

**PROPOSAL FOR INCLUSION OF SPECIES ON THE APPENDICES OF THE CONVENTION ON
THE CONSERVATION OF MIGRATORY SPECIES OF WILD ANIMALS**

A. PROPOSAL: Inclusion of *Diomedea nigripes* in Appendix II

B. PROPONENT: Government of the Netherlands

C. SUPPORTING STATEMENT

1. Taxonomy

- | | |
|---------------------|--------------------------|
| 1.1. Class | Aves |
| 1.2. Order | Procellariiformes |
| 1.3. Family | Diomedeidae |
| 1.4. Genus/Species | <i>Diomedea nigripes</i> |
| 1.5. Common Name(s) | Black - footed Albatross |

2. Biological Data

2.1. Distribution (current and historical)

Restricted to North Pacific. Most breeding pairs on North western Hawaiian Islands, mainly on Laysan I., Midway Atoll. Pearl and Hermes Reef, smaller colonies on French Frigate Shoals, Lisianski I., Kure I., Necker I., Nihoa I. Previously extirpated from Wake and Marcus Is., Torishima, Johnston Atoll and Taongi Atoll (Marshall Is). Moko Jima and Iwo Juna (Bonin Is). In Japan currently breed in small colonies in The Senkakus, in the Bonins, and on Torishima in the Izu Islands (Rice and Kenyon 1962a. Harrison 1990, McDermond *et al* 1993).

The marine distribution of *D. nigripes* has recently been summarised by McDermond *et al* (1993) and the following information comes from this source and references therein. Generally, *D. nigripes* is considered to be an offshore species and the pelagic range of breeding birds is most restricted in February, when *adults* are making the shortest and most frequent foraging trips to feed chicks (McDermond *et al.* 1993). After breeding, birds disperse widely over the North Pacific, reaching as far north as the Bering Sea. Most fledglings fly northwest towards Japan, whereas adults disperse towards the west coast of North America, moving northwards during the non breeding season. There appear to be differences in marine distribution with age of juveniles. Robbins and Rice (1974) suggested that in their first summer and autumn after fledging most juveniles favour the waters west of 180°, whereas they spend subsequent summers and winters in the eastern half of the Pacific. Most adult *D. nigripes* spend winter in the Central Pacific, an area where few first winter birds are observed (Robbins and Rice 1974).

D. nigripes are reportedly more abundant over the outer continental shelf, and in areas with strong persistent upwelling. Concentrations over continental slope areas *have* been attributed, at least in part, to be a reflection of the distribution of fishing vessels. Wahl and Heineman (1979) suggested that the distribution of fishing boats exerts an influence on the distribution of *D. nigripes* over a considerable area.

2.2. Population

Annual breeding pairs - 43,160

During the early part of the Twentieth Century *D. nigripes* populations plummeted due to mass slaughter by feather hunters and on many islands populations were completely eliminated (Fisher 1949, Harrison *et al.* 1984). Populations also suffered as a result of habitat degradation

after the introduction of rabbits on Laysan I. and Lisianski I. (Harrison 1990). The most recent estimates of the total number of *D. nigripes* nesting in Hawaii were based on surveys conducted between 1979 - 1982 when the breeding population was estimated at 50,000 pairs, the largest population being 14,000 - 91,000 pairs on Laysan I. (Fefer *et al* 1984 in McDermond *et al.* 1993; Harrison 1990; Harrison *et al.* 1984). The population trends of *D. nigripes* breeding on Laysan I., Midway Atoll and French Frigate Shoals, which together represent 67% of the population, have recently been summarised by McDermond *et al* (1993). On Laysan I. the population increased from ca 7 700 pairs in 1912 - 13 to a peak of 33 500 pairs in 1957 - 58, the latest estimate being 21 000 in 1919 - 80 (McDermond *et al.* 1993 and references therein). It is not known whether this apparent decline is real or a reflection of annual variability in number of nesting birds, amplified by differences in timing of sampling and methodologies used (McDermond *et al.* 1993). The population throughout the French Frigate Shoals had been recovering and reached a peak of ca. 6,200 in 1987. The latest estimate in 1990 - 91 showed a reduced population, with only 3,900 pairs (McDermond *et al* 1993). Over the last 4 years this breeding population has decreased by an average of 12.8% per year (Gould and Hobbs 1992). On Midway Atoll, the US military killed at least 2,500 *D. nigripes* between the mid 1940s and 1950s, and the population numbers have remained relatively stable since the late 1950s and the last 1981 - 82 survey. Sporadic counts of a study colony on Midway Atoll between 1960 - 90 showed that the average geometric growth for this population was +9.4% (Gould and Hobbs 1992). However, due to variations in census techniques and fluctuations in the annual breeding population, McDermond *et al* (1993) caution that much of the perceived population growth may be an artefact. This, together with the apparent declines at some colonies, may be signalling a decline in numbers throughout the Hawaiian range. Insufficient data, however, preclude an assessment of the current status of the *D. nigripes* population in Hawaii (McDennond *et al* 1993, see Table 4).

