

The effects of the introduced mink and habitat disturbance on ground-nesting songbirds
in the Cape Horn Biosphere Reserve, Chile

Brett Maley^{1,2*}, Christopher B. Anderson^{1,2} and Amy D. Rosemond¹

¹ Institute of Ecology, University of Georgia, Athens Georgia, USA

² Omora Ethnobotanical Park, Cape Horn Biosphere Reserve, Chile

* bmaley@uga.edu

To be submitted to the *Journal of Field Ornithology*

Key words

Disturbance, exotic, invasive species, *Mustela vison*, Navarino Island, Tierra del Fuego

Abstract

Invasive species are one of the greatest threats to global biodiversity. The American mink, (*Mustela vison*, Schreber) was introduced to Tierra del Fuego Island (southern Chile) to begin a fur trade in the 1940's. The fur industry was not successful, and now the predatory mink has invaded a large part of the Fuegian/Cape Horn Archipelago. The mink has only been present on Navarino Island, where our study was conducted, for approximately three years. We expected mink to have a significant negative impact on ground-nesting songbird nests via egg predation, because mink are known to be predators of such nests and they lack an evolutionary history with birds in this area. To test this hypothesis, we used artificial nests and eggs and the Mayfield method (Hensler and Nichols 1981) to quantify nest predation in four different habitats: anthropogenically impacted shrublands, beaver meadows, secondary forest, and primary forest. Predators were identified by the teeth marks in the plasticine eggs. The mink was found to be the most frequent nest predator, with birds being the other major source of attack on artificial eggs. Mink depredation events were more frequent in disturbed habitats, i.e. shrublands and beaver meadows than in less disturbed habitats. While artificial nests may overestimate the importance of small avian predators, this study showed that mink can be a major threat to ground nesting songbirds in the Cape Horn area. Further, the greater effect of mink in more disturbed habitats suggests that conservation and management efforts may need to be prioritized in those areas.

Introduction

Invasive species are recognized as one of the greatest threats to global biodiversity (Mack et al. 2000). While humans have always introduced species both purposefully and accidentally, the rate and threat associated with modern invasive exotic species has become a global problem since humans have become capable of frequent and massive intercontinental travel. Therefore, the study of invasive species is a pressing issue for ecology and conservation as biological invasions increase around the world (Vitousek et al. 1997).

The consequences of invasion are variable, but it appears that islands are particularly vulnerable to the deleterious effects of introduced species because the inhabitants often evolved in isolation. For example, over the last 200 years, many of the avian extinctions that have occurred have been the result of species introductions to island habitats (Savidge 1987). Invasive predators can be especially detrimental on islands. Classic examples include accidental and deliberate predator introductions to Hawaii, Guam, and New Zealand, which caused very large impacts on native avifauna (Levy 2003).

Hawaii provides a particularly well-documented case study of an island with a history of introduced predatory species. The Polynesian rat (*Rattus exaltnus* Peale), for example, was introduced 1000-1500 years ago, and other rodents colonized after European contact in 1778 (Tameran and Malecha 1972). Then, the mongoose was introduced to Hawaii from 1883 onwards with the intention of controlling rats in sugar-cane fields (www.issg.org). Both of these species then also became important consumers of birds and bird eggs, and they have since been devastating to bird populations of

Hawaii, where since 1893, twenty-nine species of birds have become extinct (Levy 2003).

Other islands show the same patterns. Guam has also lost many avian species due to the introduction of the brown tree snake (*Boiga irregularis* Merrem), whose appearance was first documented during World War II from U.S. naval ships. The brown tree snake is responsible for the loss of ten avian species since its introduction (Conry 1988), and the southern part of the island is now characterized as an “avian desert” (Savidge 1987). New Zealand was also plagued by the introduction of the brown rat (*Rattus norvegicus* Berkenhout), brought to the island by the Maori when they first colonized the island. In addition, New Zealand saw as many as fifty mammals introduced to its islands since the late 18th century, causing the extinction of sixteen to twenty-four species of moas, a flightless endemic bird (Veblen and Stewart 1982).

Some predators have been introduced in multiple places around the world due to their commercial or sport value. For example, American mink (*Mustela vison* Schreber) have been introduced globally since the 1920's for the fur trade. They currently inhabit Sweden, Norway, Denmark, Iceland, Great Britain, Ireland, Finland, parts of Russia, throughout Europe, and now Chile and Argentina (Long 2003). Mink are affecting each of these areas, often causing problems for native biota. For instance, the mink has devastated bird populations in Iceland, in Sweden they are endangering fish populations, and in England they have reduced the numbers of water vole (Long 2003). The mink is now also a resident of Navarino Island in the Cape Horn Archipelago, Chile, but no studies have evaluated its impacts in this insular ecosystem.

