According to the deer counts conducted in 2010 and 2011, there were at least 71 deer on PI in 2010 and at least 78 in 2011 (See *Table III.A*).

Table III.A, Protection Island Deer Density

Protection Island Deer Density			
Date	Density km²	Density mi²	Density per acre
February 8, 2010	48.2 / km²	124.8 / mi ²	0.195 / acre
March 5, 2011	53.0 / km²	137.1 / mi ²	0.214 / acre

Note: Protection Island is **1.473057 km²** or **0.56875 mi²** (364 acres)

IV. Rhinoceros Auklet (*Cerorhinca monocerata*) Ecology

Description

Rhinoceros auklets are a species of pelagic seabirds that range widely across the North Pacific and nest primarily on undisturbed islands in excavated underground burrows (Sowls et al. 1980). They are the only extant species of the genus *Cerorhinca* and are members of the alcidae family which includes puffins. The name rhinoceros is derived from a horn-like extension of the beak present in breeding adults. Similar to the more elaborate sheath on the puffin's bill, this distinguishing feature is shed annually. Rhinoceros auklets are sometimes referred to as horn-billed puffins or unicorn puffins (Wikipedia 2011). In fact, RHAU are so similar to puffins that the American Ornithologists Union's Committee on Classification and Nomenclature recommended re-naming the species rhinoceros puffin in 2008 (AOU 2008).

Rhinoceros auklets are a medium-sized auk with dark upper plumage and a large orange/brown bill. Males are generally about 10% larger than females and both breeding adults have white plumes above their eyes and behind their bills (Wikipedia 2011). Like puffins, RHAU hunt visually by "flying" underwater or "pursuit diving" using their wings for propulsion (Wilson 1998). To catch prey they dive as deep as 57 m (187') for as long as 148 seconds (Kuroki et al. 2003). Given their reliance on saltwater fish and their susceptibility to predation when nesting in ground burrows, the relatively safe nesting habitat found on islands is critical to the success of the species. Rhinoceros auklets have no known fresh water requirements (Sanders 2009).

Nesting

Rhinoceros auklets nest in burrows which they dig in the ground with their toe nails and beaks, or in natural caves and cavities generally between 0.5 and 5 m deep depending primarily on substrate (Leschner 1976, Wilson 1977), although burrows as long as 8 meters have been reported on Pine Island in Queen Charlotte

Sound (Richardson 1961). They prefer to dig burrows in firm sandy soil with some surface roots providing stability (Richardson 1961, Leschner 1976, Speich and Wahl 1989). Burrows can have branching tunnels, multiple entrances and interconnected chambers and are often near the surface of the ground or under roots and grass tufts where they can be easily collapsed by anything heavy walking on top of them (Leschner 1976, Sowls et al. 1980).

Vegetation type in burrow site selection does not appear to be as important as soil type. On Destruction Island, an important colony off the Washington coast, RHAU are reported to nest in tall grass and salmonberry habitat which is a mixture of salal, salmonberry and willow (Leschner 1976, Wilson and Manuwal 1986). On PI, salmonberry habitat is not available in the traditional breeding areas and the highest RHAU densities are in grassland habitat (Wilson and Manuwal 1986, USFWS 2010b) which is also preferred by deer. However, both locations offer firm friable soils stabilized by surface roots.

Rhinoceros auklets prefer nesting sites on moderate slopes or along the edge of ridges (Richardson 1961, Leschner 1976) and there is significant correlation between burrow density and slope (Wilson and Manuwal 1986). Slopes below 35 degrees are seldom utilized on PI and the preference appears to be for slopes between 37 and 45 degrees on the Island (Richardson 1961). They utilize slopes as steep as 60 degrees in other parts of their range (Vermeer 1979). Steep slopes make it possible for RHAU to dig nearly horizontal burrows where dirt can be easily pushed out (Richardson 1961). They also prefer slight inclines to aid in take-off as they are relatively poor fliers.

Adult RHAU spend their days feeding away from the colony and work on their burrows and feed their young nocturnally (Speich and Wahl 1989). Upon arrival they usually land and walk to entrances rather than flying directly in because locating burrows is difficult in darkness (Leschner 1976). In the colony at night RHAU tend to interact socially and spend much of their time sitting on the hillsides or exploring other burrows (Leschner 1976). Scott et al. (1974)

speculated that their use of burrows and nocturnal habits are in response to predation and kleptoparasitism by gulls. Furthermore, seabird colonial nesting strategies and island habitat selection evolved, at least in part, due to predator pressures (Buckley and Buckley 1980).

Rhinoceros auklets tend to lay a single egg in early May. Both parents share incubation which takes about 45 days. The average nesting period is 54 days. They are particularly sensitive during nesting and brooding and will readily abandon their nests when disturbed (Sowls et al. 1980). They frighten easily which can cause them to drop fish intended for chicks. In one instance on PI, researchers observed a deer startle a RHAU as it approached a burrow with a bill full of fish causing it to drop the food (Hodum and Pearson pers. corr. 2009). Dropping fish can have significant implications since RHAU are susceptible to kleptoparasitism by gulls (Wilson 1993) and such occurrences likely present opportunities for competitors to steal food intended for chicks.

History on Protection Island

The first recorded account of RHAU on PI was by physician and naturalist George Suckley who accompanied General Isaac I. Stevens on the Pacific Railroad Survey starting in 1853. “This curious bird, first described by Bonaparte from specimens said to have been brought from the west coast of America, is found moderately abundant on the lower part of Puget Sound and in the Strait of Fuca. . . . Protection Island . . . is said to be a favorite breeding ground of the species where, according to the accounts given me by Indians, they breed in holes dug in the steep banks, like those of the black guillemot, and are said to have much the same habits. The most remarkable feature of the bird is the characteristic singular wax yellow protuberance on the bill.” (Suckley 1859, pg. 284).

The first comprehensive RHAU surveys on PI conducted in 1958 and 1959 estimated a breeding colony between 6,000 and 8,000 birds (3,000 to 4,000 adult pairs) (Richardson 1961). In 1975 and 1976 the estimated number of burrows was

27,549 (Wilson 1977) and the number of birds was estimated at more than 34,000 adults (17,000 breeding pairs) (Wilson and Manuwal 1986). In 1981, PI was believed to be the largest RHAU colony in the contiguous U.S. (Larsen 1982). By 1983, the burrow estimate on PI had declined to 27,059, although a difference in estimate methodology may account for the apparent discrepancy between the 75-76 and 1983 counts (Thompson et al. 1985). However, by 2000 it appeared the colony had declined to just 24,000 breeding adults (12,000 pairs) (Wilson unpublished, reported by Wilson 2005).

Burrow counts conducted in 2008 estimated 54,113 (+/- 9,390 at 95% CI), significantly higher than all previous studies and double the 1983 estimate. Also, the burrow occupancy rate during breeding was estimated at 66% (+/- 5% at 95% CI) of the 54,113 burrows. Using those estimates, a total breeding colony of 71,430 individuals (+/- 13,514 at 95% CI) was calculated for PI (Pearson et al. 2009). This was conservatively two and a half times the 2000 estimate. However, once again variation in methodology may account, to some degree, for the substantial differences between surveys (Pearson et al. 2010).

Population Trends

Figure IV.1, North American Rhinoceros Auklet Distribution



Source: Bird Life International 2011a

Rhinoceros auklets breed colonially on forested or grass and forb covered islands up to several thousand hectares (Gaston and Dechesne 1996b). They are found in the North Pacific ranging from the Channel Islands in California (McChesney and Tershy 1998) to the Aleutian Islands in Alaska (Gaston and Dechesne 1996b) (See *Figure IV.1*, North American Rhinoceros Auklet Distribution). They are also found on Hokkaido and Honshu Island groups in the Japanese archipelago; on the northern tip of North Korea; on the island of Sakhalin in Russia; and at two locations on the far eastern Siberian coast in Asia (Bird Life International 2011b). Because RHAU are extremely difficult to count while away from the colony, numbers are typically estimated based on burrow occupancy. However, estimates based on burrow occupancy are generally unreliable due to the fact the burrows are often so extensive that researchers cannot access the nest chambers (Gaston and Dechesne 1996b).

Nonetheless, the global RHAU population was estimated at roughly 1 million breeding adults in 1993 (Byrd et al. 1993), which may imply an additional 1 - 2 million pre-breeders (Gaston and Dechesne 1996b) since RHAU don't typically breed until they are 3 – 5 years old (USFWS 2005). However, in 1996, Gaston and Dechesne estimated the North American breeding population alone at just below 1 million breeding adults and speculated that Byrd et al. had underestimated the 1993 global population and that 1.5 million breeding adults was probably a more accurate estimate. A 2011 estimate placed the global adult RHAU breeding population at 1.3 million with a range of roughly 1,140,000 km² (Bird Life International 2011a). It is believed that the global population is now in decline due to predation and competition from invasive species (Gaston and Dechesne 1996b, Bird Life International 2011b).

Regional Population Trends

Currently, more than 95% percent of the North American population of RHAU occurs in Washington, British Columbia, and southeast Alaska (Gaston and Dechesne 1996a) (See *Table IV.A*). Almost all of these birds breed in one of eight

large colonies of which the colony on PI is currently estimated to be the third largest on the continent and by far the largest in Washington State (Pearson et al. 2009). In 2000, Washington State’s RHAU population was estimated at 55,662 breeding adults with about half of those (27,872) occurring on the outer coast (*Tenyo Maru OS Trustees 2000*) primarily on Destruction Island with a few small colonies on other islands. The inland population is found primarily on PI and Smith Island but smaller numbers nest at a few other sites (Puget Sound Science Update 2011).