In Japan, small breeding populations are present at least three sites, the largest being on Tonshima where Hasegawa (1984) estimated that ca. 500 pairs visit the colony. This colony is apparently increasing, and they are expanding their nesting areas into the areas used by *D. albatrus* (Hasegawa and DeGrange 1982, Hasegawa 1984).

2.3. Habitat (Breeding habitat and biology)

Annual, monogamous synchronous breeder. Mainly nest in open, exposed coasts with nests consisting of shallow, cup-like depressions in sand. There are few published accounts of the demographic and breeding parameters of *D. nigripes* and most of the following information comes from Rice and Kenyon (1962 a,b) and summaries by Harrison (1990) and McDermond *et al.* (1993).

The first breeding adults arrive at the colonies in mid - late October with females arriving about 5 days later than males. Immatures return between January and March. The eggs (1 egg clutch) are laid in November (mean laying date 21 November, range 8 - 30 November) and both parents share the 65 day incubation period (Rice and Kenyon 1962). There are on average 6 incubation bouts, the first of which lasts 18 days and the duration of bouts progressively decreases to the final bout which lasts 7 days (Rice and Kenyon 1962b). *D. nigripes* chicks begin to hatch in late January (mean hatching date 25 January, range 15 Jan - 7 Feb) and hatching success varies from year to year, usually between 60 - 80% of eggs laid survive to hatch (Harrison 1990). Chicks are continuously guarded for 19 days (range 12 - 25 days) after which guarding is intermittent until 30 days (range 21 - 42 days). Both parents alternate in guarding the chick, with bouts lasting 2 - 3 days (Rice and Kenyon 1962b) Chicks are fed until they fledge, which usually occurs 140 days after hatching. A high proportion of chicks that hatch survive to fledge although catastrophic weather related events may cause high rates of nesting failures (Harrison 1990). For example, high surf drowned nearly half of the chicks at Laysan I. in 1958 and hundreds of nests at the French Frigate Shoals in 1982 (Harrison 1990). Breeding success (eggs laid to chicks

fledged) on Sand and Eastern islands varied from 42 - 67% in the 1950s (Rice 1959 in Rice and Kenyon 1962a). Failed and non-breeders begin to depart the islands in April and most *D. nigripes* have fledged by mid June. By the end of July, all *D. nigripes* have departed the colonies.

Philopatry is high with most birds returning to their natal atoll when they are 3 years old (Rice and Kenyon 1962b). Sexual maturity is attained usually at 7 - 10 years of age. Once a breeding territory (nest site) has been selected it is retained for life (Rice and Kenyon 1962b). No data regarding recruitment survival rates or breeding frequency for *D. nigripes* is available. The number of birds nesting in any one year is highly variable, annual variations at one island ranging between 1 - 55% (Gould and Hobbs 1992). The reasons for this high variability are not known.