The American mink was introduced to Tierra del Fuego Island to begin a fur trade in the 1940's. The fur industry was unsuccessful, and now the mink inhabits several islands on the archipelago (Rozzi and Sherriffs 2004). The mink is a predator that feeds on muskrats, mice, hares, rabbits, birds, bird eggs, insects, amphibians, reptiles, and mollusks. They are semi aquatic, which gives them the potential to disperse well in an archipelago. With regards to the austral archipelago, we expected that the mink may be detrimental to the other fauna because several of the islands in the chain evolved without terrestrial predators (Venegas and Sielfeld 1999). As a result, some of the songbird species nest on the ground, including the austral thrush (*Turdus falklandii* Quoy and Gaimard) and the rufous-collared sparrow (*Zonotrichia capensis* Meyen) (S.M. McGehee pers. comm.) Because so many Chilean forest birds are endemics (Vuilleumier 1985, Fjeldsa and Krabbe 1990), they are a priority for conservation (Willson et al. 2001). Now that the mink has invaded islands such as Navarino that previously had no terrestrial predator, ground-nesting songbird populations may be negatively affected through impacts on reproductive success.

Another factor that can influence reproductive success of birds and also the impact of introduced species is habitat disturbance. Habitat fragmentation increases the perimeter to area ratio, which in turn has been shown to increase predation and brood parasitism for forest dwelling species (Gates and Gysel 1978). Another project conducted on Chiloe Island, Chile (41° 55 S, 73° 35 W) showed that habitat loss and modification directly caused loss of nest sites and could lead to the loss of safe nesting sites through associated increases in the risk of nest predation (Willson et al. 2001).

In order to gain a better understanding of the effect the introduced mink has on forest birds of Navarino Island, we conducted an artificial nest study along with monitoring real ground-nesting songbirds in four habitat types with differing levels of disturbance. Minks are found at higher densities along the coast than along rivers (Anderson et al. 2006), and consequently we expected to find higher depredation rates associated with coastal habitats. We further predicted that mink may more heavily influence nesting success of songbirds in areas of greater disturbance, as disturbance has been shown to decrease safe nesting sites (Willson et al. 2001). To test these questions we measured nest depredation rates in primary forest, secondary forest, beaver meadows and anthropogenic shrublands at sites near and far from the coast. We also compared our results with naturally occurring nests that were monitored in combination with this study.

Methods

Study site

The study took place on Navarino Island in the Cape Horn Biosphere Reserve (55° S) Chile. The eco-region is known as the Magellanic Sub-Antarctic Forest Biome, and it is one of the few pristine wilderness areas left in the world (Mittermeier *et al.* 2001). The temperate forests of southern South America are characterized by having low species richness but a relatively high degree of endemism, when compared to Northern hemisphere temperate forests (Armesto et al. 1995, Veblen et al. 1996). While the archipelago is in many ways unimpacted by humans, it is also replete with exotic species. The exotic assemblage on the island includes the American mink (*M. vison*), as well as other exotic mammals, such as North American beavers (*Castor canadensis* Kuhl),

muskrats (*Ondatra zibethicus* Linnaeus), feral dogs, cats and livestock (Anderson et al. in press). Introduced species of birds, such as the house sparrow (*Passer domesticus* Linnaeus), pigeon (*Columbia livia* Gmelin) and trout are also found on Navarino Island (Anderson et al. in press).

Artificial nest study

To study the impact the mink has on ground-nesting songbirds we used artificial nests with plasticine eggs to measure the amount of predation events occurring on the nests.

The artificial nests used in the study were made out of 100 % dehydrated and sterilized coconut fiber, and the artificial eggs were made using a mix of white and blue colored plasticine clay to mimic the color of natural austral thrush eggs. Nests used were approximately 120 mm wide and artificial egg size was approximately 30mm to mimic natural conditions (S.M. McGehee, field ornithologist, Omora Park, pers. comm.). To standardize the clutch size, three eggs were placed in each nest. The study was done from October through December 2005, which corresponds to the austral spring.