Table IV.A, Major North America Rhinoceros Auklet Colonies in 1993

Major North America Rhinoceros Auklet Colonies in 1993			
Location	Estimated Breeding Adults (x 1,000)	Location	Estimated Breeding Adults (x 1,000)
California		British Columbia	
Farralon Islands	< 1	Cleland Island	1
28 other locations	< 1	Pine and Storm Islands	140
Oregon		Triangle Island	42
Entire State	< 1	Moore Group	91
Washington		Lucy Island	25
Destruction Island	24	Queen Charlotte Islands	34
Protection Island	34	Gulf of Alaska	
Smith Island	3	Middleton Island	6
Southeastern Alaska		Semidi Island	1
Forrester	55	Aleutian Islands	< 1
St. Lazaria Island	1		
Total	< 461		

Source: Birds of North America Online, 2011.

Protection, Destruction, and Smith Islands are the major colonies in Washington State and make up over 90% of the U.S. population south of Alaska (See *Table IV.B*). There are some indications that the West Coast RHAU population is increasing in general but there is no evidence of any significant new colonies in the State (Speich and Wahl 1989, Seattle Audubon 2011). Furthermore, the Smith Island colony has little expansion potential due to the Island’s small size (0.25 km²) and a lack of moderate slopes, the burrow habitat RHAU prefer (Wilson and Manuwal 1986). Their expansion potential on Destruction Island is also limited due to the Island’s small size (0.15 km²) (Wilson and Manuwal 1986) and

potential competition from invasive rabbits (Pearson et al. 2010). By contrast PI is much larger at just under 1.5 km² and has considerably more slope habitat further underscoring the Island's importance to the species.

Table IV.B, Rhinoceros Auklet Estimates, Primary Washington State Colonies

Rhinoceros Auklet Estimates, Primary Washington State Colonies									
Year	Protection Island			Destruction Island			Smith Island		Source
	Breeding Adults	Burrows	Occupied Burrows	Breeding Adults	Burrows	Occupied Burrows	Breeding Adults	Burrows	
1958-59	6,000-8,000	1,500-2,000							Richardson 1961
1973	18,400								Speich and Wahl 1989
	25,000								Robel 1973
Mid 1970's					23,621			1,194	Wilson & Manuwal 1986
1976				32,324	27,394	16,162			Leschner 1976
1975-76	34,000+	27,549	17,108						Wilson 1977, Wilson & Manuwal 1986
1978							1,200		Manuwal et al. 1979
1979				34,324			2,588		Speich and Wahl 1989
1983	40,600 ₁	27,059							Thompson et al. 1985
1983-84							3,000		USFWS 2010b
1984	34,216			23,600			2,588		Speich and Wahl 1989
1985	34,000								USFWS 1985
1986							1,200		USFWS 2010b
2000	24,000								Unpublished data cited in Wilson 2005
2008	71,430 ₂	54,113 ₃	35,715 ₄						Pearson et al. 2009
2009				13,018 ₅	11,222 ₆	6,509 ₇			Pearson et al. 2010
Latest Estimate	71,430₂			13,018₅			1,200		

₁ Reported in USFWS Seabird Conservation Plan. 2005. Attributed to S.P. Thompson et al. 1985

₂ +/- 5% (13,514) at 95% CI

₃ +/- 5% (9,390) at 95% CI

₄ +/- 5% (6,757) at 95% CI

₅ +/- 5% (4,226) at 95% CI

₆ +/- 5% (3,036) at 95% CI

₇ +/- 5% (2,113) at 95% CI

Another important difference is that PI and Smith Island lie in the dry rain-shadow of the Olympic Mountains and receive only about 41 cm of rain annually. Destruction Island receives an annual rainfall of over 192 cm. Some of this precipitation comes in the form of heavy downpours from Pacific Storms that can flood burrows (Leschner 1976) and disrupt or terminate incubation or cause chicks to abandon their burrows. PI and Smith Island burrows are occupied in the dry season and typically do not have problems with flooding during incubation

(Wilson and Manuwal 1986). While no damage from flooded burrows has been quantified for Destruction Island, burrow flooding reduced Atlantic Puffins (*Fratercula arctica*) by 50% on Great Island, Newfoundland (Nettleship 1972).

Salish Sea Rhinoceros Auklets

Recent research suggests the Salish Sea RHAU population is healthy and increasing, however, researchers caution against making direct comparisons with earlier counts as the methodologies differ (Pearson et al. 2010). In sharp contrast, the only other large RHAU colony in the state, located on Destruction Island, has experienced a dramatic decline which may be due in part to non-native rabbits competing for burrows (Pearson et al. 2010). The Destruction Island and PI colonies were comparable in size in the mid 1970's but have headed in opposite directions. At that time Destruction Island had an estimated 16,162 occupied burrows (32,324 breeding adults) (Leschner 1976) and PI had an estimated 17,108 occupied burrows (34,216 breeding adults) (Wilson and Manuwal 1986). In 2009, it was estimated that there were just 6,509 (+/- 2,113) occupied burrows on Destruction Island while the 2008 burrow occupancy estimate for PI was 35,715 (+/- 6,757) (Pearson et al. 2010). This is a dramatic difference for colonies that had been similar just three decades earlier and one that underscores the importance of the success of the PI colony to the species as a whole.

Threats to Rhinoceros Auklets

Due to their feeding and breeding habits and low reproductive rates, RHAU are particularly vulnerable to a great number of threats including fluctuating ocean climate conditions such as El Niño events which can impact food resources (Wilson and Manuwal 1986, U. Wilson 1991, M. Wilson 1998, U. Wilson 2005). During the severe 1983 El Niño event an abundance of dead seabirds washed up on California beaches suggesting that major changes in oceanographic conditions can lead to heavy mortality. From 1973 to 1977, the RHAU colony on Farallon Island near San Francisco doubled annually, but during the 1983 El Niño few

individuals could be found (Gaston and Dechesne 1996b).

El Niño events result from shifts in prevailing winds which tend to impact RHAU food resources in coastal areas more than in the Salish Sea. Coastal upwelling is a function of prevailing winds and introduces nutrients into the euphotic zone promoting fish populations. El Niño events tend to be accompanied by a decrease in upwelling which reduces the nutrient supply and subsequently the RHAU food supply. In contrast, Salish Sea fish species preferred by RHAU are supplied nutrients primarily by tidal mixing in the more stable inshore water system (Wilson and Manuwal 1986).

During the 1983 El Niño event Destruction Island RHAU diet shifted from their normal prey species, primarily northern anchovy (*Engraulis mordax*) with lesser amounts of rockfish species (*Sebastes* sp.), herring (*Clupea harengus*), and night smelt (*Spirinchus starksi*), to Pacific saury (*Cololabis saira*) (Wilson 1991).

During the 1976 El Niño event researchers found uneaten Pacific saury in burrows on Triangle Island, B.C., and speculate that chicks may have difficulty eating these fish (Vermeer 1980). Also, Pacific saury are usually found farther offshore than typical RHAU prey and are lower in nutritional and energetic value (USFWS 2005). Declines in normal prey species resulting from changes in ocean conditions were likely responsible for RHAU breeding failures in northern British Columbia in 1976 (Vermeer 1978, 1980).

By contrast, Salish Sea RHAU colonies rely primarily on Pacific sandlance (*Ammodytes hexapterus*) and herring and only infrequently on northern anchovy. During El Niño years, sandlance and herring continue to be a fairly typical Salish Sea RHAU diet component (Wilson and Manuwal 1986, Wilson 1998) suggesting that these fish stocks may be relatively stable in the inland waters where PI RHAU feed. Because their food supply is more stable, the PI RHAU colony is somewhat buffered from fluctuating ocean conditions including El Niño events.

It is also possible that the calmer waters of the Salish Sea make foraging for food

easier than in the rough open seas (Wilson and Manuwal 1986). A number of factors support this concept. For example, PI and Smith Island RHAU tend to return to burrows with heavier bill loads of fish and subsequently their chicks tend to grow faster and weigh more at fledging than in coastal colonies (Wilson and Manuwal 1986). On average between 1974 and 1983, PI RHAU returned to their burrows with 12.4% more fish than did Destruction Island RHAU (Leschner 1976, Wilson and Manuwal 1986). In addition, comparing the years for which data are available suggests the PI colony tends to have a generally higher reproductive rate than coastal colonies (Leschner 1976, Wilson 1977, Wilson and Manuwal 1986).

Disturbance

In addition to fluctuating ocean conditions, RHAU face a number of other threats. For example, human disturbance can cause seabirds to flush and abandon nests leaving eggs and chicks vulnerable to predation and starvation. Disturbance can also interfere with the ability for adult birds to rest and feed and can increase their predation exposure. Sources of such disturbances include the activities of researchers. Wilson (1977) reported that 28-30% of occupied RHAU burrows on PI were deserted as a result of observer activities. Coast Guard search and rescue operations and military exercises have caused un-quantified disturbances to RHAU colonies. Noise from helicopters and other aircraft and searchlights at night cause birds to flush (Speich and Wahl 1989) which can increase the potential for injury and likely reduces RHAU reproductive success.

Human disturbance from boats passing close to colonies and from people trespassing on breeding islands can also be significant. Such disturbances can alter RHAU feeding behavior, flush birds, and result in nest abandonment. Trespassers often do not understand the extent of their impacts. Pets in particular can have detrimental consequences for seabird colonies. For example, in 1973 a single dog owned by a lighthouse keeper killed 10% of the Smith Island's RHAU colony (Manuwal et al. 1979). Smith Island is approximately 16.5 km north of PI.

Human Caused Habitat Changes

One of the greatest threats to seabirds including RHAU is human disturbance of burrow habitat. These disturbances can be direct, such as collapsed and destroyed burrows, or indirect, i. e., actions that cause changes in drainage patterns affecting soil moisture retention and erosion; soils compaction and hardening; reduced soil biotic activity; and changes in vegetation structure and composition including the establishment of invasive species and decreases in plant soil retention qualities. Habitat alteration as a result of development has historically been a threat to seabirds in general and to RHAU in particular.

For example, PI has a history of substantial habitat loss from human intrusions that include direct destruction of burrows during construction of roads and other infrastructure, as well as indirect disturbance from heavy machinery, motor vehicles and aircraft (Larsen 1982, USFWS, 1988, Clark 1995). Although the development of U.S. Coast Guard facilities on Smith and Destruction Islands and construction of the residential community on Protection Island have all been discontinued, the legacies of those endeavors linger in the form of unstable slopes, roads, trails, and at least one active human residence within a RHAU colony (PI) (USFWS 2010b).

