

APPENDIX D

CRITIQUE OF THE RISK ASSESSMENT MODEL EMPLOYED TO CALCULATE TAKES IN THE HAWAII RANGE COMPLEX SUPPLEMENTAL DRAFT ENVIRONMENTAL IMPACT STATEMENT

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Abstract

Rather than using a fixed received level threshold for whether a take is likely to occur from exposure to mid-frequency sonar, the Navy proposed a method for incorporating individual variation. Risk is predicted as a function of three parameters: 1) a basement value below which takes are unlikely to occur; 2) the level at which 50% of individuals would be taken; and 3) a sharpness parameter intended to reflect the range of individual variation. This paper reviews whether the parameters employed are based on the best available science, the implications of uncertainty in the values, and biases and limitations in the model. Data were incorrectly interpreted when calculating parameter values, resulting in a model that underestimates takes. Errors included failure to recognize the difference between the mathematical basement plugged into the model, and the biological basement value, where the likelihood of observed and predicted takes becomes non-negligible; using the level where the probability of take was near 100% for the level where the probability of take was 50%; and extrapolating values derived from laboratory experiments that were conducted on trained animals to wild animals without regard for the implications of training; and ignoring other available data, resulting in a further underestimation of takes. In addition, uncertainty, whether due to inter-specific variation or parameter values based on data with broad confidence intervals, results in the model being biased to underestimate takes. The model also has limitations. For example, it does not take into account social factors, and this is likely to result in the model underestimating takes. This analysis has important management implications. First, not only do takes occur at far greater distances than predicted by the Navy's risk model, the fact that larger areas are exposed to a given received level with increasing distance from the source further multiplies the number of takes. This implies takes of specific individuals will be of greater duration and be repeated more often, resulting in unexpectedly large cumulative effects. Second, corrections need to be made for bias, and corrections will need to be larger for species for which there are no data than for species for which there are poor data. Third, the greater range at which takes would occur requires more careful consideration of habitat-specific risks and fundamentally different approaches to mitigation. The value of the model is that it provides a focus for future research on the effects of noise on marine mammals. In particular, the sensitivity analysis indicates the primary need for data is determining response probabilities of a wide range of species when exposed to received levels near the level at which 50% of individuals respond.

Introduction

The Navy distinguishes two types of takes: Level A, in which there is immediate injury or death; and Level B, in which there is no immediate injury, but cumulative exposure may lead to harm at the population level. However, in certain contexts, Level B harassment may lead to Level A takes through indirect mechanisms.

The population effects of Level A takes on populations are relatively easy to assess, as individuals that are killed are obviously removed from the population, and those that are injured are more likely to die whenever the population is next exposed to stress.

Calculating the population effects of Level B takes is a topic of contemporary research (Trites and Bain 2000). For example, Bain (2002a) explored using energetic consequences of behavior change in conjunction with population dynamics models to estimate population effects of Level B takes. Stress concurrent with Level B harassment would have additional population consequences. Stress may occur in the absence of behavioral change, or the absence of change in significant behavioral patterns such as foraging or nursing, or exclusion from optimal habitat. Lusseau et al. (2006) concluded disturbance caused a decline in and posed a significant threat to the survival of the bottlenose dolphin population in Doubtful Sound, New Zealand. While they noted vessel strikes were occurring (Level A takes), cumulative behavioral effects (Level B takes) were believed to be the primary threat to the population.

Models relating acoustic exposure to takes thus are not sufficient by themselves to interpret the effects of noise on populations. It is likely that different magnitudes of effect, whether physical harm, behavioral change that leads to physical harm, disruption of significant behavioral activities, or behavioral changes that pose negligible risk to populations when they occur only rarely but can become significant when exposure is prolonged or repeated, will have different relationships to noise. The different magnitudes of takes will have different population consequences. Thus it will be challenging to synthesize results of multiple studies, as different measured endpoints may belong on different curves relating them to noise, and different endpoints will have different population consequences. Further, the population consequences can depend on the health of the population (Bain 2002a). All these factors need to be considered when evaluating the environmental consequences of exposing marine mammals to noise.

Unconditional effects

Temporary Threshold Shifts in captive marine mammals are commonly used as an index of physical harm (e.g., Nachtigall et al. 2003, Finneran et al. 2002 and 2005, Kastak et al. 2005). Limiting experimental noise exposure to levels that cause temporary effects alleviates ethical concerns about deliberately causing permanent injury. However, repeated exposure to noise that causes temporary threshold shifts can lead to permanent hearing loss. In fact, chronic exposure to levels of noise too low to cause temporary threshold shifts can cause permanent hearing loss. Animal models (e.g., rats, cats,

monkeys, chinchillas) have been used for tests of noise causing permanent physical harm (Henderson et al. 1991, Gao et al. 1992, Blakeslee et al. 1978, Clark 1991). Damage to hearing from noise exposure is an example of unconditional injury from noise. OSHA (2007) requires limiting human exposure to noise at 115 dB above threshold (equivalent to 145 dB re 1 μ Pa for killer whales, Szymanski et al. 1999) to 15 minutes.

Stress reactions are another available index (e.g., Romano et al. 2004). Ayres (personal communication) found evidence suggesting that whale watching results in increased levels of stress hormones in wild killer whales.

Conditional effects

Changes in behavior resulting from noise exposure could result in indirect injury in the wild. A variety of mechanisms for Level B harassment to potentially lead to Level A takes have been identified.

Gas bubble lesions have been observed in beaked whales (Jepson et al. 2003, Fernandez et al. 2005, Cox et al. 2006). A variety of mechanisms have been proposed for this. While some have proposed these may be due to acoustically mediated bubble growth, and hence are an unconditional consequence of noise exposure (Crum and Mao 1996), it is more likely that these result from decompression sickness. That is, changes in dive behavior may prevent clearance of nitrogen gas from the body, resulting in larger bubbles than would occur in undisturbed dive patterns. One possible change is that beaked whales may remain submerged for an unusually long period of time, and then rapidly ascend. The rapid ascent is a change in behavior that prevents nitrogen from remaining in solution in the blood. Zimmer and Tyack (2007) questioned whether the rapid ascent mechanism would actually result in lesions, and proposed another behavior change that might occur is interruption of deep dives. Deep dives allow the lungs to collapse, preventing nitrogen from reaching the body. Further, a series of rapid breaths at the surface can be used to clear nitrogen absorbed under pressure. Interruption of the normal surface interval can allow nitrogen to build up over time. Changes in depths of dives are of more concern than rapid ascents as this mechanism would be applicable to a wide range of species, while if the rapid ascent mechanism is involved, it would be primarily a concern for deep diving species (Zimmer and Tyack 2007).

While failure to flee may lead to injury in beaked whales, flight may lead to injury in other species. Minke whales have been found stranded after sonar exercises (NOAA and Navy 2001). A minke whale was observed traveling at high speed during exposure to mid-frequency sonar in Haro Strait in 2003. It is easy to see how such behavior would lead to stranding when a beach is located in front of the whale, as minke whales lack echolocation and visibility is limited underwater. Exhaustion from rapid flight leading to heart or other muscle damage (Williams and Thorne 1996) could also account for increased mortality such as was observed in harbor porpoises following sonar exercises in Juan de Fuca and Haro Straits in April and May of 2003. Harbor porpoises, in contrast to

Dall's porpoises, rarely engage in sustained high energy activities such as rapid swimming or bow riding, and hence are less adapted to long distance flight responses.

Even successful flight may have negative survival consequences. In the absence of disturbance, individuals will tend to occupy optimal habitat. Displacement from optimal habitat will have consequences that will depend on the duration of the displacement, the quality of the alternate habitat, and the condition of the individuals at the time of displacement.

Separation of individuals from social units is another consequence of noise exposure that may lead to mortality. In 2003 in Haro Strait, some killer whales responded to mid-frequency sonar by seeking shelter behind a reef. Others chose to flee, resulting in splitting of a pod that historically spent all of its time together as a single unit. While no deaths resulted from this particular incident, other killer whales have been observed separated from their social units resulting in death prior to reunion or requiring human intervention to restore the individual to its social unit (Schroeder et al. 2007).

Temporary threshold shifts may conditionally lead to harm. Impaired hearing ability increases vulnerability to ship strike. In 2003, blunt force trauma was identified as a cause of death in the investigation of harbor porpoise mortalities following exposure to mid-frequency sonar in Washington State. A minke whale was nearly struck by a research vessel in the area where one had been observed fleeing mid-frequency sonar exposure. These species are familiar with boats in that area, and normally avoid them by a wide margin when they can hear them coming.

Impaired auditory ability may also increase predation risk. For example, Dahlheim and Towell (1994) reported an attack by killer whales on white-sided dolphins. The approach by the whales went undetected due to the noise of the research vessel. Further, impaired hearing may impair foraging ability and communication (Bain and Dahlheim 1994).

The Risk Function Model

The risk function uses three parameters. B is the received level at which the most sensitive individuals start to respond with changes in significant behaviors such as foraging. K is the difference in received level between the level at which half of individuals respond and the level at which the most sensitive individuals respond. That is, $B+K$ is the level at which 50% of individuals respond. A is a shape parameter that attempts to capture the variability in responsiveness of the population. That is, are essentially all the individuals the same and the bulk of them become responsive when the received level is near $B+K$, in which case a simple threshold model would provide a good approximation, or is there a lot of variation in the population, in which case many individuals become responsive when received levels are near B ?

The model is based on the hypothesis that some individuals start to respond at lower levels than others. It anticipates that some individuals will hold out until very high levels

before responding. The model includes parameters that allow it to be applied appropriately to species with differing noise tolerance. However, the Navy used one set of parameter values to predict the responses of all species. This paper reviews the accuracy of the choice of parameter values, the implications of using the wrong parameter values, and whether the model makes unbiased predictions when uncertainty in the parameter values exists.

Limitations

Like many models, the risk model has limitations. It fails to take into account social interactions. For example, the model anticipates that individuals may move away from a source at different exposure levels, but fails to recognize that this would result in individuals becoming separated from the group. This is likely to lead to the curve becoming asymmetrical, with the "holdouts" responding to the behavior of their schoolmates rather than the sound. As the area exposed to lower levels of noise is larger than the area exposed to higher levels of noise, this would result in more individuals being affected than the model predicts for social species.