2.4 Migratory patterns

See Distribution

3. Threat data

Despite protection at the major breeding sites (designated as National Wildlife Refuges), *D. nigripes* has not yet recovered to the levels prior to their persecution. Further, some of the major populations are apparently declining. Given the available information, it appears that fisheries related mortality is the most serious threat facing *D. nigripes* populations across their range. Although Harrison (1990) acknowledged the mortality associated with longlining and driftnetting he considered that it did not yet pose a major threat to Hawaiian albatross populations, but recommended that developments in fisheries should be monitored. It is estimated that driftnetting killed 2.2% of the population per year (Gould and Hobbs 1992), and there are no data of the extent of mortality associated with longlining.

3.1. Direct threat to the population

The most dramatic interaction between *D. nigripes* and fisheries which has been documented is the mortality of these birds caught in drift nets, particularly the Japanese and Korean squid fisheries. World wide, driftnet fisheries have been shown to be responsible for the deaths of hundreds of thousands of seabirds every year but the capture rates of birds in these fisheries are not well recorded except for the North Pacific salmon and squid fisheries (Northridge 1991). Since 1981, the equivalent of 1.5 million kilometres of drip gillnet was set for squid each year near the North Pacific subtropical convergence (Harrison 1990). It was estimated that 5300 *D. nigripes* were killed in 1989 by the North Pacific squid driftnetting fishery alone (Northridge 1991). In 1990, an estimated 4246 *D. nigripes* were killed by driftnets in the North Pacific High Seas drip net fisheries, 49% (n=2093) being taken by the Japanese squid fishery, 27% (n=1146) by the Korean squid fishery, 16% (n=685) by the Japanese large mesh fishery and 8% (n=322) taken by the Taiwanese squid and large mesh fishery combined (Johnson *et al.* 1992). Trying to assess the factors determining the rates of bycatch is complex although some of the variability in the data was consistent with the seasonal changes in distribution of the birds (Johnson *et al.* 1999). Although significantly more Laysan albatross (*D. immutabilis*) were caught by these fisheries (mean ratio of 3.7 *D. immutabilis* to every *D. nigripes* caught in the high seas drift nets in 1989 and 1990) this ratio is much lower than the ratio of 12.8 to 1 for the total estimated population sizes (Gould and Hobbs 1992). As a percentage of the total population, the incidental take of *D. nigripes* was estimated to have been in the order of 2.2% per year (Gould and Hobbs 1992). The estimated driftnet mortality was therefore likely to impact more on *D. nigripes* given their smaller population size, slower rates of demonstrated population recovery and their tendency to be more interactive with the fisheries (Gould and Hobbs 1992, McDermond *et al.* 1993).

Driftnets which capture and hold both fish and squid would allow the provision of a food source available to the albatross. There are many reported instances of *D. Nigripes* feeding on driftnets but there are no data by which to assess the degree of contribution of scavenged food to the diet. It is also impossible to assess the effects of other consequences of, for example, reducing numbers of other species which prey of the same species as the albatross, or reducing the abundance of albatross prey. Given these factors, and the fact that the population structure and dynamics of *D. nigripes* are poorly known, the effects of driftnet fisheries on *D. nigripes* populations cannot be assessed. In 1989 the United Nations General Assembly adopted a resolution calling for an end to drift netting on the high seas by mid - 1992, unless fishing nations could prove that the practice was not harmful. Japan was also to have phased out all drift netting by the end of that year (Northridge 1990, McDermond *et al.* 1993). On 31 December 1992 the UN moratorium was signed, and in theory, driftnetting operations in the areas were halted. However, some nations, including, Taiwan, continue to driftnet and the magnitude of the bycatch is unknown.