Sixty nests were made and placed in four different habitat types (15 nests per habitat type): anthropogenic shrubland, beaver meadow, secondary forest, and primary forest. The nests were spaced at appropriate sites (i.e. under logs and vegetation) along a transect at approximately 10 meter intervals. The nests were checked every three days, and we wore nitrile gloves when handling the nests and eggs. The nests were considered depredated if they had scratch marks on the eggs, eggs were missing, had puncture holes, or eggs were torn apart.

The anthropogenic shrubland habitat included disturbed areas off of roads, the first site was a coast area 10 meters from the Beagle Channel and the second area was off of the road 1 km away from the ocean. The nests were placed along the roads no more than 2 meters from the road. The beaver impacted habitat included two separate beaver ponds, the “near” pond was 1 km away from the Beagle Channel and the “far” pond was 2 km away from the Beagle Channel. For the secondary forest habitat only one site was used. The primary forest habitat included two sites, one site away from the river and the other site along the river.

Characterization of study plots

To estimate potential small mammalian nest predators in the study sites other than mink, Sherman traps were set for two consecutive days after remaining artificial nests were collected. To account for the amount of vegetative cover surrounding the artificial nests, we split the area around the nest into four quadrants, and scored the amount of vegetation 1 meter surrounding the nest by percentage: 25, 50, 75, and 100% (Figure 2). Each of the percents were scored on a scale of 1 to 4, 1 being 25% and 4 being 100% vegetative cover. The average was then taken from each meter transect, to quantify vegetation surrounding the nest. The nests were positioned in the different habitats on October 3, 2005, checked every three days and collected on October 30, 2005.

Naturally occurring nests

Real nests were also sought during the study and monitored every three days after discovery for predation, fledging, or abandonment. Nests were considered ground-nests

if they were less than 1 meter off of the ground. Species included in the study were: Patagonia sierra finch (*Phrygilus patagonicus* Lowe), austral thrush (*Turdus falklandii*), and rufous-collared sparrow (*Zonotrichia capensis*).

Results

Depredation rates of artificial nests

A total of 66% of the artificial nests were depredated during the study. Only mink and small song birds were found to be preying on artificial nests. Based on beak imprints done of songbirds while mist-netting, the nests disturbed by birds in this study appeared to be the result of pecking by wrens (*Troglodytes aedon* Vieillot) (see Discussion for other evidence of this occurrence).

Of the total nest depredated, 68% were attacked by birds and 32% were bitten by mink. Based on these results, mink were having an important impact on ground nests in the Cape Horn Biosphere Reserve.

Habitat effects on depredation rates

We found a trend of habitat type affecting daily survival rates of artificial nests (Figure 1). The daily survival rates showed an increasing trend of survival from anthropogenic shrublands to beaver meadows to secondary forests and finally the highest DSR was primary forest ($p=0.02$).

Overall, we observed that the anthropogenic shrubland site on the coast had the highest nest failure rate (Table 1) of the nests failed. In the beaver meadow a total of 60% of the nest failures were due to a bird, making it the highest percentage of bird

depredations of all the habitats. In the beaver meadow habitat type, two of eight (25%) nests in the “far” beaver meadow (2 km from the coast) were depredated by the mink. All of the nests in the “near” beaver meadow (1 km from the coast) were attacked by a bird. In the primary forest habitat the lowest proportion (two out of fifteen) nests were depredated by the mink, and an equivalent number of nests were attacked by a small bird.

The nests that were placed in more disturbed habitat were also depredated more than the nests placed in secondary and primary forest. We determined that this habitat effect was not simply a function of changes in vegetation cover, however. While days observed did show a significant relationship with cover (Figure 2), percent vegetation cover itself was not related to habitat type ($F_{3,56}=0.53$, $p=0.67$). So, while vegetation cover may influence success of a particular nest, it was not influencing the outcome of the effect of habitat on overall survival rates.

Fate of naturally occurring nests

A total of seven real nests were found during the study and monitored until they fledged or failed. A total of 29% of the nests found fledged, and 71% failed. During the austral spring two nests failed due to inclement weather. One nest, an austral thrush, was found in the primary forest, and failed due to a heavy snow during the austral spring. Another nest of a Patagonian sierra finch was found along the side of the road and failed due to heavy rains. The other nest, including an austral thrush and two Patagonian sierra finches, failed due to unknown reasons, but did not appear to be depredated due to the appearance of the nest. During the austral spring two nests fledged; both were found in secondary forest.

Discussion

Invasive species have caused the extinction of native bird populations on other insular systems, such as Hawaii, Guam and New Zealand. It is apparent in our artificial nest study that the mink has the capability to likewise devastate native bird populations on Navarino Island, Chile. The disturbed sites in this study were most impacted by a nest predator, which means that humans are not only impacting the flora but also ecosystems as a whole making it easier for the mink to depredate nests. The beaver impacted meadows were depredated the most after the human impacted areas, mostly by other birds.