The model does not account for multiple sources. Kruse (1991), Williams and Ashe (2007) and Bain et al. (2006) noted that killer whale responses to vessels varied with the number of vessels present. The magnitude of certain responses increased on the order of 10% per source, although Williams and Ashe (2007) noted that large numbers of sources could result in changes in the opposite direction of small numbers of sources, potentially canceling out the effect. That is, rather than a risk function that simply identifies how likely a response is to occur, one that takes into account the magnitude of the response would be ideal.

Pingers have been used to reduce entanglement in gillnets. Kraus et al. (1997) were able to reduce entanglement of harbor porpoises by 90%. Gearin et al. (1996, 2000) used more pingers, and were able to reduce entanglement by 95%. While this could be accounted for by the fact that more pingers increase the minimum sound level at the net (Bain 2002b), Laake et al. (1997, 1998, 1999) found porpoises typically remained much farther from the net than the spacing between pingers, even after the avoidance response declined due to habituation. Thus, the effect of multiple sources seems larger than the effect of fewer sources. Pingers have also been successful in protecting other species from nets (Barlow and Cameron, 1999; Cameron 1999, Stone et al. 1997).

In addition to quantitative changes in response to multiple sources, there may be a qualitative change in the response. For example, noise is used in drive fisheries of many odontocete species to cause stranding or near strandings. That is, multiple sources were used to displace individuals in a particular direction, and the consequences (stranding) were more serious than displacement from the source alone as would result from exposure to a single source.

The risk to the population of qualitatively different responses varies not only with the type of response, but the circumstances. If the response is going ashore, fatalities are highly likely to result. If the response is slowly moving away for a short period of time, no fatalities are likely to result. However, if the response is to slowly move away from a prime feeding area for an extended period of time, and the population is food limited, fatalities may result, and the number is likely to be related directly to the duration of exclusion from the feeding area, and only indirectly to the cumulative sound energy received.

Finally, the model assumes that marine mammals behave independently from each other. This is not likely to be the case. Even species that are normally solitary, like harbor seals, have been observed to school in response to high energy noise (personal observation). To remain a member of a group, individuals must remain in geographic proximity to each other. As more sensitive individuals move away, others who are not sufficiently disturbed by the sound itself would need to move as well to remain members of the group. The result is likely to be a step function at moderate exposure levels rather than the gradual increase in risk predicted by the model. The result would be that risk is underestimated. The proportion of individuals necessary to lead all individuals to respond in a similar manner to noise is likely to vary among species, and propensity to mass strand may be a good predictor of the importance of this effect.

Datasets

The Navy chose to rely upon three datasets.

Captive cetaceans

Studies of captive marine mammals provide an excellent setting for identifying direct effects of sound. E.g., one of the datasets employed by the Navy consists of studies relating short-term exposure of bottlenose dolphins and belugas to high levels of noise to Temporary Threshold Shifts. The Navy (Dept. Navy 2008b, p 3-7) noted aggressive behavior toward the test apparatus, suggesting stress was another consequence of the test (see also Romano et al. 2004). Such effects would be unconditional results of noise exposure.

However, extrapolation of the level at which aggression was observed to the level at which behaviorally mediated effects might occur in the wild is problematic, as this depends on how well trained the subjects were. For example, the Navy has been a leader in training dolphins and other marine mammals to cooperate with husbandry procedures. Tasks like taking blood, stomach lavage, endoscopic examination, collection of feces, urine, milk, semen and skin samples, etc. once required removing individuals from the water and using several people to restrain them. With training, painful and uncomfortable procedures can be accomplished without restraint and with a reduction in stress that has significantly extended lifespans of captive marine mammals (Bain 1988).

That is, the absence of avoidance or aggressive behavior does not imply an absence of physical harm, much less the absence of potential for behavior changes that may lead to indirect harm.

Physical harm may occur in the wild without avoidance responses as well. Yano and Dahlheim (1995) found killer whales continued to predate on longlines despite being physically injured by deterrents such as gunshots. Reeves et al. (1996) reviewed other examples from fishery interactions of injurious approaches to deterrence failing.

If belugas and bottlenose dolphins are like killer whales, and the 50% risk level is about 15 dB below the 50% risk level for behavioral change in trained animals (see below), this would put their value around 170 dB re 1 μ Pa. Even this is likely to be an overestimate, as boat motors with a source level of 165 dB re 1 μ Pa can cause behavioral changes in bottlenose dolphins (Nowacek et al. 2001.) This new value, 170 dB re 1 μ Pa, averaged with the other Navy datasets, would drop the average 50% risk level to 160 dB re 1 μ Pa.

Killer whales

The second dataset is killer whales exposed to mid-frequency sonar from the USS Shoup in Haro Strait, Washington, in May, 2003. The level quoted in the HRC SDEIS (Dept. Navy 2008b) is an estimate of the received levels experienced when mid-frequency sonar was transmitted from about 3 km away. This level caused major behavioral changes in 100% of exposed whales (Risk=1 for Level B takes of a magnitude that in other contexts or species could lead indirectly to physical harm), but was not believed to have caused Level A takes (the whales did not strand, and received levels were estimated to be too low to have caused threshold shifts, NMFS OPR 2005) in any individuals (Risk = 0). However, much more data are available from the May, 2003 Shoup incident. Behavioral changes were first observed at 47 km (where the received level was estimated to be 121 dB). The behavioral response was tail slapping by about 25% of the individuals observed, which is consistent with observed responses to vessel noise at a similar level. At a distance greater than 22 km, the direction of travel changed away from a feeding area, and hence foraging behavior was disrupted. At this distance, the received level may have increased to the neighborhood of 135 dB re 1 μ Pa with about 6 dB of reduced spreading loss and 6 dB reduced absorption. This would be comparable to a vessel traveling at low speed approaching to within 10 m, which is very difficult to accomplish without causing whales to turn away. 100% of killer whales responded by abandoning their feeding ground and moving away from the noise source at this received level. While vessels cause diversion from straight-line paths, they have not been observed to displace killer whales from feeding areas (vessels have been observed to displace killer whales from resting areas, but this is likely mediated by presence rather than noise, as the effect is observed in the presence of silent vessels, Trites et al. 1995). Thus it is not surprising that a qualitatively different behavioral response was exhibited. The peak exposure level was estimated to be 175 dB re 1 μ Pa (HRC SDEIS, although NMFS noted that estimated levels tended to overestimate measured levels by 1-10 dB [NMFS OPR 2005]), so the peak exposure level may have been only 165 dB). In addition to changing

travel patterns, the pod split, with approximately 50% of the pod continuing to shelter in an acoustic shadow zone, and the other 50% fleeing at high speed. Such behavior has not been observed in the presence of vessels alone. It should be emphasized that 100% of killer whales exhibited a disruption of a significant life process, foraging, at a level that may have been less than 135 dB re 1 μ Pa, in contrast to the value used in the SDEIS, 169.3 dB re 1 μ Pa for a 50% response.

Additional datasets are available for killer whale responses to noise. E.g., in Bain and Dahlheim's (1994) study of captive killer whales exposed to band-limited white noise in a band similar to that of mid-frequency sonar at a received level of 135 dB re 1 μ Pa, abnormal behavior was observed in 50% of the individuals. This is far lower than the level observed in bottlenose dolphins. In addition, Bain (1995) observed that 100% of wild killer whales appeared to avoid noise produced by banging on pipes (fundamental at 300 Hz with higher harmonics) to the 135 dB re 1 μ Pa contour. This indicates the difference between wild and captive killer whales (non-zero risk in captive marine mammals might correspond to 100% risk in wild individuals of the same species), as well as implying that risk of 100% may occur by 135 dB re 1 μ Pa for this genus in the wild.

Further, killer whales begin responding to vessel traffic at around 105-110 dB re 1 μ Pa with minor behavioral changes. By 135 dB re 1 μ Pa, disruption of foraging may approach 100%. Received level appears to be more important than proximity (Bain 2001). For risk to increase from near 0 at 105 dB re 1 μ Pa to near 100% by 135 dB re 1 μ Pa, with $A=10$, the 50% risk level would need to be about 120 dB re 1 μ Pa. Substituting 120 for 169 dB re 1 μ Pa reduces the average level for 50% risk by about 16 dB to 144 dB re 1 μ Pa. Substituting 135 dB re 1 μ Pa would reduce the average by 8 dB to 157 dB re 1 μ Pa.

Finally, the Navy's characterization of the killer whale dataset is incorrect. They indicate the effects observed in the presence of mid-frequency sonar in Haro Strait were confounded by the presence of vessels. However, the effects of vessels on killer whales have been extensively studied (e.g., Kruse 1991, Williams et al. 2002ab, Bain et al. 2006). Behavioral responses attributed to mid-frequency sonar are qualitatively different than those observed to vessels alone. While the observations are anecdotal, they were not inconsistent. The sonar signal was blocked from reaching the whales with full intensity by shallow banks or land masses during three segments of the observation period. The "inconsistencies" can be attributed to differences in behavior depending on whether there was a direct sound path from the Shoup to the whales. It should be noted there was extensive study of this population prior to exposure (see Bigg et al. 1990 and Olesiuk et al. 1990 for a description of typical research protocols), as well as extensive post-exposure monitoring (e.g., Bain et al. 2006).

Right whales

Similarly, the right whale data relied upon are of limited value. While they clearly illustrate that the value at which 50% of animals are influenced is below 135 dB re 1 μ Pa

and are therefore helpful in determining the upper limits of the B+K value, they lack sufficient low level exposures needed to fit the low end of the curve. As with killer whales, the Navy misused the data. They averaged values which resulted in 100% response. Thus the average value exceeds the level resulting in a 50% risk.

Right whales exposed to alerting devices consistently responded when received levels were above 135 dB re 1 μ Pa. Due to the small sample size (six individuals), it is unclear whether this is close to the 50% risk, the 100% risk level, or both. These data do not allow identification of B, as lower exposure levels were not tested. In mysticetes exposed to a variety of sounds associated with the oil industry, typically 50% exhibited responses at 120 dB re 1 μ Pa. Thus right whales may be similar to killer whales.