Burrow collapse

Burrow collapse in seabird colonies is well documented (Bancroft 2005, 2009). Causes include self-destruction and erosion facilitated by the resident birds themselves and collapses caused by heavy animals and humans walking on the surface above. For example, large mammals caused extensive damage resulting in significant loss of manx shearwater (*Puffinus puffinus*) burrows on the Pembroke Islands, U.K., and nearly 70% of the wedge-tailed shearwater (*Puffinus pacificus*) burrow entrances on Rottneest Island, Western Australia, exhibited partial damage or complete collapse from humans in just one year, 2005 (Bancroft 2009). Estimated accidental collapses on Rottneest Island occurred at a

rate of 0.93 burrow collapses per person hour for every 0.1 burrows / meter² (Bancroft 2009).

Of course the timing of the collapse is an important factor to consider when assessing the actual impacts to reproductive success. Seabirds typically re-excavate burrows prior to the nesting season which presumably has little effect on breeding success (Wilson and Manuwal 1986, Bancroft et al. 2005). However, collapses during the breeding season can cause mortality to eggs, chicks, and even adults (Bancroft 2009). The indirect effects of collapses on RHAU reproduction (i.e. beyond direct mortality) are far more difficult to assess.

Threats to Foraging Seabirds

Other threats to seabirds come from recreation and fisheries activities including collisions with commercial and private vessels and bycatch in working and derelict driftnets. For example, in 1993 and 1994, common murre (*Uria aagle*) and RHAU were the two most frequently entangled species in sockeye (*Oncorhynchus nerka*) and chum (*Oncorhynchus keta*) salmon gillnets in northern and central Puget Sound where they comprised more than 90% of all entangled seabirds (Thompson et al. 1998). Fisheries also directly compete with RHAU for food resources. While the full impacts to RHAU from net entanglements, collisions with boats, and competition with humans for food have not been quantified, their combined effects on the Salish Sea RHAU population may be significant. (Thompson et al. 1998, Wilson 1998)

Contaminants

Ingestion and/or exposure to even small amounts of oil can compromise a seabird's health, lead to hypothermia, and cause death. This includes oil from spills as well as from non-point sources such as storm-water runoff. Serious spills can kill hundreds of thousands of marine birds in a single occurrence and represent a major threat to local colonies. Because they spend so much time resting on the water's surface and diving for food and because they have few large

breeding colonies in North America, RHAU are one of the most susceptible seabirds to oil pollution (Remsen 1978, Gaston and Dechesne 1996b). The PI colony and the smaller one on nearby Smith Island are at greater risk than most RHAU because their colonies lie along a major shipping route used by crude oil tankers heading for the refineries in the Salish Sea.

Pollutants such as organochlorines which encompass a broad array of highly toxic compounds are extremely persistent and can create serious health risks for individual birds as well as for seabird populations, especially when considering the effects of bio-accumulation (Elliot and Noble 1993). Complications can include; lowered reproductive rates including reduced shell thickness; organ failures; neurological disorders; embryonic deformities and other abnormalities; and higher mortality rates (Mills et al. 2005). Trace elements including cadmium, lead, mercury and selenium have also been found in North Pacific seabirds in levels known to cause adverse effects in other species (Ohlendorf 1993).

In addition, RHAU are undoubtedly affected by marine debris, especially plastics which are ubiquitous and can cause mortality. Plastics resemble food to wildlife and cause serious health conditions ranging from strangulation to starvation (Kingfisher et al. 2009). There are other ways plastics can impact seabirds.

Because plastic can absorb and concentrate toxins such as those mentioned above from the surrounding seawater, they can facilitate toxic accumulations in seabird tissues if they are consumed leading to the health complications mentioned previously.

Plastics can also indirectly impact seabirds by altering ecosystem functions such as reducing the exchange of gasses between seawater and sediments, and by altering the chemical composition of the benthic environment (Mills et al. 2005). These changes can impact water quality and seabird food resources. How marine debris such as micro-plastics effect wildlife is an emerging topic but research is beginning to show a disturbing pattern of pervasive impacts on seabirds (Ryan et al. 1988, Kingfisher et al. 2009)

Predation

Both native and non-native predators present a serious threat to RHAU. For example, substantial RHAU colony size reductions (26,000 to 4,000 in about 6 years) have been attributed to raccoons introduced on Helgesen and Saunders Islands in the Queen Charlotte Islands, B.C. (Gaston and Dechesne 1996b, *Birds of N.A.* Online 2011). On PI, RHAU face threats from native birds of prey including eagles and owls. In fact, the most common bones encountered when studying eagle prey remains on the Island were from RHAU. Their bones were found in greater quantities than gull bones suggesting that eagles preferentially prey on RHAU (Hayward and Henson 2008). Owl pellets also contained a very high percentage of RHAU remains; 93% of owl pellets collected on PI contained RHAU bones and nearly 94% of those contained RHAU bones exclusively (Hayward and Henson 2008). The threat to RHAU from eagles is likely to increase with the growing eagle population (Hayward and Henson 2008, Davis pers. corr. 2010).

In addition to owls and eagles, PI RHAU face threats from other predators and competitors including river otters and gulls. For example, each year a large number of RHAU chicks are killed by gulls as they attempt to reach the water after fledging (Hayward and Clayburn 2004). They could also face threats from potential invaders including rats and raccoons and from introduced domestic animals including dogs and cats which is a primary reason the Island is closed to the public.

Mitigating Threats

Rhinoceros auklet breeding and feeding habitat are under a number of threats. Many of the threats are not fully quantified, or even studied at present, and most are out of the control of DFW and FWS. Some threats such as climate change and El Niño events are beyond the regulatory scope of resource agencies, while other threats caused by fisheries, pollutants, and predation are difficult to influence.

Mitigating threats to breeding habitat is far more realistic. In fact, Refuge and Sanctuary managers can effectively mitigate two significant threats, direct human disturbance and non-native invaders. Human disturbance from residents, staff and researchers can be addressed through policy, and human and pet disturbance from trespassers can be addressed to some degree through public education and law enforcement. Monitoring and rapid response can potentially mitigate impacts from new invasive arrivals. In the cases of PI and Destruction Island management actions can address the threat from deer and rabbits respectively.

For the most part the agencies that manage PI have effectively minimized the threat from human activity on the Island by purchasing the land and regulating its use including closing it to the public and limiting the number of people on the Island. The presence of a resident caretaker is a trespass deterrent and those individuals that are allowed to visit are reminded to avoid sensitive habitat unless specifically involved in approved research. Furthermore, staff and researchers are on the constant lookout for invasive animals and would undoubtedly be quick to inform managers in the event of a sighting. But simply documenting arrivals and/or sightings is not enough. Managing agencies must act on that information if the threat is to be mitigated.

V. The Effect of Introduced Ungulates on Island Biodiversity

Biodiversity

The term biodiversity (biological diversity) can be defined as the summary of multivariate statistics that quantify the characteristics of a community (van Wieren and Bakker 2008), that is the variation within and between life forms in a given ecosystem, in this case within island communities. Just how to fully measure the parameters that define biodiversity is a question that continues to engage scientists, however, species richness, evenness, and diversity are among the most commonly used biodiversity indicators even though biodiversity has come to be defined in much broader terms (Groom et al. 2006, van Wieren and Bakker 2008). Certain changes in biodiversity are believed to be indicative of an ecosystem's health. Reductions in diversity (species richness and evenness) generally lower a system's stability and productivity and can lead to an increased vulnerability (Gaston and Martin 2002) to invasion by exotic species.

Non-native ungulates impact island species richness and diversity in a variety of direct and indirect ways and their impacts can be positive or negative. For example, ungulate browsing can stimulate plant growth and increase plant competitive abilities, increase seed dispersal, increase fertilization by increasing rates of nutrient cycling (feces and urine), and control undesired vegetation growth. However, in most cases non-native ungulates tend to decrease species richness and diversity on islands (Courchamp et al. 2003, Skarpe and Hester 2008), a phenomena closely linked to ungulate density (see below).

Ungulate grazing and browsing can selectively alter species evenness and species richness; modify soil characteristics and drainage patterns; reduce the quality and availability of certain habitat types thereby increasing competition between affected species for suitable habitat; and can indirectly elevate predation pressures on specific species such as seabirds through habitat alteration and disturbance. Such effects can trigger trophic cascades and thus have far reaching implications

for the ecosystem. Ungulates have the ability to modify the composition of island communities and in severe cases they can significantly alter ecosystems beyond the point of recovery (Gaston et al. 2008).

Ungulate Density; Cattle, Goats, Sheep and Deer

Ungulate impact on island biodiversity is primarily a function of the island's history of exposure to ungulates and ungulate density (Albon et al. 2007). While most communities can tolerate low ungulate densities, and many can tolerate moderate densities without dramatic changes in biodiversity, few can tolerate high ungulate densities (McShea et al. 1997, Harrison and Bardgett 2008). The ungulate impacts documented in this section are associated with cattle, goats, sheep and deer (including reindeer).

In general, sheep and cattle tend to have highly localized impacts on vegetation and soils likely due to the fact that these species tend to aggregate into groups and have limited ranging behavior. Deer on the other hand usually impact vegetation and soils less than other ungulates at similar densities because they range further and generally do not aggregate into large groups. However, Albon et al. (2007) found that red deer (*Cervus elaphus*) in Scotland, which are larger than Columbia black-tailed deer, at high densities (>40 individuals / km²) tended to form larger groups and range less causing impacts more closely resembling those of sheep herds.

Deer have the potential for rapid population growth due to their ability to exploit a wide variety of food resources, early reproductive age, annual reproduction, and capacity for twinning (Gillingham 2008). For example, a group of six white-tailed deer (*Odocoileus virginianus*) (4 females and 2 males) introduced into a deer free enclosure in the George Reserve in Michigan grew into a herd of 162 animals in just 6 years (McCullough 1997). It is important to note that deer can reduce species richness and diversity at less than 25% of their carrying capacity (de Calesta and Stout 1997).