It is unknown if the predation on the nests by other birds was incidental or purposeful, but previous studies have shown wrens depredating nests in North America and that the behavior might be a common feature of the wren family (Picman 1980). The marks on the eggs were similar to the marks made by the house wren (*Troglodytes aedon* Vieillot). Other studies have shown that small birds can be significant nest predators, such as the long-billed marsh wren (*Cistothorus palustris* Wilson), which heavily impacts marsh-dwelling passerines in North America (Picman 1980). We found that nests attacked by wrens had puncture marks on either some eggs or all eggs, but there were no signs of an attempt to break into the egg and eat it, like the marks the mink left on the eggs. The same result was found in the study on cactus wrens (*Campylorhynchus brunneicapillus* Lafresnaye) in the US Southwest, they did not consume the contents of the quail eggs so the destruction may represent intra or interspecific competition for nest sites (Kendeigh 1941, Belles-Isla and Picman 1986, Simons 1990). A wren is acting as a

nest predator to the same extent as the mink is, with the wren being the cause of 24 nest failures and the mink causing 16 nest failures.

Inclement weather was found to have an effect on nesting success in natural nests. Nests that were built early in the breeding season (austral spring) had a much higher percent of failure than nests built later in the austral spring, habitat also played a role in the success of the nests with more nests fledging in secondary forest rather than anthropogenically impacted areas. While year round weather patterns in the sub-Antarctic ecoregion are at times adverse, the weather during the austral spring has a greater chance of inclement conditions, such as the late snowfalls and cold rain storms that caused several nest failures in the study.

Despite their potential problems (Faaborg 2003), artificial nest experiments remain a useful tool for evaluating the potential risk of nest predation (Willson et al. 2001). Since real nests are difficult to find and nests are usually not distributed across habitats and densities that satisfy a study, artificial nest studies were developed and are of use in determining relative aspects of such important ecological aspects as depredation among habitat types (Faaborg 2003). Although artificial nest studies may overestimate or underestimate actual predation rates, they do allow us to see a trend and behavior in depredation of nests (Faaborg 2003).

If the mink were to continue to be allowed to reproduce and establish themselves on the island, the bird population would suffer based on our data, but an interesting caveat is whether these ground-nesting birds will, in fact develop defensive strategies, such as placing their nests in trees in the future. On the other hand, species such as coastal and wetland shorebirds do not have that luxury, and current work at the

Omora Park is also addressing these species. However, our finding that both introduced species and disturbance together impact ground-nesting songbirds highlights the fact that it is essential to implement a conservation program for the island to ensure that primary forests remain unfragmented to maintain native avian reproduction. The birds on the island carry out specific functions, such as seed dispersal, insectivores (maintaining insect populations), and they are also a source of ecotourism for the island.

Acknowledgments

The authors wish to thank the Rosemond Lab, Dr. Bob Cooper, and Steve McGehee.

This research is part of the Omora Park's ongoing studies of the invasive, exotic fauna of the Cape Horn Biosphere Reserve (www.omora.org) and was coordinated by the Omora Sub-Antarctic Research Alliance (www.osara.org).

Literature Cited

- Anderson, C.B., R. Rozzi, J.C. Torres-Mura, S.M. McGehee, M.F. Sherriffs, E. Schuettler and A.D. Rosemond (*in press*). Exotic vertebrate fauna in the remote and pristine sub-Antarctic Cape Horn Archipelago region of Chile. *Biodiversity and Conservation*.
- Conroy, P.J. 1988. High nest predation by Brown tree snakes on Guam. *The Condor*. 90.1 pp.478-482.
- Faaborg, J. 2003. Truly artificial nest studies. *Conservation Biology*. 18.2 pp.639-370.
- Gates J.E. and Gysel L.W. Avian nest dispersion and fledging success in field forest ecotones. *Ecology*. 1978. 59.51. pp.871-883.
- Hensler, G.L. and Nichols J.D. The Mayfield Method of estimating nesting success: a model, estimators and simulation results. *The Wilson Bulletin*. 1981. 93.1 pp.42-53.
- ISSG Global Invasives Species Database 2004 Nov 24. www.issg.org. Accessed 2006 Feb 18.
- Levy, Sharon. 2003. Getting the drop on Hawaiian invasives. *BioScience*. 53.8. pp.694-699.
- Long, J.L. 2003. *Introduced Mammals of the World*. United Kingdom.
- Mittermeier, R., Mittermeier C., Robles-Gil, P., Pilgrim J., Fonseca. G., Brooks, J. and J. Konstant. 2001. *Wilderness: Earth's Last Wild Places*. Conservation International, Washington, DC: pp. 573.
- Picman, J. 1988. Experimental study of predation on eggs of ground nesting birds: Effects of habitat and nest distribution. *The Condor*. 90.1. pp. 124-131.