The consequences of using incorrect values can be seen by comparing the observed results of the right whale exposures to alert signals (Nowacek et al. 2004) with those predicted by the Navy model. Using the values of B=120, K=45, and A=10 in the HRC SDEIS (Dept. Navy 2008b), the probability of responses for the exposed whales are shown in column two of Table 1. The formula underestimated the number of takes by a factor of over 500. The Navy proposed using A=8 for mysticetes in recognition of this, and the results are shown in column 3. While improved, the model still underestimated takes by a factor of 183. One could try B=105 and K=15. Using A=10 provides a reasonable approximation, overestimating takes by 20% (column 4). A better approximation is provided by A=2, which predicts the number of takes within 2% (column 5). While the probability of all four right whales exposed to the highest alert signals responding is much less than one in a billion based on the Navy model and allows one to unequivocally reject the Navy's choice of parameter values as applying to that species, numerous other combinations of parameter values would fit the data as well as the values shown in the table here. Substituting 120 dB re 1 μ Pa for 139 dB re 1 μ Pa results in an average 6 dB lower at 159 dB re 1 μ Pa.

Table 1. Risk for right whales (model vs. observed)

Received Level (dB re 1 μ Pa)	RISK B=120,K=45,A=10	RISK B=120,K=45,A=8	RISK B=105,K=15,A=10	RISK B=105,K=15,A=2
Responded				
148	0.008647	0.022021	0.999973	0.891548
143	0.001217	0.004641	0.999908	0.86521
137	5.92E-05	0.000415	0.999488	0.819864
135	1.7E-05	0.000153	0.999026	0.800039
133	4.06E-06	4.86E-05	0.998059	0.777052
No Response				
134	8.52E-06	8.79E-05	0.998633	0.788974
Error Factor	502	183	0.83	1.01

Datasets not considered

The Navy incorrectly concludes that additional datasets are unavailable. In addition to the other killer whale datasets mentioned above, data illustrating the use of acoustic harassment and acoustic deterrent devices on harbor porpoises illustrate exclusion from foraging habitat (Laake et al. 1997, 1998 and 1999, Olesiuk et al. 2002). Data are also available showing exclusion of killer whales from foraging habitat (Morton and Symonds 2002), although additional analysis would be required to assess received levels involved. The devices which excluded both killer whales and harbor porpoises had a source level of 195 dB re 1 μ Pa, a fundamental frequency of 10 kHz, and were pulsed repeatedly for a period of about 2.5 seconds, followed by a period of silence of similar duration, before being repeated. Devices used only with harbor porpoises had a source level of 120-145 dB re 1 μ Pa, fundamental frequency of 10 kHz, a duration on the order of 300 msec, and were repeated every few seconds. Harbor porpoises, which the Navy treats as having a B+K value of 120 dB re 1 μ Pa (with A large enough to yield a step function) in the AFAST DEIS (Dept.Navy 2008a), 45 dB lower than the average value used in the HRC SDEIS, may be representative of how the majority of cetacean species, which are shy around vessels and hence poorly known, would respond to mid-frequency sonar. Even if harbor porpoises were given equal weight with the three species used to calculate B+K, including them in the average would put the average value at 154 dB re 1 μ Pa instead of 165 dB re 1 μ Pa.

Harbor porpoise responses to various acoustic devices have been documented in captivity and the wild. Pingers with a source level of 130 dB re 1 μ Pa displace wild harbor porpoises to a distance of at least 100-1000 m, where the received level was likely in the

neighborhood of 80-90 dB re 1 μ Pa. Studies of harbor porpoises in captivity also found responses to acoustic deterrent devices, but could not be tested at such distances due to limitations in facility size (Kastelein et al. 1997, 2001). This is another example of how studies with captive cetaceans can produce misleading results. Airmar devices with a source level of 195 dB re 1 μ Pa displaced an estimated 95% of harbor porpoises to a distance of 3 km. While received levels were not measured, they could have been in the neighborhood of 120-130 dB re 1 μ Pa. These findings are well modeled with a B value of 70 dB re 1 μ Pa, a K value of 25, and an A value of 4.

Many species are poorly known, due in part to difficulties approaching them from boats and in part because they do not fare well in captivity. Species that may exhibit vulnerability to noise comparable harbor porpoises include many species of *Stenella* (e.g., striped dolphins), beaked whales, sperm whales (which are best studied from sailboats rather than motorized vessels, and show disruption of foraging at levels below 130 dB re 1 μ Pa, Jochens et al. 2006), and numerous poorly known species. In contrast, Dall's porpoises are known to bow ride, and appear far less easily disturbed by noise from airguns than harbor porpoises (Calambokidis et al. 1998). They may be an example of a relatively noise tolerant species like the bottlenose dolphins included in the SDEIS.

There are also data that are based on other noise sources. E.g., effects of vessel traffic on whale and dolphin behavior could be interpreted in terms of received levels. While engine noise tends to be continuous rather than intermittent like sonar, in a reverberant environment, mid-frequency sonar may be received as a nearly continuous sound (personal observation).

Likewise, records of marine mammal responses to broadband noise sources like airguns are also likely to be informative. While it may be difficult to extrapolate levels resulting in takes due to potential differences in perception of broadband and narrowband signals, and pulses rather than continuous sounds, they can give an idea of the range of intra-specific and inter-specific variation in B and K values and be applicable to determining the A parameter.

E.g., Calambokidis et al. (1998) found harbor seal responses to airguns typically consisted of visually orienting at received levels from 143 to 158 dB re 1 μ Pa and moving away at received levels from 158 dB to 185 dB re 1 μ Pa. However, one harbor seal oriented at 163 dB re 1 μ Pa rather than moving away. The highest measured received levels for Dall's porpoises were about 170 dB re 1 μ Pa, but only about 142 dB re 1 μ Pa for harbor porpoises. Similarly, the highest received levels measured for California sea lions were about 180 dB re 1 μ Pa, but only about 160 dB re 1 μ Pa for Steller sea lions. The highest measured received level was also 160 dB re 1 μ Pa for gray whales. That is, closely related species pairs may differ in their responsiveness to noise by over 20 dB, and taxonomically diverse species pairs may exhibit similar responsiveness.

TTS data similar to those available for cetaceans have been collected from harbor and elephant seals, and California and Steller sea lions (Kastak et al. 1999, 2005). As with cetaceans, field data suggest the Navy parameter values will underestimate takes of some

pinniped species, though they may provide a reasonable approximation for harbor seals and California sea lions (e.g., the data described above). Pinniped hearing in species studied to date is less sensitive than in cetaceans (e.g., California sea lions, Schusterman et al. 1972; Steller sea lions, Kastelein et al. 2005; harbor seals, Møhl 1968; northern fur seals, Moore et al. 1987; odontocetes, Au 1993), and it is commonly assumed they are less vulnerable to noise as a result. However, comparisons of Steller sea lions with Dall's porpoises and gray whales exposed to airgun noise indicates this is not always the case. A detailed consideration of pinnipeds is beyond the scope of this paper.

Using the datasets discussed above, 50% risk levels based on trained cetaceans may be 165 dB re 1 μ Pa, 120 dB re 1 μ Pa for killer and right whales, and 95 dB re 1 μ Pa for harbor porpoises. The average of 95, 120, 120 and 165 is 125 dB, 40 dB lower than the 50% risk value of 165 dB used in the Navy model. Even if one uses more stringent criteria for what constitutes takes (120 dB for harbor porpoises, 135 dB for killer and right whales, and 170 dB for bottlenose dolphins), the average would be 140 dB, which is 25 dB lower than the Navy model. Setting B to 100, K to 40, and A to 10 would result in roughly 40 times the number of takes than the model predicts using the Navy's parameter values.

Parameter values

The use of default values for model parameters is problematic. The available data are likely to be biased toward noise tolerant species. That is, species that are intolerant of noise are difficult to approach closely enough to study. They tend to fare poorly in captivity. E.g., spinner dolphins and harbor porpoises showed very poor survivorship in captivity, in contrast to bottlenose dolphins (Bain 1988). Thus averages based on available data are likely to underestimate effects on species for which data are not available.

While the Navy has proposed assuming noise tolerance is predictable along taxonomic lines, which correlate with hearing ability, empirical data do not support this assumption (Bain and Williams 2006). Likewise, there is interspecific variation in noise tolerance in fish (Kastelein 2008).

B Value

The basement value should be set low enough that the risk function predicts takes at the lowest of the level resulting in unconditional injuries, the level at which behaviorally mediated injuries are possible, and the level resulting in minor behavioral changes or stress that can have population level effects with sustained or repeated exposure.

An important property of the model is that the biologically observed basement value is different than the mathematical basement value. The Navy proposes using 120 dB re 1 μ Pa as the basement value. They indicate the selection of this value is because it was commonly found in noise exposure studies. However, 120 dB re 1 μ Pa has broadly been

found as the value at which 50% of individuals responded to noise, not a small percentage. Further, a mathematical B of 120 dB corresponds to a risk of less than 2% at 150 dB (with $K=45$ and $A=10$), which would be difficult to detect in empirical studies. That is, the studies should be re-evaluated to determine the level at which a small percentage of individuals responded, and then a further correction for the difference between mathematical B and the empirically determined biological B would be needed.

However, further consideration should be given to the nature of the responses used in those studies to determine whether they represent significant behavioral changes or are only likely to have a population scale effect with sustained or repeated exposure.

For example, many looked at changes in migration routes resulting from noise exposure, and found that 50% of migrating whales changed course to remain outside the 120 dB re 1 μ Pa contour (Malme et al. 1983, 1984). These results might be interpreted in several ways. They could be seen as minor changes in behavior resulting in a slight increase in energy expenditure. Under this interpretation, they would not qualify as changes in a significant behavior, and are irrelevant to setting the basement value. They could be interpreted as interfering with migration, even though the whales did not stop and turn around, and hence 120 dB would make an appropriate B+K value rather than B value. Third, the change in course could have been accompanied by a stress response, in which case the received level at which the course change was initiated rather than the highest level received (120 dB re 1 μ Pa) could be taken as the biological basement value.

As discussed above, sensitive species like harbor porpoises may be significantly affected by levels below 100 dB re 1 μ Pa (Kastelein et al. 1997, 2000, 2001). Foraging behavior of killer whales can be disrupted by levels on the order of 105-110 dB re 1 μ Pa or less (Williams et al. 2002ab, data in Bain et al. 2006). These are far below the 120 dB re 1 μ Pa level proposed, and as mentioned above, the mathematical B value needed to predict detectable changes at 110 dB would be far lower than 110 dB. For example, $B=80$, $K=45$, and $A=10$ predicts a risk of less than 2% at 110 dB.