The principal source of band returns analysed by Robbers and Rice (1974) was via the longline tuna industry. *D. nigripes* are caught by the Japanese tuna longline fishery operating north of Hawaii (Harrison 1990) although there have been no reports of actual catch rates. Globally, the Japanese tuna fleets are most dependent on the Pacific region (Bergin and Harvard 1991) and it is in this area that some Japanese fishermen have commented to N. P. Brothers (pers. comm.) that seabird catch rates are highest. The local Hawaiian longline fleet report that "hundreds" of albatrosses are killed by this fishery, both as a result of birds taking baited hooks during line setting and by intentional killing to prevent loss of gear and reduced catch (McDerrmond *et al.* 1993). The Hawaiian fleet has increased threefold since 1987 and during the first half of 1991, more than 7 million hooks were set by vessels operating out of Hawaii (McDermond *et al.* 1993). Recently imposed regulations which control the number of longline vessels operating in the Hawaiian fleet and prohibit longline fishing within 80 km of the North western Hawaiian Islands (McDermond *et al.* 1993) may serve to reduce the extent of mortality, at least during the breeding season, but there are no reports of seabird catch rates prior to the regulations. Given this, it is not possible to assess the magnitude of the impact of longlining on *D. nigripes* either in the past, or at present.

One other source of potential interaction is that both longline and pole-and-line fisheries require supplies of good baitfish. In Hawaii several species of locally occurring surface schooling fish are used as bait at least for the pole-and-line fishery, and it is this type of fish which are also available as prey to the surface seizing albatross.

3.2. Habitat destruction

Late introduction of rabbits to Laysan and Lisianski earlier in the century probably contributed to declines in the population through habitat erosion and devegetation increasing storm related mortality due to sandstones (Harrison *et al.* 1984).

3.3. Indirect threat

Oil, plastics and agricultural pesticides from boats and coastal activities occur in the waters around Hawaii and impact on the resident albatross. Oiled birds, including albatross, have been observed and these birds often die as a result of stress and lack of thermoregulatory control (Harrison 1990). The visceral fats of Hawaiian albatross have been found to contain appreciable levels of DDT, DDE and PCBs but as yet, there has been no significant reduction of egg shell thickness measured (Harrison 1990).

The ingestion of plastics obtained as marine debris represents an area of concern for the

albatrosses in Hawaii. Flying fish ova, which contribute 44% (by mass) to the diet of *D. nigripes*. are available to the albatross as they float on the surface, often attached to flotsam (Harrison *et al* 1983, Harrison 1990). Incidence of plastic in this species is among the highest of any seabird and ingestion of large volumes results in dehydration and reduced growth and fledging rates (McDermond *et al.* 1993 and references therein). Most of the plastic debris ingested by these birds originates from Japan (Harrison 1990).

On Midway I., introduced mosquitoes have been implicated as vectors of an avian pox virus to the birds (Harrison 1990). On Kure I. it has been reported that Polynesian rats (*Rats exulans*) occasionally attack incubating albatross and also take the eggs (Moors and Atkinson 1984) Sharks also prey on fledglings as they enter the water for the first time. Floods after heavy winter rainfall, high surf associated with storms and heat stress may also cause loss of eggs and dead of chicks on the atoll breeding locations (Harrison 1990). Airstrikes also cause the death of albatross in Hawaii, *D. nigripes* to a lesser extent than *D. immutabilis*, and there have been many attempts to discourage the birds using these airfield area (McDermond *et al* 1993). On the Japanese island of Torishima the resident *D. nigripes* are faced with same threats as the co-occurring *D. albatrus*, the worst of which is habitat vulnerability in that the island is an active volcano.

3.4. Threat connected especially with migrations

4. **Protection status and needs**

4.1. National protection status

4.2. International protection status

4.3. Additional protection needs

The impact of rats and other introduced predators should be investigated and measures should be instigated if they are shown to be significant predators of eggs and chicks. The extent of interactions with fishing practices must be addressed both in local waters and on the high seas. The only way to establish the nature of the bycatch associated with any fishery is to have observer coverage on a wide scale basis, over both time and space. Once the nature of the problem is identified, then measures can be instigated which would eliminate, or at least significantly reduce the catch. These measures should not be counter productive to the target fishery or they will not be adopted on the high seas. This direction of management would require cooperation and collaboration between the different federal agencies responsible for effective management of both seabirds and fisheries.

5. **Range states**

Japan
USA

6. **Comments from Range States**

7. **Additional remarks**

8. **References**

See Reference at the very end of this document (pp. 182-187).