- Savidge, Julie A. 1987. Extinction of an island forest avifauna by an introduced snake. *Ecology*. 68. pp. 660-668.
- Simons, L. S. and Simons L.H. 1990. Experimental studies of nest-destroying behavior by Cactus Wrens. *The Condor*. 92. pp.855-860.
- Tamarin, R.H. and Malecha, S.R. 1972. Reproductive parameters in *Rattus rattus* and *Rattus exalonus* of Hawaii, 1968 to 1970. *Journal of Mammology*. 53.3 pp.513-528.
- Veblen, T.T. and Stewart, G.H. 1982. The Effects of introduced wild animals on New Zealand forests. *Annals of the Association of American Geographers*. 72(3):372-397.
- Vitousek, P.M. D'Antonio, L.L.Loope and R. Weterbrooks 1997. Biological invasions as global environmental change. *American Scientist* 84: 273-478.
- Willson, M.F. et al. 2001. Patterns of predation risk and survival of bird nests in a Chilean-agricultural landscape. *Conservation Biology*. 15(2):447-456.

Table 1. The percent depredation of mink and birds of each habitat, and the total % nest depredation of each habitat

Habitat	% Mink	% Bird	% Total
Anthropogenic shrubland	53	40	93
Beaver meadow	13	60	73
Secondary forest	20	40	60
Primary forest	13	20	33
Total	26	40	66

Figure 1. Mean daily survival rates (\pm SE) for each habitat type. Trend shows increasing survival from shrublands to beaver meadows to secondary forest to primary forest, but this result is not statistically significant ($F_{3,56}=1.4$, $p=0.24$).

Figure 2. Days observed (DO) until failure were graphed as a function of vegetation cover per nest. The regression showed that DO was significantly influenced by vegetation cover at a particular nest.