K Value

The K value reflects the difference between the mathematical B value and the level at which 50% of individuals respond. Since determining the B value has problems of its own, this critique will focus on determining the B+K value. The 50% risk level is relatively easy to determine, and has been commonly reported in the literature, as noted in the SDEIS. However, the most common value was 120 dB re 1 μ Pa, as noted in the SDEIS, yet these studies were not used to calculate B+K. Instead, other datasets were used, and the numbers derived were not the 50% risk levels. As mentioned above, there are problems with extrapolation of responses in trained animals to wild animals, and the right and killer whale values were based on levels that resulted in nearly 100% risk, not 50% risk. (It may not be possible to determine a level at which 50% risk occurred in killer whales, but perhaps collaboration among killer whale researchers, whale watch operators, and the Navy might identify the B+K level for that event).

The 50% risk level is the median level at which individuals begin to respond, not the mean as calculated in the SDEIS. While there are data suggesting risk of threshold shift is related to duration of exposure, and hence the consequences of exposure to continuous noise sources would be different than exposure to intermittent sources, there are no such data for behaviorally mediated effects. Many species strongly avoid motorized vessels, and hence are more vulnerable to noise than the average of the species considered above. Such species are likely to include those in the sperm and beaked whale families, Pacific right whales, blue whales, melon-headed and pygmy killer whales, right whale dolphins, and Clymene, striped and rough-toothed dolphins. A smaller number of species, like Dall's porpoises, are more tolerant of noise sources than the average of the species considered above. Thus it is unlikely that the average value of B+K across cetacean species would be above 120 dB re 1 μ Pa, although the value would vary across species.

A value

While the A value is described as relating to the sharpness of the risk function, it also influences the symmetry of the function. As A increases, risk is redistributed from low noise levels to higher noise levels. The relative risk to the population, as opposed to risk to individuals, can be described as the risk to individuals at a given received level times the relative number of individuals receiving that level. As the sound spreads to larger areas, more individuals are exposed to lower levels of noise. The shape of the risk function and the spreading loss model determine the received level that poses the most risk to the population. At high received levels, the risk to the population may be small, because although the risk to individuals is high, the number of individuals likely to be exposed is small. At low levels, the risk to the population may be again small, because although the number of individuals exposed is high, the risk to those individuals is low. At intermediate values, the population experiences the most risk. When A is low, the risk to the population peaks near B, and at high A values, the risk is concentrated near B+K.

The choice of A value appears arbitrary. The Navy indicated they wanted to allow for more response at low levels, and adjusted the A value to accomplish this. However, this would have been better accomplished by lowering the B and B+K values as suggested above.

The significance of an A value underestimating the number of individuals responding to low levels of noise and overestimating the number of individuals responding to high levels of noise is that the area exposed to low levels of noise is larger than the area exposed to high levels of noise, so the calculation would lead to an underestimate of takes.

Calambokidis et al. (1998) employed an appropriate methodology for obtaining data for calculating A values of marine mammals exposed to airguns. They used a small vessel which moved toward and away from the seismic survey vessel, and hence were able to observe behavior and measure received values at distances of over 70 km as well as close

to the seismic survey vessel. Thus they were able to observe normal behavior in the presence of low levels of noise, as well as identify levels above which 100% of individuals exhibited behavioral change, and note inter-specific variation in response curves.

Interaction of Terms

It appears that B+K is a stronger predictor of the number of takes than either factor separately. As a result, similar risk curves can be generated for many different pairs of B and K as long as the sum is held constant. K and A together determine the range over which risk rises from 5% to 95%. Similarly, pairs of K and A over a range of values can generate similar risk curves.

With B=120, K=45, and A=10, the risk function predicts risk is near zero at received levels near 120, and that over 99.9% of takes will occur above 138 dB re 1 μ Pa. Even with A = 8, 99.9% of takes occur at levels above 135 dB. With A values this large, B is better described as the level at which the risk function is undefined (it requires dividing by 0) rather than the level at which risk becomes negligible. That is, the mathematical basement value and the biological basement value are different. The level at which data from marine mammals show barely detectable risk will be far above the mathematical basement value when K is 45 and A is 8 or 10. When K or A are small, the mathematical and biological B values become similar.

Another way of looking at the difference between the mathematical and biological basement value is to ask how much risk is detectable. In field studies, it will be difficult to distinguish responses that occur in only 5% of individuals from baseline behavior. Even if a study were sensitive enough to detect this, the received level to cause 5% risk is more than 30 dB above the mathematical B value for B=120, K=45 and A=8 or 10. That is, if risk becomes biologically detectable at 120 dB, the B value used in the equation for risk should be far lower. When the model uses the biological B value as the mathematical B value, it does not accurately predict the observed pattern of takes.

Long range effects

The Navy expressed uncertainty over whether there would be long distance effects, even when sound levels were received that are known to cause effects at close range. While I am not aware of observations at 65 nautical miles, responses at over 20 miles have been observed in killer whales to mid-frequency sonar, as well as at over 15 miles to mid-frequency sonar in Dall's porpoises, and harbor porpoises appeared to respond to airguns at over 40 nm (personal observation). The porpoises were responding at distances greater than they would respond to natural predators (killer whales), which are not believed to be detectable at those ranges.

Further evidence of long range responses to noise can be seen in differences in detection rates of some species using acoustic means and ship-based observations. Such studies indicate that species like Pacific right whales and blue whales avoid motorized vessels at distances which place them over the horizon (Wade et al. 2006, Širović 2006).

Uncertainty and Bias

To assess the effects of uncertainty in the parameter values (B, K, and A) on bias in the estimated number of takes, the following method was used. Two spreading loss models were used. A spherical spreading loss model was used, although this was likely to underestimate received levels, particularly at long distances. The other was spherical spreading at close range followed by a cylindrical spreading loss at longer distances model. An accurate model would depend on actual conditions, which would vary from one sonar exercise to another, both as bottom topography varies from place to place and the structure of the water column varies from time to time. The two models chosen should bracket actual conditions, and will serve for purposes of illustration at this stage. In both models, absorption at 3.5 kHz was used to correct for excess attenuation (Richardson et al. 1995). A source level of 235 dB re 1 μ Pa was assumed for purposes of illustration.

Individuals were assumed to be distributed uniformly with distance from the source, although in practice, action areas will be large enough that density could reasonably be expected to vary. The action area was divided into concentric rings 10 meters across. As the diameter of the ring increased, the area within the ring increased:

$$A = \pi r_o^2 - \pi r_i^2$$

where r_o is the outer diameter and r_i is the inner diameter of the ring.

The risk was calculated for individuals within the ring using the Navy equation, and the relative number of individuals experiencing that risk level was based on the area of the ring. As in the equation for the individuals, the cumulative impact on the population was normalized to 1 based on the Navy default parameters. The effects of uncertainty were observed by allowing the parameters to vary above and below the default values.

Using this model, the contributions of the innermost rings were small, due to their small area, and the contribution of the outermost rings were small, due to the low risk experienced by individuals in those ring. Figures 1-20 show the shape of the risk function and the relative numbers of takes that would occur as a function of received level for a variety of parameter value combinations.

Selected values of B, K and A were used to calculate relative effects, and the results are shown in Table 2 for a spherical spreading model, and Table 3 for a model that assumes spherical spreading for the first 2 km and then cylindrical spreading after that. The default values are shown in bold. Take numbers are based on Alternative 3 in the Hawaii

Range Complex SDEIS (Dept. Navy 2008b), which in turn is based on the No Action Alternative, Table 3.3.1-1. Where the number of takes approaches the size of the population, the actual number of takes will be smaller than shown in the table. However, individuals will be taken multiple times and the duration of takes will be longer than if the calculated number of takes were small. Presumably, longer and more frequent takes of individuals will have more impact on the population than takes due to single exposures.

Table 2. Sensitivity Analysis based on a spherical spreading model

B	K	A	Spreading Model	Relative Effect	Humpback takes	Striped Dolphin takes	Basis
80	45	10	Inv. Square	185.29	2,826,414	867,898	Vary B
90	45	10	Inv. square	75.25	1,147,864	352,471	Vary B
100	45	10	Inv. square	23.92	364,876	112,041	Vary B
110	45	10	Inv. square	5.68	86,643	26,605	Vary B
120	45	10	Inv. square	1.00	15,254	4,684	SDEIS
130	45	10	Inv. square	0.14	2,136	656	Vary B
140	45	10	Inv. square	0.02	305	94	Vary B
120	5	10	Inv. Square	167.18	2,550,164	783,071	Vary K
120	15	10	Inv. square	62.22	949,104	291,439	Vary K
120	25	10	Inv. square	18.33	279,606	85,858	Vary K
120	35	10	Inv. square	4.47	68,185	20,937	Vary K
120	45	10	Inv. square	1.00	15,254	4,684	SDEIS
120	55	10	Inv. square	0.23	3508	1077	Vary K
120	65	10	Inv. square	0.06	915	281	Vary K
120	75	10	Inv. square	0.01	153	47	Vary K
120	45	1	Inv. square	42.40	646,770	198,602	Vary A
120	45	5	Inv. square	3.27	49,881	15,317	Vary A
120	45	8	Inv. square	1.40	21,356	6,558	Vary A
120	45	10	Inv. square	1.00	15,254	4,684	SDEIS
120	45	12	Inv. Square	0.80	12,203	3,747	Vary A
120	45	20	Inv. Square	0.52	7,932	2,436	Vary A
120	45	100	Inv. Square	0.39	5,949	1,827	Vary A
120	45	10	Inv. square	1.00	15,254	4,684	SDEIS
105	15	10	Inv. square	251.39	3,834,703	1,177,511	<i>Orcinus</i>
105	15	8	Inv. square	250.96	3,828,144	1,175,497	
70	25	10	Inv. square	1070.25	16,325,594	5,013,051	<i>Phocoena</i>
70	25	8	Inv. square	1067.49	16,283,492	5,000,123	<i>Phocoena</i>

Table 3. Sensitivity analysis based on a model with spherical spreading for 2 km followed by cylindrical spreading.