Vegetation

Perhaps the most obvious impacts from ungulates are changes in vegetation. Browsing can result in loss of plant organs, altered and reduced canopies, and changes in species assemblages, which in turn can cause vegetation gaps and changes in the colonization matrix including increased exotic plant invasions. Grazing can also result in root exudation and changes in root biomass leading to a dominance of defended plants (those not selectively browsed) at the expense of selected (browsed) plants (van Wieren and Bakker 2008).

Plants respond to foraging in a variety of ways ranging from sudden plant deaths to increased growth and competitive abilities. Herbivores can increase or decrease species richness depending on the amount of plant biomass, on grazing intensity in general, and on selective grazing intensity on a dominant species. In fact, the utilization of herbivores can be an effective land management strategy at low and even moderate intensities where the goal is to manipulate vegetation for a specific purpose such as to limit plant growth (either biomass or location coverage) or select for certain species. For example, goats are often used to reduce understory foliage or limit growth of native and/or invasive plant species.

One of the earliest recorded examples of ungulate impacts on plant species richness comes from the Great Island in New Zealand, where introduced goats trampled, overgrazed and overbrowsed the landscape to the extent that a great number of plant species went locally extinct. In the 35 years after their introduction, goats reduced the variety of flora from 143 to 70 species (Turbott 1948). There are many other examples where the introduction of ungulates on islands has resulted in severe overgrazing and reductions in species richness (Courchamp et al. 2003, Campbell and Donlan 2004, Donlan and Heneman 2007, Gaston et al. 2008). In some cases the damage is so extensive that the grazers literally starve themselves to death (Ebbert and Byrd 2000).

In many systems, low and even intermediate levels of herbivory result in high

species richness. In general, a system with a longer grazing history can sustain higher grazing pressures and greater variation in grazing density. However, even the vegetation in those systems can be degraded at high ungulate density. Where the main goal is conserving biodiversity, the best option seems to be to maintain herbivore densities similar to those under which the system evolved (van Wieren and Bakker 2008). On islands those densities are often zero.

Soil Compaction and Erosion

Another well documented impact from ungulates is soil compaction and the resultant changes to soil drainage patterns which can alter plant structures and community composition and dramatically increase erosion. For example, sheep introduced in the Channel Islands in the mid-1800's eventually reached densities far beyond carrying capacity, causing considerable destruction of native flora as well as extensive soil compaction and erosion (McChesney and Tershy 1998). Feral goats introduced in the 1700's on Guadalupe Island nearly stripped it to bare soil increasing erosion and causing extensive local extirpation of flora and fauna (McChesney and Tershy 1998). Where reindeer (*Rangifer tarandus*) have overgrazed hilly areas on Alaska islands the result has been the permanent loss of natural plant communities, increased erosion, and in extreme cases, the creation of desert conditions (Ebbert and Byrd 2000).

Soil Composition

Less obvious ungulate impacts include changes in soil characteristics including decreased soil moisture, increased soil temperatures, and reduction or increase of soil biotic activity. Herbivores can increase soil temperature by removing live vegetation which reduces litter and decreases ground insulation – a process that allows more sunlight to reach mineral soil (Pastor et al. 1993). This in turn may lead to lower soil moisture and soil hardening. In arid environments such as deserts, the disruption of biological soil crusts (the community of organisms living on the soil surface including cyanobacteria, green algae, microfungi,

mosses, liverworts and lichens (USGS 2011)), combined with changes in vegetation can lead to increased wind and rain erosion (Harrison and Bardgett 2008).

Because nitrogen can limit plant growth, nitrogen availability and cycling rates are important to plant productivity. Herbivores can significantly alter nitrogen cycling rates either increasing or decreasing the availability of nitrogen. Because carbon's availability is linked to nitrogen, it changes in tandem (Harrison and Bardgett 2008). Soil microbial activity responds to these changes either positively or negatively depending on the situation. For example, the regeneration capacity of tall grasses usually decreases with increasing grazing pressure (Hobbs 1996). However, in productive grasslands a positive feedback mechanism can occur in which dominant grazed grass species exhibit compensatory growth which inhibits colonization by late successional plants that attract grazers less and tend to produce poorer quality litter that does not decompose as efficiently.

Additional grazing attracted by the compensatory growth returns carbon and nitrogen to the soil in the form of dung and urine, and as enhanced rhizodeposition. This in turn stimulates soil microbial activity further enhancing nitrogen and carbon (nutrient) cycling (Hobbs 1996) resulting in higher quality forage and thus more grazing and more excrements. However, this positive feedback loop can be compromised when increased grazing surpasses the enhanced production levels. It is important to note that this feedback process can be localized to small areas and may be insufficient to negate overall adverse grazing impacts (Harrison and Bardgett 2008).

A negative feedback mechanism can also occur in instances where browsing is focused specifically on nutrient rich plants. The result can be non-selection of plants that tend to return nutrients to the soil as recalcitrant litter which does not decompose well and thus lowers biotic activity and nutrient cycling despite the addition of browser excrements (Pastor et. al. 1993, Hobbs 1996). The resulting soils in turn do not promote additional growth of nutrient rich plants. Negative

feedback may also occur where high grazing densities affect physical soil properties that results in microbial activity reductions or where substantial nutrient loss occurs from erosion (Harrison and Bardgett 2008).

Invasive Plants

Ungulate pressures on native plant species can facilitate the recruitment and establishment of invasive species. As the cover or biomass of certain native plants is reduced, windows of establishment opportunity are opened for the introduction of non-native plants. Vegetation gaps created by grazing, trampling, and increased erosion create conditions favorable to colonization by exotics. A good example of this is the establishment of exotic dandelions resulting from the effects of introduced cattle on Sanak Island in the Aleutian archipelago (Rudebusch 2008). On Sanak, cattle have promoted a host of changes to the plant community resulting from the combined effects of one exotic species, cattle, facilitating another, dandelions. See Sanak Island in this section.

Avian and Invertebrate Communities

Large herbivores not only disturb vegetation and impact soils, they can also affect avian communities in several ways. First, they can carry parasites such as lice and fleas which may have drastic consequences for island wildlife, especially birds. Furthermore, reduced distribution and abundance of native birds have been linked to loss of foliage associated with grazing and browsing through a loss of nesting habitat, food sources, and protective cover from predators (de Calesta and Stout 1997, Gaston et al. 2008). There is also a growing body of research suggesting that large herbivores on islands have lowered invertebrate richness and promoted exotic plant invasion by altering plant communities and changing pollinator abundance and behavior (Stockton 2002, Allombert and Martin 2002, Rudebusch 2008).

Predation

Ungulates are usually categorized as herbivores although they do consume insects

as an incidental part of consuming vegetation. There are also extreme cases where sheep and deer consume seabirds. For example, sheep in the Shetland Islands are known to eat the legs off unfledged Arctic turns (*Sterna paradisaea*) and red deer on the Scottish island of Rum consume the heads and legs of manx shearwater chicks as a way to satisfy calcium and possibly a phosphorous deficiency in the island vegetation. Deer in particular have a high demand for calcium required for annual antler growth. On the Island of Rum, deer consume on average 36 grams of shearwater bones per breeding season killing up to 4% of the colony's chicks (Owen 2003).

Queen Charlotte Islands

Perhaps one of the best case studies for understanding the impacts of black-tailed deer on island ecosystems is that of the Queen Charlotte Islands archipelago (hereafter QCI) in British Columbia, where researchers have studied impacts to island flora and fauna. Sitka black-tailed deer, which are similar to Columbia black-tailed deer, were introduced to QCI in 1878 (Golumbia et al. 2002) and flourished in the mild maritime climate and in the absence of predators (Hatter et al. 2000). Deer on QCI have likely altered species richness and diversity more than any other introduced species (Golumbia et al. 2002). QCI deer density ranged from 13 to 30 deer / km² on most islands in 2002 (Martin and Baltzinger 2002). On two islands where deer were removed, cull data suggested much higher densities at 27 - 34 deer / km² (Gillingham 2002). The most recent black-tailed deer density estimate for PI is 53 deer / km² (This paper).

Browsing and grazing results in a direct loss of foliage and heavy browsing can result in dramatic declines and local extirpation of plant species. For example, as early as 1957, researchers on SGang Gwaay in the QCI noted that introduced Sitka black-tailed deer had created large areas totally devoid of living shrubs and completely stripped salal bushes below a height of 1.5 m (Duff and Kew 1958). Heavy deer browsing has dramatically reduced and homogenized understory vegetation and retarded the regeneration of many tree species. The resulting

sapling free open understory has become a striking hallmark feature of the landscape (Pojar 2002).

Deer have selectively altered the vegetation on the nearby Island of Haida Gwaii leaving species such as western red cedar and yellow-cedar with little ability to regenerate. Numerous other species including skunk cabbage and devil's-club have almost completely disappeared from the Island's forests and historically abundant plants such as Nootka rose, Pacific crab apple, and western yew are now rarely seen (Golumbia et al. 2002). Also, a great number of other important species are in severe decline.

Research in the QCI also suggests that plants that evolved in the absence of herbivores are more susceptible to overgrazing. For example, western red cedars on Haida Gwaii have lower concentrations of terpenes, a chemical known to play a role in plant defense, than their mainland counterparts and are therefore more desirable to deer. Such findings support the hypothesis that chemical defense production is costly to plants and diminishes subject to a lack of constant selection. Concentrated browsing by deer not only has the potential to severely reduce abundance, but to also alter the genetic composition of a plant species (Vourc'h et al. 2002).

Studies of the impacts of ungulates on other components of the ecosystem suggest a web of interactions that have exacted a heavy toll on biodiversity in the QCI. Deer impacts on vegetation cascade into other components of the system. While the repercussions for invertebrate communities have not been fully explored, moderate to heavy browsing on the forest understory can greatly reduce the quantity of resources available to invertebrates leading to an overall invertebrate "impoverishment" (Allombert and Martin 2002).