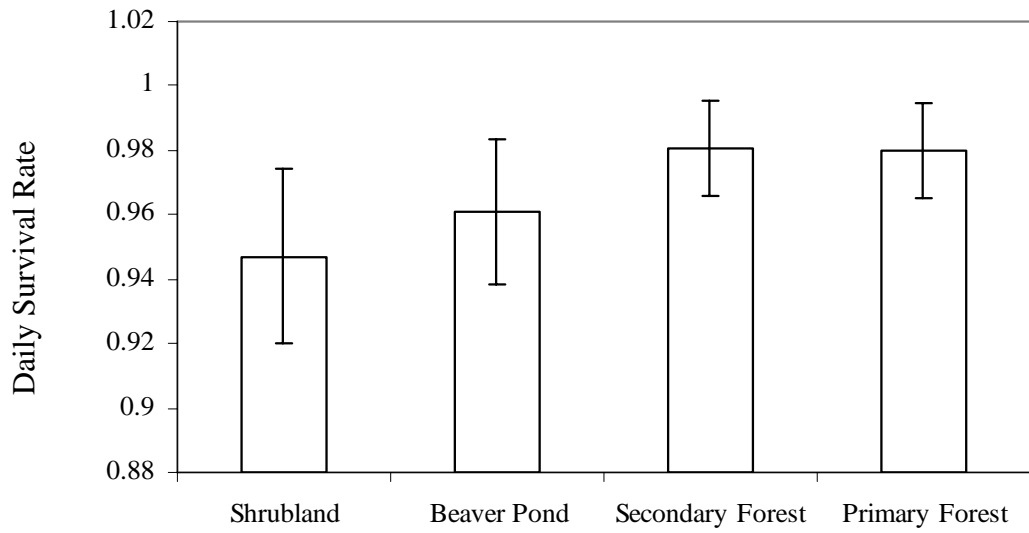


Figure 1

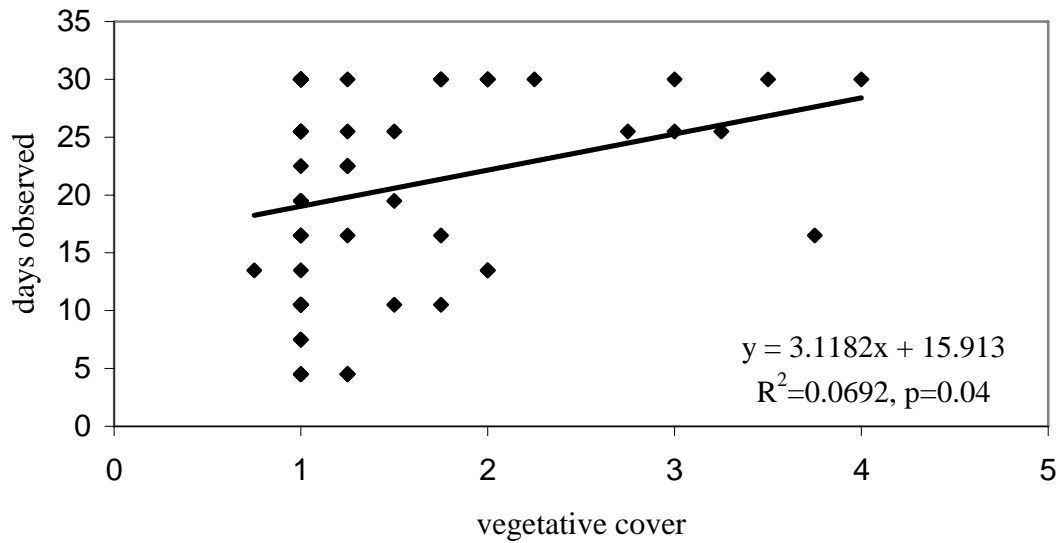


Figure 2

Appendix 1. Data for each nest (1-60) is included here. Habitat types were: The average vegetative cover was scored from 1-4 (25, 50, 75 or 100%). Distance (m) refers to the distance each nest was from the coast. The days until failure is the amount of time each nest remained out until the 30 days study period. The daily survival rate (DSR), measured as 1-(number of failed nests/ total days of exposure).

Nest	Habitat	Average vegetative cover	Distance (m)	Days till failure	DSR
1	A	2	0	18	0.944444
2	A	1	0	15	0.933333
3	A	3	0	30	0.966667
4	A	3.75	0	18	0.944444
5	A	3	0	30	0.966667
6	A	1.25	0	18	0.944444
7	A	1	0	12	0.916667
8	A	1	0	12	0.916667
9	S	1	250	30	0.933333
10	S	1.25	260	15	0.866667
11	S	1	270	30	0.933333
12	S	1	280	6	0.666667
13	S	1.25	290	9	0.777778
14	S	3.25	300	30	0.933333
15	S	1	310	12	0.833333
16	S	1	320	30	0.933333
17	S	1	330	30	0.933333
18	S	1.5	340	30	0.933333
19	S	1.25	350	15	0.866667
20	S	1	360	30	0.933333
21	S	1	370	30	0.966667
22	S	1	380	12	0.916667
23	S	2.25	390	30	0.966667
24	B	1	500	6	0.833333
25	B	2	500	18	0.944444
26	B	1	500	18	0.944444
27	B	1	500	18	0.944444
28	B	4	500	6	0.833333
29	B	1	500	30	0.966667
30	B	1	500	18	0.944444
31	B	1.5	500	30	0.966667
32	P	1	2000	30	0.9

Appendix 1 continued

33	P	1.25	2000	6	0.5
34	P	1.25	2000	30	0.9
35	P	1	2000	30	0.9
36	P	2	2000	18	0.9
37	P	1	2000	30	0.9
38	P	1	2000	15	0.8
39	P	1	2000	30	0.9
40	B	2	2500	9	0.888889
41	B	2	2500	30	0.966667
42	B	1	2500	12	0.916667
43	B	1	2500	30	0.966667
44	B	1.5	2500	30	0.966667
45	B	1.75	2500	12	0.916667
46	B	1	2500	30	0.966667
47	P	1.25	2200	30	0.9
48	P	1	2200	30	0.9
49	P	1	2200	30	0.9
50	P	1	2200	30	0.9
51	P	1	2200	30	0.9
52	P	0.75	2200	12	0.75
53	P	1	2200	15	0.8
54	A	1.25	1500	30	0.966667
55	A	3	1500	6	0.833333
56	A	1	1500	6	0.833333
57	A	1.75	1500	24	0.958333
58	A	3.5	1500	24	0.958333
59	A	1.75	1500	30	0.966667
60	A	2	1500	30	0.966667