B	K	A	Spreading Model	Relative Effect	Humpback takes	Striped Dolphin takes	Basis
80	45	10	Hybrid	132.20	2,016,579	619,225	Vary B
90	45	10	Hybrid	65.31	996,239	305,912	Vary B
100	45	10	Hybrid	25.30	385,926	118,505	Vary B
110	45	10	Hybrid	6.67	101,744	31,242	Vary B
120	45	10	Hybrid	1.00	15,254	4,684	SDEIS
130	45	10	Hybrid	0.08	1,220	325	Vary B
140	45	10	Hybrid	.005	76	23	Vary B
120	5	10	Hybrid	127.23	1,940,771	595,947	Vary K
120	15	10	Hybrid	59.67	910,213	279,496	Vary K
120	25	10	Hybrid	21.39	326,238	100,177	Vary K
120	35	10	Hybrid	5.37	81,901	25,149	Vary K
120	45	10	Hybrid	1.00	15,254	4,684	SDEIS
120	55	10	Hybrid	0.18	2,724	836	Vary K
120	65	10	Hybrid	0.04	570	175	Vary K
120	75	10	Hybrid	0.01	143	44	Vary K
120	45	1	Hybrid	34.16	521,077	160,005	Vary A
120	45	5	Hybrid	3.65	55,665	17,093	Vary A
120	45	8	Hybrid	1.51	23,016	7,067	Vary A
120	45	10	Hybrid	1.00	15,254	4,684	SDEIS
120	45	12	Hybrid	0.73	11,103	3,409	Vary A
120	45	20	Hybrid	0.35	5,353	1,644	Vary A
120	45	100	Hybrid	0.17	2,593	796	Vary A
120	45	10	Hybrid	1.00	15,254	4,684	SDEIS
105	15	10	Hybrid	171.9	2,622,166	805,181	<i>Orcinus</i>
105	15	8	Hybrid	171.3	2,612,718	802,279	
70	25	10	Hybrid	516.41	7,877,318	2,418,864	<i>Phocoena</i>
70	25	8	Hybrid	514.46	7,847,573	2,409,731	<i>Phocoena</i>
80	45	10	Hybrid	132.20	2,016,579	619,225	“Average” species
100	40	10	Hybrid	40.88	623,525	191,464	Stringent criteria
120	45	10	Social75	1.004	15,315	4,703	75% step
120	45	10	Social50	1.06	16,169	4,965	50% step
120	45	10	Social25	1.49	22,728	6,979	25% step
120	45	10	Social10	3.02	46,067	14,146	10% step

An interesting characteristic of the Navy model is that uncertainty causes it to be biased to underestimate risk. The reason for this bias is that the area receiving higher than the level of sound associated with a 50% risk based on default values is smaller than the area receiving lower levels. Thus if a species is 10 dB more sensitive than predicted (the B value), the cumulative risk is underestimated by a factor of 5.68, while if it is overestimated by 10 dB the correction is 0.14. Similarly, if the error is 20 dB, the correction factors are 23.92 and 0.02, respectively. However, the values average to 6.15, not 1 as would be the case if the default values provided an unbiased estimate. Errors in K show a similar pattern.

Likewise, if the default value of A is too low, it makes little difference in the estimated number of takes. However, if the default value of A is higher than the actual value, the effect on the population can be seriously underestimated when default values are used.

It should also be noted that the bias increases with increasing uncertainty.

Another source of uncertainty is propagation. As noted above, there is uncertainty over propagation that depends on the structure of the water column. Expectations can be based on historical measurements, and actual conditions can be measured to allow re-running propagation models with actual conditions. However, when received levels as a function of distance are higher than predicted, the result is asymmetrical relative to an error of the same magnitude in the opposite direction, as is the case for errors in the receiver parameters. E.g., when a sound channel forms, the area receiving enough noise to cause takes will dramatically increase.

Finally, the magnitude of the difference between parameter values based on reanalysis of the datasets used by the Navy (with harbor porpoises added, a species included in the AFAST Draft DEIS, Dept. Navy 2008a), and the Navy analysis should be emphasized. The number of takes predicted for an average species differs by a factor of more than 100. For humpbacks, this suggests individuals would be taken an average of about 250 times. Of course, when refresh times are taken into account, the number of retakes would be below this number, but the duration of takes would go up as a result. The cumulative effect on the population is likely to be far higher with the increased number and duration of takes predicted when more realistic parameters are used than when the Navy parameters are used.

SEL vs. SPL

Studies with captive marine mammals suggest that SEL provides a good predictor of Temporary Threshold Shift. That is, there is a tight relationship among signal strength, duration, and TTS. However, for behaviorally mediated effects, this relationship is likely to be different. SPL is likely to qualitatively determine the response for signals longer than 1 ms in duration. As long as signals are produced sufficiently often, the duration from the first signal to the last is likely to be more important than the SEL. That is, for

low received levels, one second signals produced every 40 seconds for 120 minutes are likely to have more impact than a continuous signal that lasts 10 minutes, even though the latter contains far more sound energy (600 seconds versus 180 seconds), as a behavioral response will be sustained for hours rather than minutes.

When attempting to predict effects of takes on the population, a take table with multiple columns should be developed. One based on SEL could be used to characterize direct effects such as threshold shifts. The next two should be based on SPL. The first of these should be analyzed to evaluate the total number of individuals that would change their behavior as a result of noise exposure, with particular attention paid to exposure in high risk areas (canyons, near shore, near shipping lanes) for potential indirect injuries. The third analysis would consider duration of exposure (in hours of exercise rather than in the SEL sense) to determine whether factors such as stress, displacement from preferred habitat, changes in foraging success and predation risk, etc., would result in cumulative effects that would alter population growth in a manner equivalent to lethal removals (Bain 2002a).

Summary

In summary, development of a function that recognizes individual variation is a step in the right direction. However, the selected equation is likely to produce underestimates of takes. This is due both to social factors increasing the likelihood of a response at low exposure levels, and asymmetries in the number of individuals affected when parameters are underestimated and overestimated due to uncertainty. Thus it will be important to use the risk function in a precautionary manner.

The sensitivity analysis reveals the importance of using as many datasets as possible. First, for historical reasons, there has been an emphasis on high energy noise sources and the species tolerant enough of noise to be observed near them. Exclusion of the rarer datasets demonstrating responses to low levels of noise biases the average parameter values, and hence underestimates effects on sensitive species. In particular, exclusion of the Navy's own interpretation of harbor porpoise data resulted in an increase of B+K by 11 dB, and a reduction in estimated takes by a factor of about 5. Second, uncertainty is correlated with bias. That is, even if a representative set of noise exposure-response data are used to calculate parameter values, the statistical uncertainty resulting from small samples results in biased parameter estimates that lead to underestimation of effects. Thus when estimating takes, it will be important to correct for bias. When estimating population effects on poorly known species, it will be important to be precautionary.

An important error in the selection of parameter values was in interpretation of existing data. Extrapolating behavioral changes in beluga and killer whales and bottlenose dolphins trained to tolerate physical harm that is in their long-term best interest to the threshold for onset of any physical harm in wild individuals is problematic. A similar mistake was made with the right whale data. The level at which 100% of individuals responded was used as the value at which 50% of individuals responded (B+K).

Likewise, the level at which 100% of killer whales responded to mid-frequency sonar is less than the value derived for B+K in the HRC SDEIS (Dept. Navy 2008b).

The “broad overview” of studies reported responses to received levels of 120 dB re 1 μ Pa by 50% of individuals. That is, 120 dB re 1 μ Pa should be taken as a “default” value for B+K, not B. Studies which looked at the level at which statistically significant changes were observed, rather than the level at which 50% of individuals responded found lower levels for B. As a result, B is overestimated, and B+K (the level at which risk is 50%) is as well. The use of data from trained dolphins and white whales biased the average B+K value upward. The exclusion of the effects of AHD’s and ADD’s on harbor porpoises further biases these values, though the sensitivity analysis suggests that using average values to extrapolate takes is unlikely to be accurate due to the broad range of inter-specific variation.

It is likely that biological B values should be in the range from just detectable above ambient noise to 120 dB re 1 μ Pa. The resulting mathematical B value could be tens of dB lower, not the 120 dB re 1 μ Pa proposed. For many species, risk may approach 100% in the range from 120-135 dB re 1 μ Pa, putting K in the 15-45 dB range. A values do not seem well supported by data, and in any case, are likely to be misleading in social species as the risk function is likely to be asymmetrical with a disproportionate number of individuals responding at low noise levels. Re-evaluating the datasets identified by the Navy and including harbor porpoises, an average B+K value of 125 dB was found, and the over-representation of species that fare well in captivity likely biases the average above what it would be for all species. Rather than one equation fitting all species well, parameters are likely to be species typical. As realistic parameter values are lower than those employed in the HRC SDEIS (Dept. Navy 2008b), AFAST DEIS (Dept. Navy 2008a) and related DEIS’s, take numbers should be recalculated to reflect the larger numbers of individuals likely to be taken. The difference between the parameter values estimated here and those used in the SDEIS suggests takes were underestimated by two orders of magnitude.

The large number of takes predicted when more sensitive species are used as sources of the parameters indicates that many individuals are likely to be taken many times, and the potential for population scale effects to result from small behavioral changes becomes significant.

Assuming spherical spreading out to 2 km followed by cylindrical spreading, B=120, K=45 and A=10 (the Navy values), most takes occur where the received level is greater than 157 dB re 1 μ Pa and the distance is less than 13 km. With stringent criteria for what constitutes a take derived in the reanalysis (B=120, K=20, A=10), most takes would occur where the received level is below 145 dB re 1 μ Pa and the distance is over 43 km. With the average values calculated here (B=80, K=45, and assuming A=10), most takes would occur where the received level is below 135 dB re 1 μ Pa and the distance is over 80 km. These values predict over 100 times more takes as the Navy values, as well as the need for very different approaches to mitigation.

The Navy recognizes that the occurrence of conditional effects is important to assessing the impact of noise exposure. As such effects are the result of both received levels and environmental conditions, permit conditions will be important in determining these. The potential for conditional harm suggests using mitigation to limit the potential for actual harm. E.g., the risk of causing stranding can be minimized by restricting exercises to areas far from shore. Limiting the duration of exposure can limit the consequences of long-term displacement, risk of injury from prolonged flight, and limit cumulative effects. The risk of causing gas bubble lesions can be minimized by restricting use near canyons, for extended periods of time, and limiting the number of sources. The absolute effects can be minimized by conducting exercises in areas where population density is low, or at times of year when species of concern are absent.