Deer browse tends to lower the habitat quality of litter leading to a variety of invertebrate responses. For example, on the islands in the QCI where deer have been present for 50 years or more, forest arthropod abundance is conservatively

estimated to be six times lower and species richness five to ten times lower. The decrease in species richness occurs in all major taxonomic groups of insects (Allombert and Martin 2002). Allombert and Martin (2002) speculated that the loss of numerous pollinating insects could have additional wide ranging implications. Evidence from Haida Gwaii also suggests a direct correlation between the length of deer browsing history and a reduction in songbird distribution and abundance (Martin et al. 2002). The biodiversity declines on Haida Gwaii linked directly to deer are as severe as seen anywhere in the world (Gill 1999).

The QCI are a dire warning regarding the vulnerability of island ecosystems. Todd Golumbia, a QCI researcher articulated the dilemma when he wrote; “One could argue that over time, the introduction of species could be an inevitable step in the natural progression towards a more homogenous global ecosystem. Ecological conditions could be allowed to evolve untouched with the hope that introduced species will reach equilibrium. Yet, to do nothing is contrary to the goal of maintaining biodiversity, since certain species may be eliminated” (Golumbia et al. 2000, pg. 27). Deer on QCI have induced a trophic cascade that has resulted in substantially reduced species richness and diversity that may be leading to what has been termed an "invasional meltdown" (Simberloff and Von Holle 1999). Meltdown occurs when a non-native species impacts an ecosystem to the point where that species is facilitating the establishment of other exotic species thus significantly compounding the impacts to native species.

Gulf and San Juan Islands Archipelagos

A recent study of deer impacts on island biodiversity in the Gulf and San Juan Island archipelagos looked at 18 U.S. and Canadian islands, ten with no or few deer and the rest with densities ranging from 13 / km² (Wallace Island) to 114 / km² (Sidney Island). Six were considered moderate with 13, 21, 22, 25, 30 and 38 deer / km², and two were considered high with 105 and 114 deer / km². The study revealed that low deer and deer-free islands had twice as many bird species as

islands with moderate and high densities defined as 13 – 114 individuals / km². Furthermore, there was no significant difference between avian species evenness and diversity on islands with moderate versus high deer density. On those islands the forest understory vegetation was reduced and simplified in the 0.5 – 1.5 meter height strata and contained areas devoid of shrub cover (Martin et al. 2011).

Deer browsing can prevent growth of culturally important species such as common and great camas (*Camassia* sp.), fawn and chocolate lilies (*Erythronium* sp.), sea blush (*Plectritis congesta*), blue-eyed Mary (*Omphalodes verna*), lupine (*Fabaceae* sp.), onions (*Allium* sp.) and various brodia (*Brodiaea* sp.) (Arcese and Martin 2011). Browsing at moderate and high densities had dramatic effects on both vegetation cover and architecture and significantly reduced bird species that depend on understory vegetation. For example, on islands with moderate and high deer density compared to islands with no and low deer density, rufous hummingbirds (*Selasphorus rufous*) and fox sparrows (*Passerellia iliaca*) were 9 times lower, spotted towhees (*Pipilo maculatus*) were 25 times lower and varied thrushes (*Ixoreus naevius*) were 29 times lower. Dark-eyed juncos (*Junco hyemalis*) were the only species with a significantly higher abundance on high deer density islands likely because this species prefers open forest with sparse vegetation cover (Arcese and Martin 2011, Martin et al. 2011).

Islands with low deer densities had considerably higher bird diversity and significantly lower evenness (Martin et al. 2011). High browsing pressures lowered the distribution and abundance of native plants and birds and reduced the number of bird species. Importantly, researchers hypothesize cohorts of palatable shrub species are relatively old on islands with medium and high deer densities because young replacement plants are selected by browsers. This potentially represents an “impending extinction debt” arising when the older plants die and there are no young plants to replace them. Thus the full impact to island biota from high deer densities may not be realized for decades (Martin et al. 2011). Martin et al. (2011) suggested that islands with dense deer herds may act as “population sinks”, and deer densities above 10 / km² are likely too high to

maintain diverse bird communities.

Sanak Island, the Invasional Meltdown Concept

Evidence from Sanak Island, one of the easternmost islands in the Aleutian archipelago, suggests that introduced feral cattle (*Bos taurus*) are responsible for a multi-trophic cascade on the Island. This cascade is occurring through the following process; 1) cattle facilitate the establishment of Kentucky bluegrass and dandelions which indirectly results in smaller Island vole populations, which in turn leads to changes in the community composition of insects; 2) cattle support higher abundances of cattle-associated non-native insect species, which in turn alters insect community composition which effects insect behavior including pollinator routines. Dandelions likely also promote the establishment of non-native insects and; 3) non-native ground vegetation further reduces vole populations and promotes non-native plant and insect establishments (Rudebusch 2008). Sanak Island is a case where a non-native species has facilitated the establishment of other non-natives which in turn further facilitates non-native establishments. It appears that through this process Sanak Island is indeed experiencing an "invasional meltdown".

Invaded Insular Ecosystems

There are several reasons why non-native species have relatively high impacts on insular island ecosystems. Because they have evolved in isolation, species on islands typically have not developed traits necessary to respond to exploitation. Islands may provide better or more resources for non-native species because those islands may have fewer competitors and predators. In fact, plant communities on oceanic islands are often composed of very palatable and vulnerable species lacking toxins and physical deterrents, while demonstrating high fecundity and compensated growth (Courchamp et al. 2003). By exploiting such conditions, ungulates including deer have the ability to modify whole island communities and in severe cases they can significantly alter ecosystems beyond the point of recovery (Gaston et al. 2008).

Studies from the Queen Charlotte Islands and the Gulf and San Juan Islands archipelagos suggest a remarkably consistent pattern of ecosystem change after the introduction of deer. They confirm that deer, especially at high densities, reduce island species richness and diversity and offer insight into what can be expected on islands that have evolved in the absence of such invaders. Specifically, they offer a warning of what is likely to occur on PI if deer are not actively managed.

Ungulate History on Protection Island

Beginning as early as 1874, PI experienced grazing pressure from introduced ungulates including cattle, horses, and sheep. Between 1888 and 1912, sheep and cattle "rapidly overgrazed the Island, causing erosion that took parts of the grassland down to bare rock" (Clark 1995, Actions: 1888-1912). By 1912, areas on the upland plateau had been reduced to hardpan, and by 1920, blowing eroded sands created large dunes that consumed trees (Powers 1976). Up to 40 retired racehorses grazed on the Island into the early 1960's and sheep grazing continued until 1968 (Clark 1995, Larsen 1982).

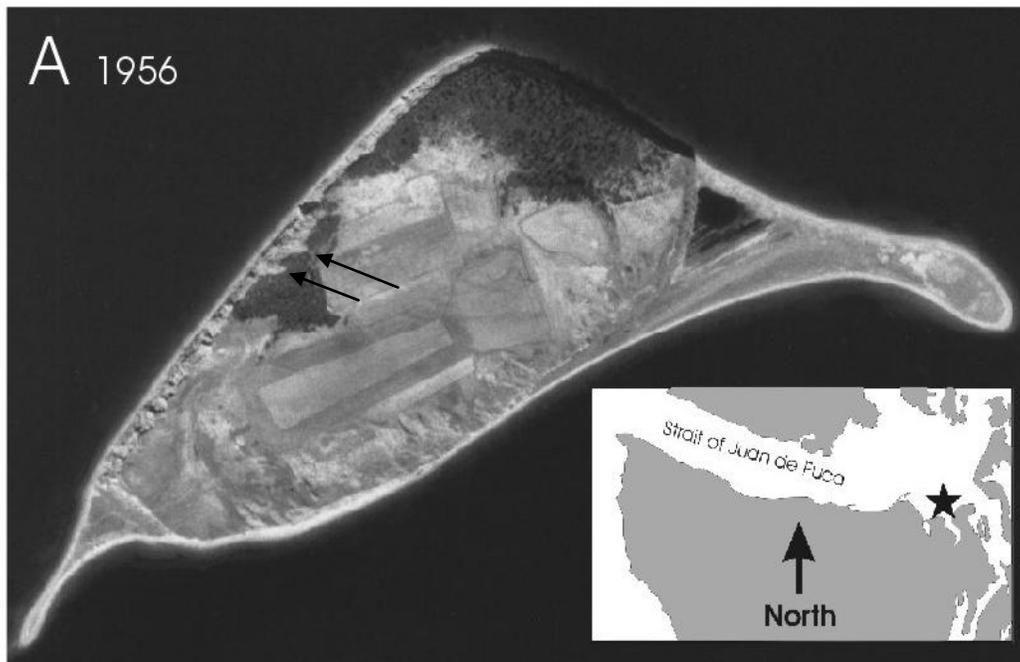
Frank Richardson, an early RHAU researcher, noted that; "... some 100 to 300 sheep have grazed freely over the island and are seriously affecting the breeding slopes of the auklets. Grazing chiefly on annual grasses and what alfalfa is left [from farming], followed by frequent trampling of certain parts of the slopes as they become dry in early summer, has led to the formation of many slide areas of loose sand and soil. Auklet burrows have thus been buried in some regions and the slopes made unusable, or auklets in less severely affected areas must persistently dig out partly filled-in burrows" (Richardson 1961, pg. 458).

Although burrows are typically re-used year after year, Richardson noted that at least 57 of the 76 burrows in use in 1957 were used in 1958 but just 32 of those were used again in 1959. Richardson surmised that much of the reduction was "due to burrows being caved in or buried" (Richardson 1961, pg. 462). In fact, 46% of 76 RHAU burrows in the study area were collapsed by hoofs or buried by

slides in 1958 and 1959 (Richardson 1961).

By the time sheep were finally removed from the Island there was significant evidence of overgrazing. A photograph from 1956, shortly before grazing ceased, shows areas of bare compacted soil and slumping sections along the northwest bluffs with pronounced erosion channels (Cowles and Hayward 2008) (See *Figure V.1*, Historical Photograph A).