Finally, it will be important to assess the cumulative effects of noise combined with other factors and population status (Wade and Angliss 1997) to assess the likely effects of sonar exercises on marine mammal populations.

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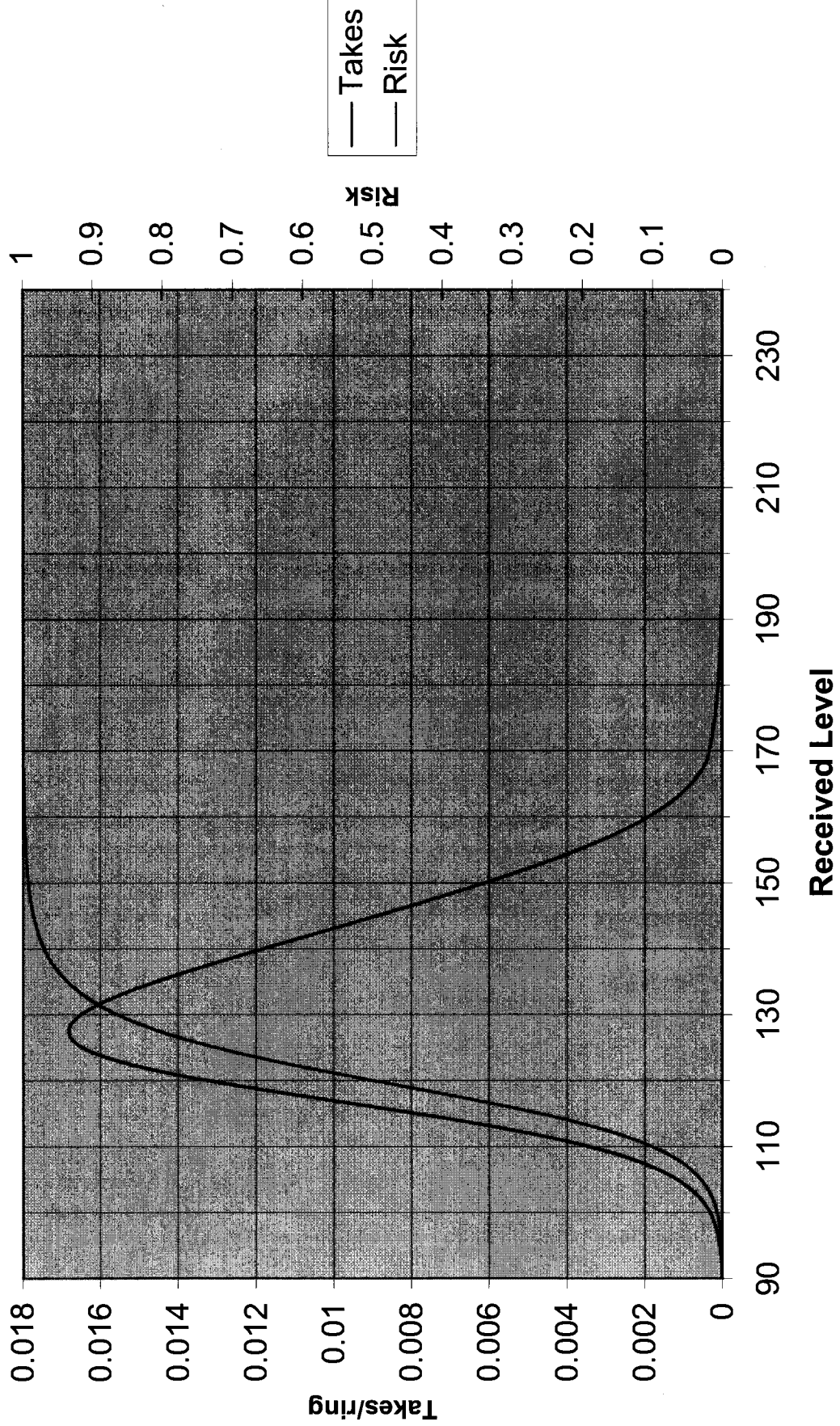