Figure V.1, A: 1956 Protection Island Photograph



Note: arrows indicate slumping sections.

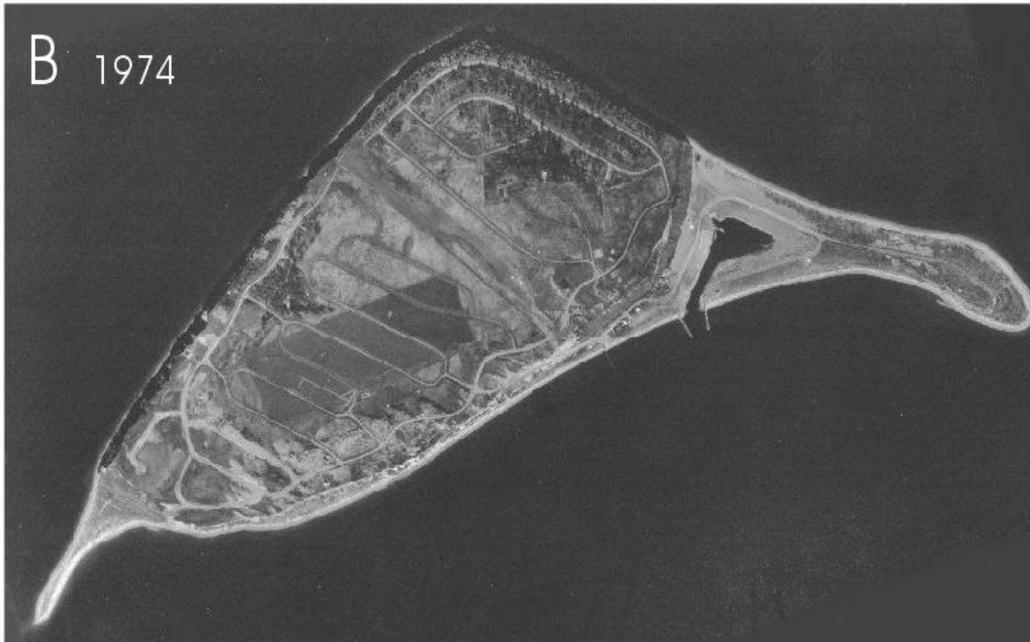
Source: Cowles and Hayward 2008

The extent to which these features were a direct result of overgrazing is unclear since PI experiences a tremendous amount of natural erosion. It has been estimated that sloughing of the bluffs and cliffs results in a net perimeter loss of about 15 cm per year (Larsen 1982). The Island is surrounded by a shallow area known as Dallas Bank which encompasses approximately 16.84 km². Dallas Bank extends north 5 km and south approximately 0.64 km (USFWS 1985) constituting the base of a once substantially larger Island (Larsen 1982).

The role ungulates played in the transformation of the plant community on PI is not well understood. In general, it is difficult to assess non-native species impacts

on native species because in most cases the data needed to compare conditions before and after introduction simply does not exist (Courchamp et al. 2003). That is certainly the case with PI. Assessing ungulate impacts on the Island is further complicated by development that substantially altered vegetation in the 1960's and 70's including clearing areas for roadways, cul de sacs, home sites and an airstrip (See *Figure V.2*, Historical Photograph B).

Figure V.2, B: 1974 Photograph of Protection Island



Source: Cowles and Hayward 2008

What is clear is that in 1999, only 41% of non-woody grassland species of sampled vascular plants were native to the Island. Intensely disturbed upland plateau areas exhibited significantly higher species richness when compared with less disturbed areas although that was due primarily to the presence of introduced species (Cowles and Hayward 2008). Many decades of farming and grazing has substantially altered the Island's ecosystem. Today the slopes described by Frank Richardson in 1961 are again experiencing grazing pressures. Where once there were cattle, sheep, and horses, a herd of Columbia black-tailed deer now grazes freely threatening the Island's recovery.

VI. Deer on Protection Island

Columbia Black-tailed Deer

Mature Columbia black-tailed bucks (males) and does (females) range in weight from 48 to 90 kg and 40 to 65 kg respectively. They browse in winter and early spring and add grazed grasses and a wide variety of herbs to their diet when they are available in summer and fall. Black-tailed deer species are extremely flexible and can exploit an array of habitats consuming a wide variety of forage (Gillingham 2002). Moreover, deer can have dramatic effects on plant biomass and community structure (Gaston et al. 2008) with particularly pronounced impacts in areas that lack predators such as islands (Gaston and Martin 2002).

Columbia black-tailed deer populations have tremendous potential for rapid increase under favorable conditions because they reach breeding maturity at an early age and reproduce annually, often producing more than one offspring (Gillingham 2008). Young does typically breed in their second year, although first year births are routinely observed, and usually give birth to a single fawn. Older does three to nine years of age in good condition usually give birth to twins and sometimes triplets. Does produce offspring throughout their lives and every year 90% of the female population gives birth (Hatter et al. 2000). Most deer live for no more than five years while a few live longer than ten years (WDFW 2011a). The primary causes of death are predation, starvation and hunting. Under good conditions their population can double in a few years (Hatter et al. 2000).

Black-tailed deer represent the highest harvest of any deer species in Washington State with an annual harvest of about 14,000 individuals (WDFW 2008). They are considered abundant with the Washington Natural Heritage Program's highest occurrence ranking of G5 and S5 (1 is very rare and 5 is common), indicating they are "demonstrably secure" globally (G) and within the state (S) respectively (WDNR 2009). In Coastal Region 6 which includes the Olympic Peninsula, the black-tailed deer population is increasing (WDFW 2009).

Deer Density Comparisons

Accurately estimating deer densities is difficult, and caution should be exercised when comparing density estimates from different habitat types. Based on estimates from similar habitats in other areas within the range of this deer species, at 53 deer / km² PI has a very high deer density. By comparison, black-tailed deer densities along the Columbia River typically range from 4 to 12 deer / km² (USFWS 2010b). Density estimates for Columbia black-tailed deer in forests adjacent to the Pacific Rim National Park on the southwest coast of Vancouver Island, B.C., range from 0.4 to 1.8 deer / km² (Engelstoft 2007) and Columbia black-tailed deer densities on the north Olympic Peninsula in a 1997 study ranged from 1.14 to 9.99 deer / km² (Ratti et al. 1999). Recent density estimates of Sitka black-tailed deer in the temporal coastal rainforest of southeast Alaska ranged from 7 to 12 / km² (Brinkman et al. 2011) (See *Table VI.A*). It is important to note that these areas have population limiting factors not found on PI including predation and hunting.

Table VI.A, Deer Density Comparisons

Deer Density Comparisons				
Protection Island	N Olympic Peninsula	Columbia River	SW Vancouver Island	SE Alaska
53.0 / km ²	1.14 - 9.99 / km ²	4 - 12 deer / km ²	0.4 - 1.8 deer / km ²	* 7 - 12 / km ²

* Sitka Black-tailed Deer

Deer Population Growth

A particular landscape has a theoretical maximum deer density that can be sustained over the long term based on forage availability in the absence of predation, hunting, and extreme weather mortality (de Calesta and Stout 1997), which is the case on PI. However, variables such as the amount of former agricultural acreage can directly influence that carrying capacity (K). Researchers have suggested that there is a direct relationship between the percentage of old agricultural fields and K for deer (Porter and Underwood 1997).

For example, K for white-tailed deer (*Odocoileus virginianus*), which can be as much as 30% larger than Columbia black-tailed deer (Hunting 2011b), in the contiguous oak forest of central Massachusetts where hunting is prohibited has been observed approaching 20 deer / km² (Healy 1997). In the George Reserve in central Michigan, where 26% of the acreage is old agricultural fields, K has been estimated at 38 deer / km² (McCullough 1979, 1984). In the Saratoga National Historical Park in New York, where 50% of the acreage is old agricultural fields, K has been estimated at 55-60 deer / km² (Underwood et al. 1994).

Farmed intermittently between 1856 and 1957 (Clark 1995, Cowles and Hayward 2008), PI has a legacy of old agricultural fields. Estimates from 1937-1941 (Cowles and Hayward 2008) are that 36% of PI's 136 ha was in tilled fields at that time. The model for white-tailed deer suggests that PI's K for that species would lie somewhere around 48 deer / km². Given that PI had an estimated additional 40 ha in productive grasslands at the time (Cowles and Hayward 2008), whereas the additional acreage in the cases mentioned above was in less nutritious forest land, and that white-tailed deer are generally significantly larger than Columbia black-tailed deer, we could expect PI's K to be greater. It is quite possible that the PI herd may not yet have reached K at the recently estimated 53 deer / km².

Considering the definition of carrying capacity (K) as the point at which the population is balanced between mortality and recruitment (McCullough 1984), we may be able to determine where the PI herd is heading if we can get an accurate count of the deceased and the new recruits. The caretaker reports from 2010 (Davis pers. corr. 2010) suggest death and recruitment may differ in magnitude; 15 new births (fawns) and 13 deaths, three of which were fawns, were reported during the fall and winter of 2009-2010. If those figures are correct, then 12 fawns (15-3) would have replaced 10 deceased adults resulting in a net increase. This is supported by the deer counts conducted in 2010 (71) and 2011 (78), which suggest a growth rate (R) of roughly 1.099 (78 / 71) or about 9%.

Carrying Capacity (K) and Freshwater

It is also possible that the removal of the last freshwater trough from the Island which occurred in 2009 will essentially lower K. PI has no reliable freshwater source although there are numerous leaks in the well water distribution system and the Island is often drenched in fog. Studies have shown that deer can survive about a month with little or no food, but animals have been known to die in as little as three days without water. Research has also shown that deer will lose weight and stop feeding with even a moderate restriction in water (Hunting 2011a). Deer get their water from three sources: free water, such as ponds, streams, and the dew on plants; preformed water, or that contained in plants; and metabolic water, which is produced during metabolism. Deer are believed to need about 3 to 6 quarts of water a day, depending on the outside temperature, although deer may not require any free water if lush forage is available (Hunting 2011a).