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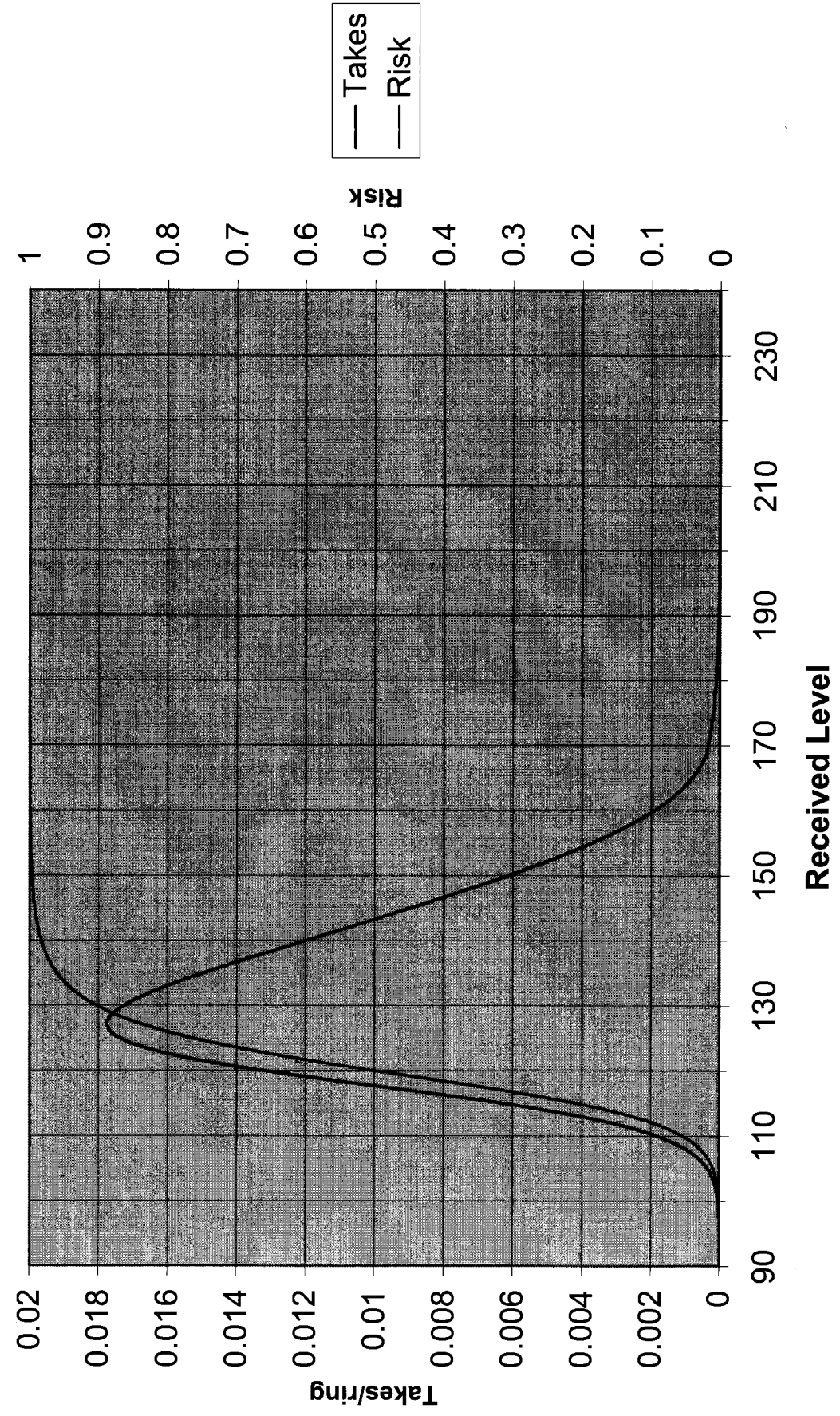
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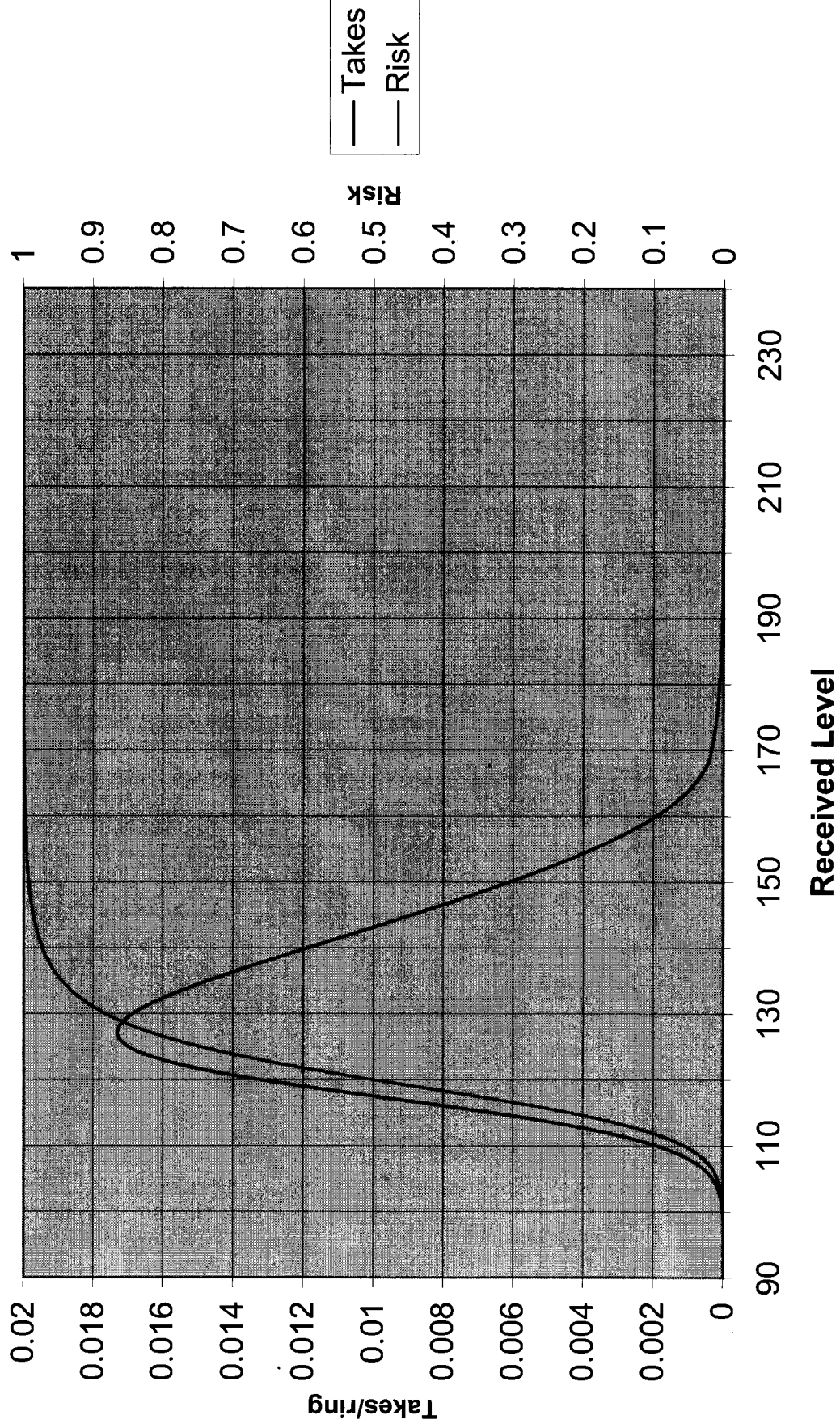
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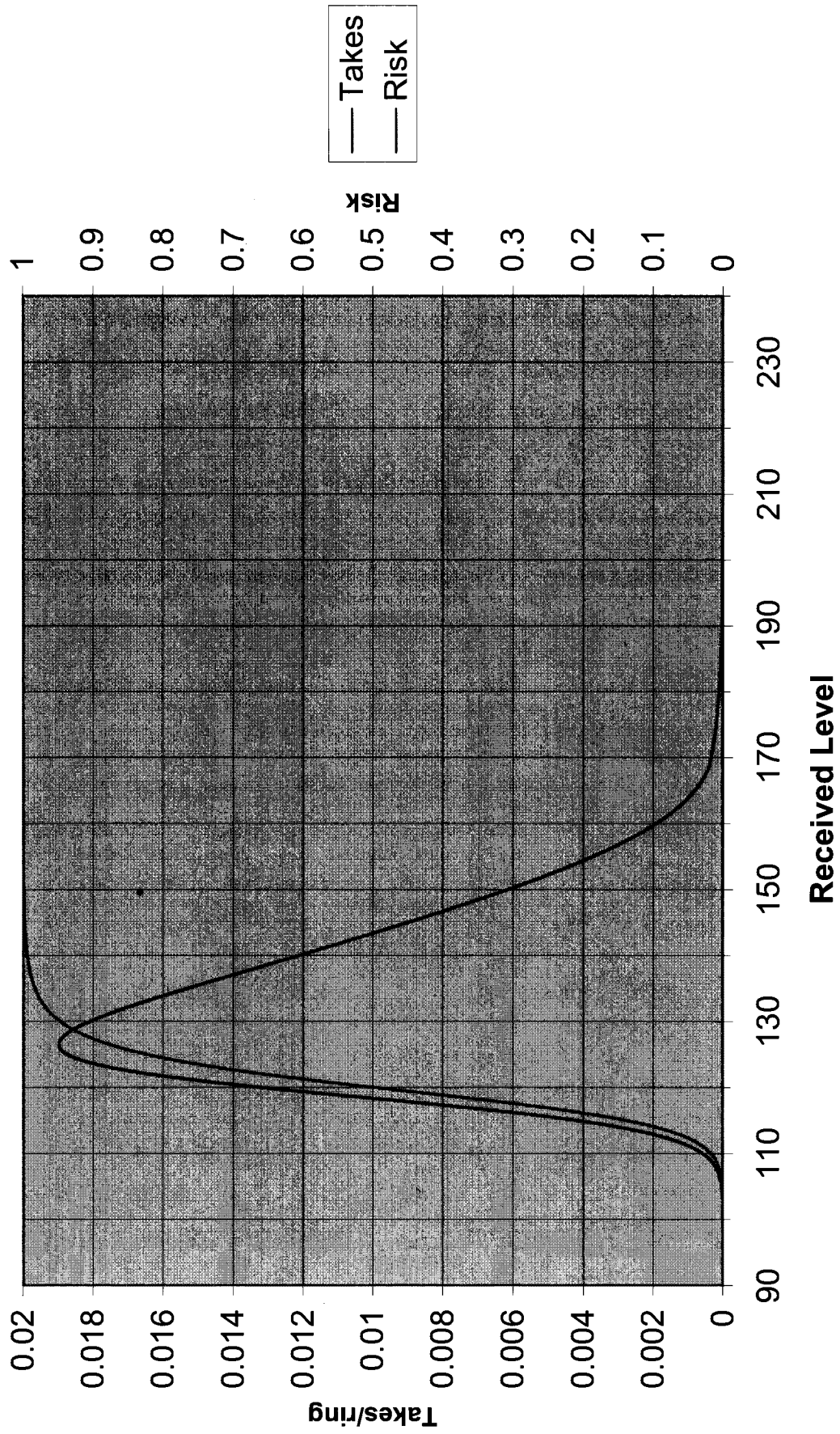
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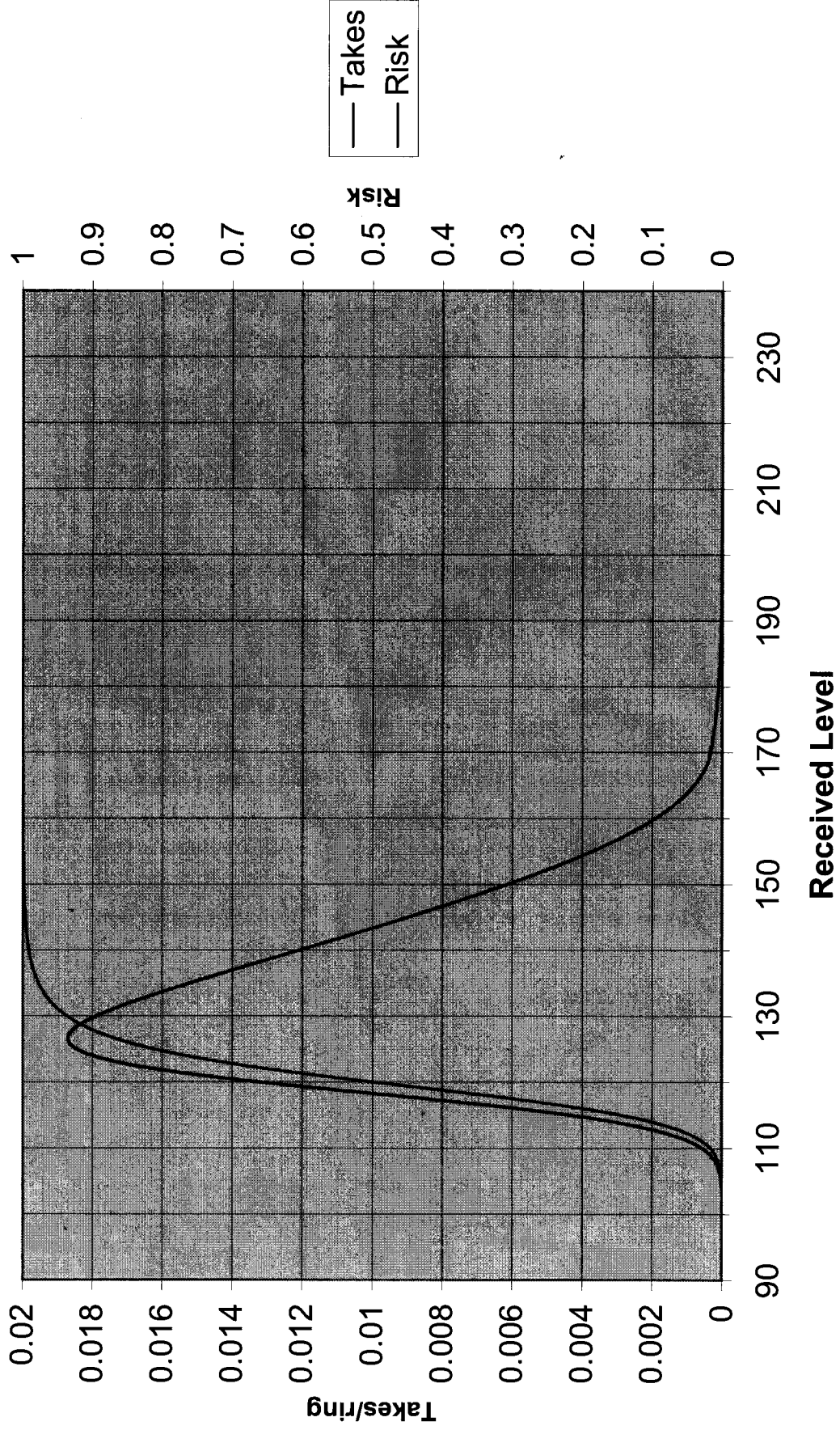