Island Invaders and Carrying Capacity (K)

Deer carrying capacity is a reflection of a landscape's maximum long term deer population potential and therefore a function of full forage utilization. However, K is difficult to determine and has limited value when assessing the potential impacts of invaders. In practicality, on islands free of the normal biotic pressures found in mainland habitats, such as predators, deer populations can expand at their maximum growth rates often overshooting K. Limited island resources are then depleted and a sudden population crash may occur (Courchamp et al. 2003). For example, after just 20 years, 29 reindeer introduced on St. Matthew Island increased to 6,000 individuals. Free of the normal biotic pressures they had evolved with on the mainland, the population rapidly increased beyond the landscape's capacity resulting in severe impoverishment of flora and a dramatic population crash (Klein 1968, Courchamp et al. 2003). The value of K was only apparent after the peak and subsequent crash, and was then of minimal value in predicting impacts.

Relative Deer Density

Perhaps a more useful tool in anticipating potential impacts is that of relative deer density (RDD) which is an expression of a deer population density as a percentage of a habitat's potential capacity, K . Because RDD is a function of K , it accounts for size difference between deer species. Researchers studying white-tailed deer population management in the Eastern U.S. hypothesize that impacts to resources should be similar at similar RDD, even on landscapes with widely varying K . Furthermore, if we can determine the RDD at which biological diversity is sustained, then, the deer population could be managed to achieve a variety of resource goals that explicitly include, biodiversity, timber harvests, harvest opportunities for deer for hunters, etc. (de Calesta and Stout 1997).

The formula for RDD is: $\text{Deer Density} / K \times 100$.

We can assume that the PI herd has not yet reached K given the Island's high quality forage (grasslands and old agricultural fields), the number of observed fawns in relation to observed dead deer, and the apparent increase from 71 to 78 individuals between 2010 and 2011. In order to explore the RDD concept assume that the herd is either at or below K . Calculating RDD using the most recent deer count, 78 in March of 2011, would yield a K of 53 deer / km^2 and a RDD of 100 ($\text{DD}:53 / \text{K}:53 \times 100$). Assuming the Island could support twice the current number of deer, 156 individuals, would yield a new K of 106 deer / km^2 and a RDD of 50 ($\text{DD}:53 / \text{K}:106 \times 100$) and therefore a range of $50 < \text{RDD} < 100$.

When compared to studies of white-tailed deer in eastern states which revealed that species richness and the abundance of tree seedlings and other sensitive flora and fauna decreased at $\text{RDD} > 16$ (de Calesta and Stout 1997), a range of $50 < \text{RDD} < 100$ is very high. Those studies suggest that an RDD of 16 or less is necessary to maintain such sensitive species. Researchers also found that at $16 < \text{RDD} < 32$, species richness and abundance of songbirds and their nesting and foraging habitat, as well as shrubs and herbaceous vegetation declined. At $\text{RDD} >$

32, hardwood species recruitment began to fail (de Calesta and Stout 1997). Such comparisons suggest that sustaining the current diversity levels on PI is not possible with the present deer density. In fact, an ecosystem's biodiversity can be reduced by deer at less than 25% of K (de Calesta and Stout 1997).

Observed Deer Impacts on Protection Island

Presently deer browse extensively throughout most of the island and have created an elaborate network of deeply eroded pathways through RHAU burrow habitat in the slopes surrounding much of the island. Such trails are devoid of vegetation and subject to increased erosion. Between 2005 and 2009 researchers observed evidence of significant slope failure possibly related to deer on two occasions. In one instance a section of slope approximately 300 square meters collapsed and destroyed a number of RHAU burrows (Hodum and Pearson pers. corr. 2009).

Extensive areas of compacted and eroding soil are present throughout RHAU burrow nesting habitat and burrows collapsed by deer hoofs have been observed by staff and researchers although no quantitative analysis has been published. In 2009, researchers observed two of 87 active research burrows completely collapsed and four additional burrows with damage to their entrance tunnels. The damage appeared consistent with hoof punctures. Deer can easily break through the soft surface soil with their narrow pointed hoofs (Hodum and Pearson pers. corr. 2009). Furthermore, photographs taken by researchers show deer tracks and holes in the soil covering RHAU burrows the size and shape of deer hoofs (See *Figure VI.2, Deer Use and Burrow Damage Photographs*).

Researchers regularly observe deer grazing in the high density RHAU colony and have commented that they may also be impacting forbs. Perennial forbs are a relatively minor component of the current grassland community but they may have been more abundant in the past. Areas around colonies are now dominated by non-native annual grasses that do not hold soil as well as deep-rooted forbs and native perennial grasses (Hodum and Pearson pers. corr. 2009).

Deer in general create areas of flattened vegetation 1.0 to 1.3 m long and 0.7 to 1.0 m wide where they bed down. They sleep in dense cover or tall grasses and may return to the same spot over many days. Since deer often travel in small groups, there may be several “deer beds” in the same vicinity (WDFW 2011a). Researchers have observed this phenomena on PI and noted that deer have bedded directly on top of burrows (Pearson and Hodum 2009). In addition to burrow damage, this activity may obstruct returning RHAU and has the potential to frighten birds causing them to lose food intended for chicks. It may also reduce vegetation cover exposing adults and chicks to increased predation and kleptoparasitism pressures, especially from opportunistic species such as gulls and eagles.

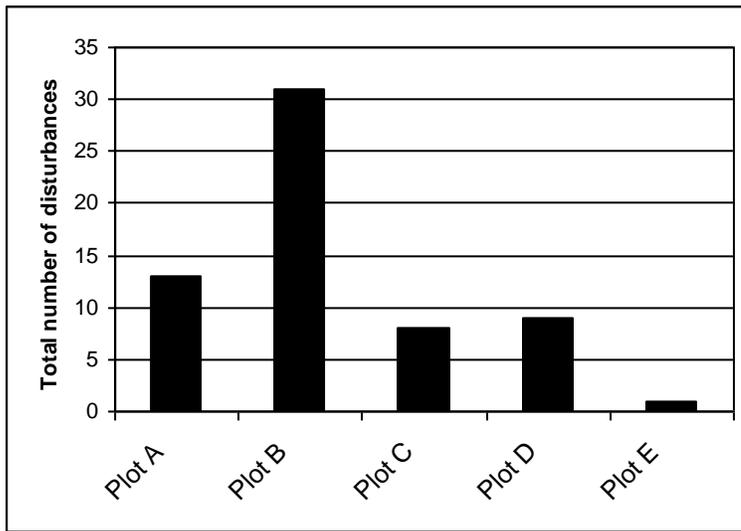
The first quantitative analysis of the direct impacts of deer on RHAU burrows on PI was conducted in 2010. In this study, 40 plots measuring 5 m² were established randomly throughout the RHAU breeding colony. Burrow damage occurred in 85% of the study plots. On average, 12.1% of the burrows within in each study plot had evidence of damage including structural damage to the tunnel and nest chamber (6.7% of burrows in the study area) and damage to the burrow entrances (6.9% of burrows in the study area). By measuring the length and width of deer trails, researchers were able to determine that deer trails covered an average of 37% of the surface area within each study plot. There was a significant correlation between burrow damage and the number of deer trails in the plot (Balbag and Hodum, unpublished). Researchers concluded that deer are using the RHAU colony extensively and are causing significant damage to RHAU burrows.

Deer Impacts to Gulls

In addition to observing deer in the RHAU colony, researchers have observed deer traversing the gull colony. The Island is home to the largest glaucous-winged gull colony in the Salish Sea despite a substantial recent decline which is likely due, in part, to eagle predation and disturbance (Hayward pers. corr. 2009). However, deer disturbance may be a factor since they regularly walk through and

browse within the colony. In 2006, researchers monitoring eagle disturbances in the gull colony also recorded the number of deer disturbances (See *Figure VI.1, Deer Disturbances Experienced by Protection Island Gulls*). They found that such disturbances were concentrated in the early morning, and to a lesser degree, in the evening hours which coincided with the highest level of gull activity (Hayward and Henson 2008).

Figure VI.1, Deer Disturbances Experienced by Protection Island Gulls



Number of black-tailed deer disturbances experienced by gulls in each of five study plots during 478 hours of observation during May, June, and July 2006.

Source: Hayward and Henson 2008

Because deer remain in an area for extended periods of time while foraging, disturbances span a longer time frame than those caused by eagles. Deer flush adult gulls and their chicks from their nests. The flushed birds run and fly around where they may be more prone to injury and even death. During deer disturbances eggs are exposed to damage by deer and other gulls, and to eagle and gull predation. Gull eggs and chicks are likely more vulnerable to injury and mortality when deer are present (Pearson and Hodum pers. corr. 2009). While there has been minimal quantification of the impacts of deer within the gull colony, the overall impacts on gull reproduction has not been measured. However, researchers report that the two experimental colony plots most frequently visited

by deer experienced the highest per capita chick mortality (Hayward and Henson 2008) and suspect their presence may be a contributing factor (Hayward pers. corr. 2009). Gulls can be extremely aggressive and are prone to kleptoparasitism of other birds. Deer disturbances in the RHAU and gull colonies create opportunities for gulls to steal food and attack RHAU and other gulls. Deer disturbances may also present additional opportunities for eagles and owls to prey on RHAU during the confusion.

The evidence from PI suggests deer are negatively impacting glaucous-winged gulls and RHAU. Deer are collapsing and damaging burrows, reducing protective vegetative cover, increasing erosion, and disturbing nesting birds which can lead to injury, food loss, increased predation, and abandonment of eggs and chicks. While our understanding would benefit from further study of the effects of deer on the Island's seabirds, there is mounting evidence that the growing deer herd negatively impacts nesting bird colonies on PI and it is likely that those impacts will increase as the deer density increases.

Future Island Restoration Efforts

According to the 2010 CCP, plans for PI include the following restoration goals;

- Control invasive plants and increase native plants
- Enhance vegetation characteristics on up to 20 acres of bluffs
- Restore up to 200 acres of savannah grassland
- Restore connectivity, crown closure, regeneration, and associated understory of 80 acres of woodland
- Monitor and control invasive plants and animals

The Island may also harbor historic seed banks that are a refuge for rare and endemic plant species that likely evolved in the absence of herbivores. Efforts to restore and enhance endemic plants and control invasive species will undoubtedly be compromised in the continued presence of deer. Their removal will likely be a necessary step in achieving any significant habitat restoration.

Figure VI.2, Deer Use and Burrow Damage on Protection Island



Source: Peter Hodum

Conclusion

Trophic webs on islands often have lower taxonomic and ecological redundancy than less isolated and more complex mainland ecosystems. As such, high densities of non-native species may more effectively facilitate imbalances within these simplified ecosystems (Courchamp et al. 2003) and are therefore considered a major threat to diversity and species richness on islands globally. In specific, non-native black-tailed deer have been linked to declines in species richness and diversity in island ecosystems (Gaston et al. 2008, Martin 2011) at substantially lower densities than found on Protection Island and are responsible for extensive damage to invaded insular ecosystems where indigenous species have evolved in their absence (Courchamp et al. 2003). Furthermore, non-native invaders including deer present a significant threat to seabirds (Donlan and Heneman 2007) which rely predominantly on islands for breeding habitat.

Deer on Protection Island

While Columbia black-tailed deer are native to the area, there are no historical accounts of their presence on PI. As such they can be considered non-native to the Island. Since their arrival in the early 1990's they have been observed feeding and resting within RHAU breeding habitat with increasing frequency as their numbers burgeon (Pearson and Hodum pers. corr. 2009). Recent counts estimate deer density to be 53 deer / km² which is very high compared to densities in most other areas within their range (e.g., north Olympic Peninsula habitats generally do not exceed 12 deer / km²) (USFWS 2010b, Engelstoft 2007, Ratti et al. 1999, Brinkman et al. 2011). Sitka black-tailed deer densities in the Queen Charlotte Islands, B.C., where biodiversity declines linked directly to deer are as severe as those seen almost anywhere in the world, range from 13 to 34 deer / km² (Martin and Baltzinger 2002, Gillingham 2002). Relative deer density comparisons with studies in the eastern U.S. suggest that sustaining existing species diversity and richness on PI is not possible with the current or even with substantially reduced deer densities (de Calesta and Stout 1997).

Black-tailed deer (genus *Odocoileus*) are generalists and able to exploit a diverse array of habitats. They can consume a wide variety of forage vegetation (Gillingham 2002) and can have a dramatic effect on plant biomass and community structure (Gaston et al. 2008). Their impacts can be particularly pronounced in areas that lack predator pressures such as islands (Gaston and Martin 2002), and they have tremendous potential for rapid population increase under favorable conditions given their high reproductive rate. Despite their current high density on PI, and the fact that the FWS has removed most artificial freshwater sources, it is possible that deer have not yet reached carrying capacity given the Island's lack of predation and hunting controls, mild climate, and high quality forage.

Deer Impacts on Protection Island

Recent unpublished research suggests deer are negatively impacting RHAU burrows on PI (Balbag and Hodum, unpublished). In addition to directly collapsing and damaging burrows deer have created an extensive network of deeply eroded pathways devoid of vegetation throughout seabird nesting habitat. Quantitative analysis concludes that deer trails cover 37% of the surface area of RHAU breeding habitat on PI and that deer damage about 12% of the burrows. There is a significant correlation between burrow damage and the number of deer trails in the colony (Balbag and Hodum, unpublished). Additionally, deer are either the direct cause or a contributing factor in slope failures that have destroyed RHAU burrows (Pearson and Hodum, pers. corr. 2009). Deer may also be exposing RHAU to increased predation pressure by bedding down on top of, or near borrows where they can obstruct and startle birds returning to feed their young.

Deer may be negatively impacting glaucous-winged gull reproductive success as well. Of five gull nest study plots monitored in a 2006 study, the two most frequented by deer had the highest per capita chick mortality. Researchers strongly suspect that deer presence may be a significant factor and suggest that

deer disturbance has a greater impact on glaucous-winged gull breeding success than eagle disturbance because deer remain in the colony for much longer periods of time (Hayward pers. corr. 2009).

Historically PI suffered extensive changes to vegetation and soils from farming, grazing of domestic animals, and development from which it has not fully recovered. In particular, sheep grazing is believed to have caused substantial erosion problems in RHAU habitat resulting in collapsed and buried burrows (Richardson 1961). Albon et al. (2007) demonstrated that at high densities (>40 individuals / km² for red deer) deer tend to form larger groups and range less causing impacts more closely resembling those of sheep herds. At 53 individuals / km² deer have the potential to exacerbate earlier damage and limit the Island's recovery.

Protection Island's Seabirds

Protection Island is an important seabird research station which is closed to public visitation to reduce wildlife disturbance. More than 70% of seabirds in Puget Sound and the Strait of Juan de Fuca nest on this single Island (USFWS 1985) which is home to one of the last tufted puffin colonies in the Salish Sea (WDFW 2011b, USFWS 2010b) and the largest glaucous-winged gull colony in Washington State (PTMSC 2011). It is also home to one of the most important RHAU colonies in North America as described herein. The importance of protecting PI's RHAU colony is elevated due to the colony's breeding success; unique location in the inland waters of the Salish Sea where it is buffered from ocean conditions and extreme weather events; and recent population declines elsewhere. Of the many threats PI's RHAU face, Refuge managers can effectively mitigate only two, human disturbance including pets and impacts from invaders.

Management Responsibilities

The Washington Department of Fish and Wildlife and the U.S. Fish and Wildlife Service collectively (hereafter the agencies) have spent in excess of \$5 million of

public monies to preserve important seabird habitat on PI. The effort to protect the nesting colonies included an expensive and lengthy legal battle that condemned private land holdings on the Island (USFWS 1987) and resulted in the establishment of the first non-game Washington State wildlife preserve and the creation of the only National Wildlife Refuge during President Reagan's administration (Palmer 2000).

Establishment of the Zella M. Schultz Seabird Sanctuary and the Protection Island National Wildlife Refuge was, in effect, a promise to forever preserve PI for the study and advancement of the Island's endemic wildlife, especially seabirds. As custodians, the agencies are mandated to maintain the Island in a condition conducive to continued seabird breeding activities, e.g. free from excessive disturbance including conflicting animal usage, and to entertain only such uses which are compatible with the primary over-riding purpose (Larsen 1982, USFWS 2010b).

While the FWS may have inadvertently neglected its custodial responsibilities by allowing, and even facilitating the establishment of deer on PI, the agencies are now obligated to mitigate the impacts of deer through active management. The FWS Final Environmental Assessment for PI from 1982 expresses the intended extent of the agencies responsibilities in regards to such dilemmas; "Management . . . is perceived to be primarily custodial, wherein natural processes would [will] be allowed to prevail. However, other management/habitat manipulation would [will] be considered on the basis of the needs of nesting seabirds." (Larsen 1982, pg. 19-20).

Implications for the Future

There is little doubt that the success of restoration plans will be compromised by deer at current and even substantially reduced levels. If attempts to restore native vegetation are to be successful, deer removal may be a particularly important step to take (Hayward pers. corr. 2009) as continued deer presence will undoubtedly

inhibit such efforts. As in most cases, the best option for mitigating the impacts of invading species is to regularly reduce their numbers, or if possible, remove them completely (Courchamp et al. 2003). It is possible that PI may harbor historic seed banks that serve as refuges for rare endemic plant species that likely evolved in the absence of herbivores. If so, continued deer presence would compromise propagation of those resources.

Lastly, increasing numbers of deer alone, or in combination with climate events that cause additional erosion, could further degrade seabird breeding habitat on the Island. RHAU face many threats both at sea where they feed and on land where they breed and rear their young. Of those many threats, the presence of deer on PI is perhaps the least expensive threat to address and one that has a high probability of success. Black-tailed deer are harvested more than any deer species in Washington State and are considered demonstrably secure within the state and globally. I conclude that removing them is likely to yield the highest single return on investment in regards to any management action currently under consideration by the agencies for protecting seabird breeding habitat and restoring Island flora. The expense and difficulty of removing deer may increase as the herd expands.

However, deer may have an important scientific role to play. Studying and understanding their impacts on species richness and diversity, especially their impacts on seabirds, and subsequently evaluating native plant and wildlife responses to their removal could satisfy both the Sanctuary's, and the Refuge's scientific purposes. Fully assessing the management action through monitoring would maximize return on the investment and may contribute information that proves valuable in restoring biodiversity in other island systems.

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Appendix

Protection Island National Wildlife Refuge Enabling Legislation

Protection Island NWR: The purposes of the refuge are to provide habitat for a broad diversity of bird species, with particular emphasis on protecting the nesting habitat of the bald eagle, tufted puffin, rhinoceros auklet, pigeon guillemot, and pelagic cormorant; to protect the hauling-out area of harbor seals; and to provide for scientific research and wildlife-oriented public education and interpretation. 96 Stat. 1623, dated Oct. 15, 1982. "... to conserve (A) fish or wildlife which are listed as endangered species or threatened species or (B) plants ..." 16 U.S.C. § 1534 (Endangered Species Act of 1973) "... for the development, advancement, management, conservation, and protection of fish and wildlife resources ..." 16 U.S.C. § 742f(a)(4) "... for the benefit of the United States Fish and Wildlife Service, in performing its activities and services. Such acceptance may be subject to the terms of any restrictive or affirmative covenant, or condition of servitude ..." 16 U.S.C. § 742f(b)(1) (Fish and Wildlife Act of 1956) "... for use as an inviolate sanctuary, or for any other management purpose, for migratory birds." 16 U.S.C. § 715d (Migratory Bird Conservation Act) (NWRS 2011)

Source:

National Wildlife Refuge System website

Refuge Station Purposes - Single Unit Search Results

http://www.fws.gov/refuges/policiesandbudget/purposes/Unit_Display.cfm