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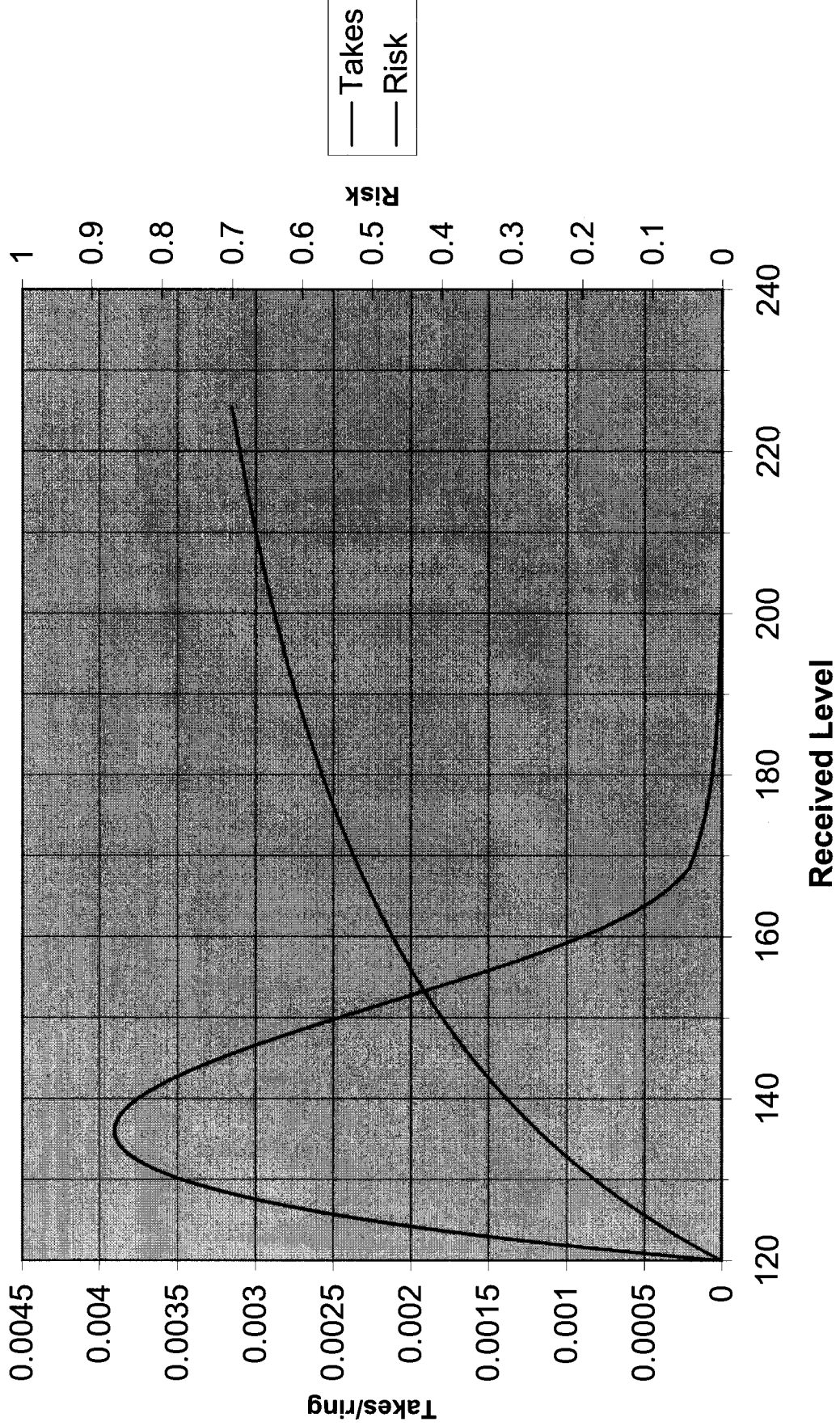
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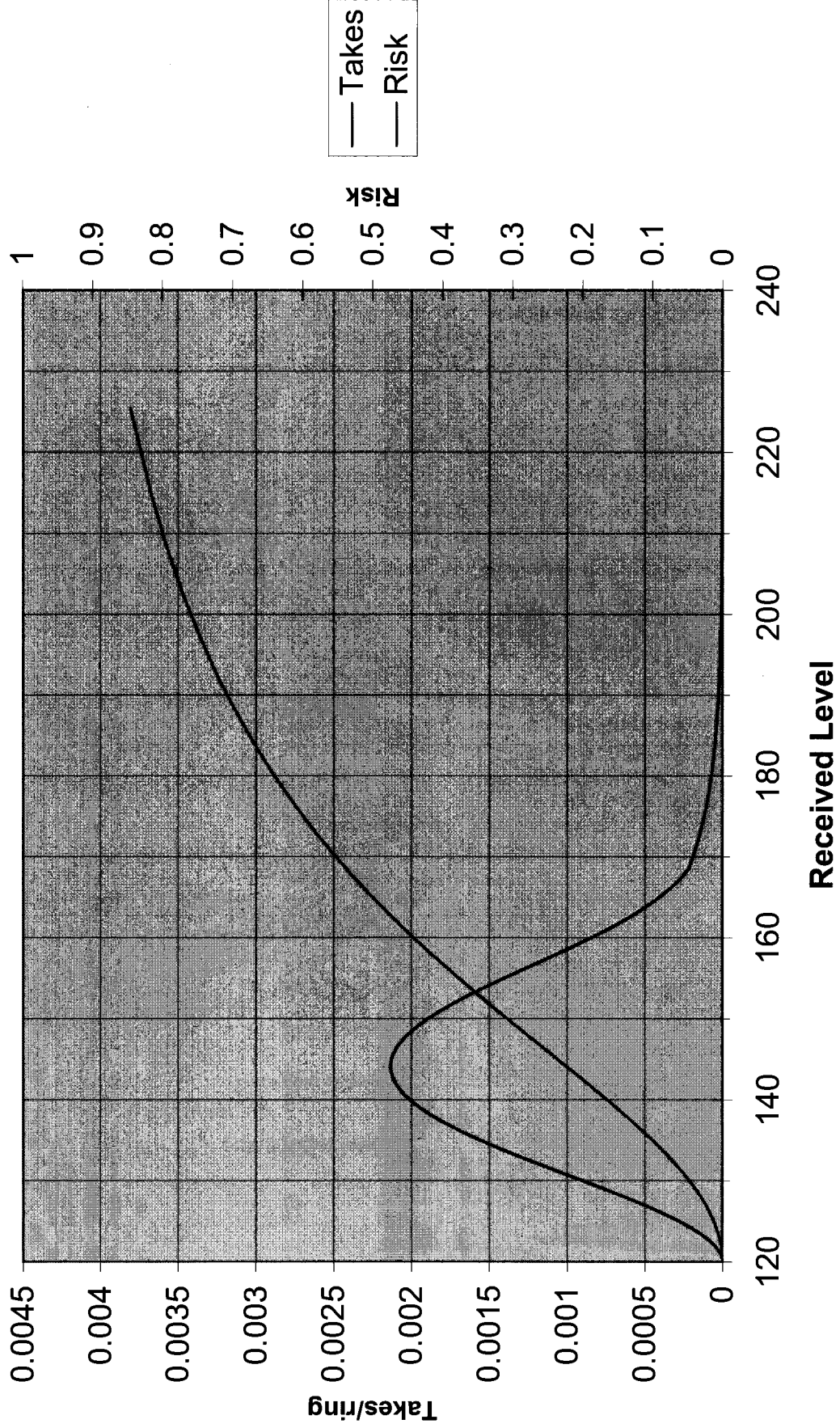
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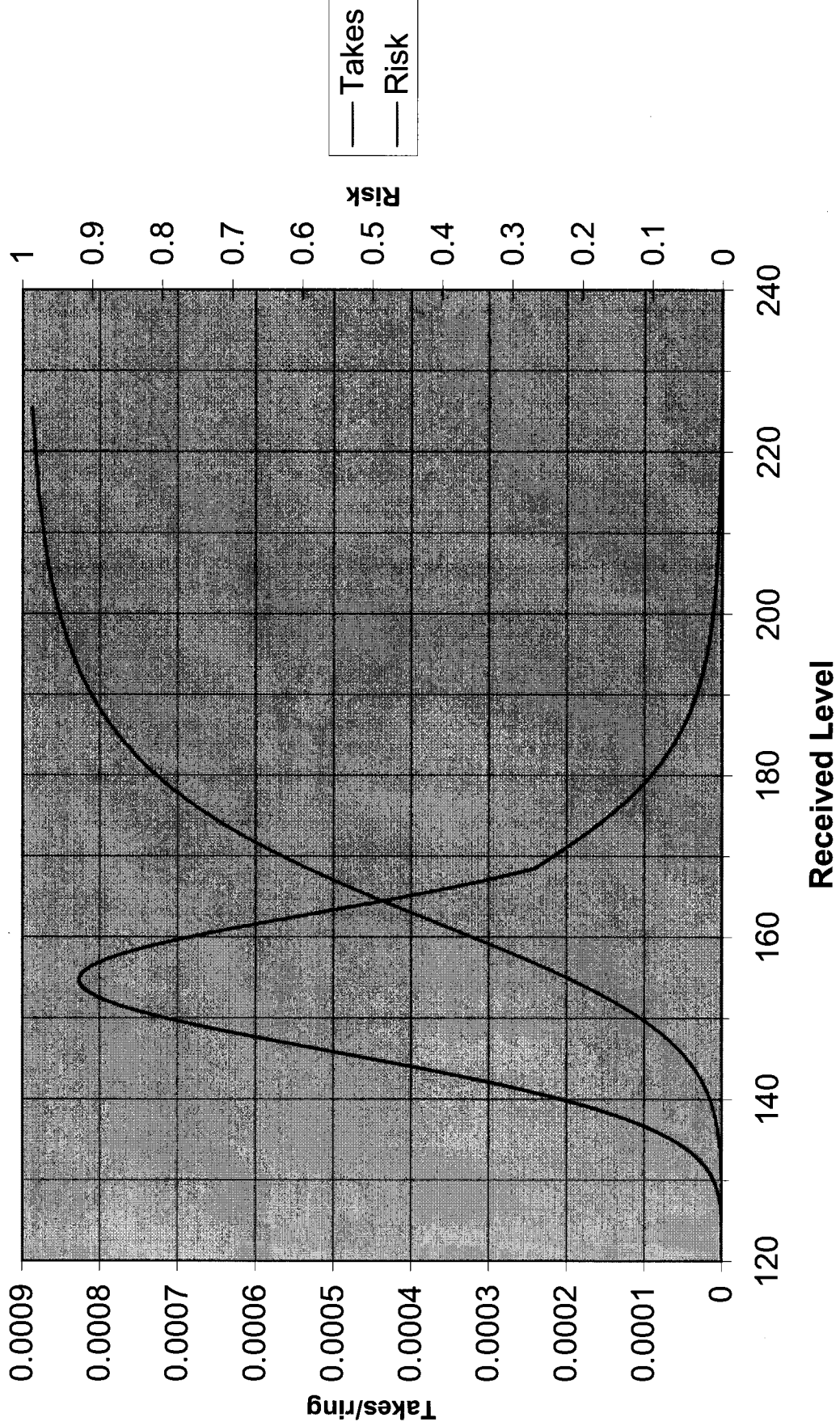
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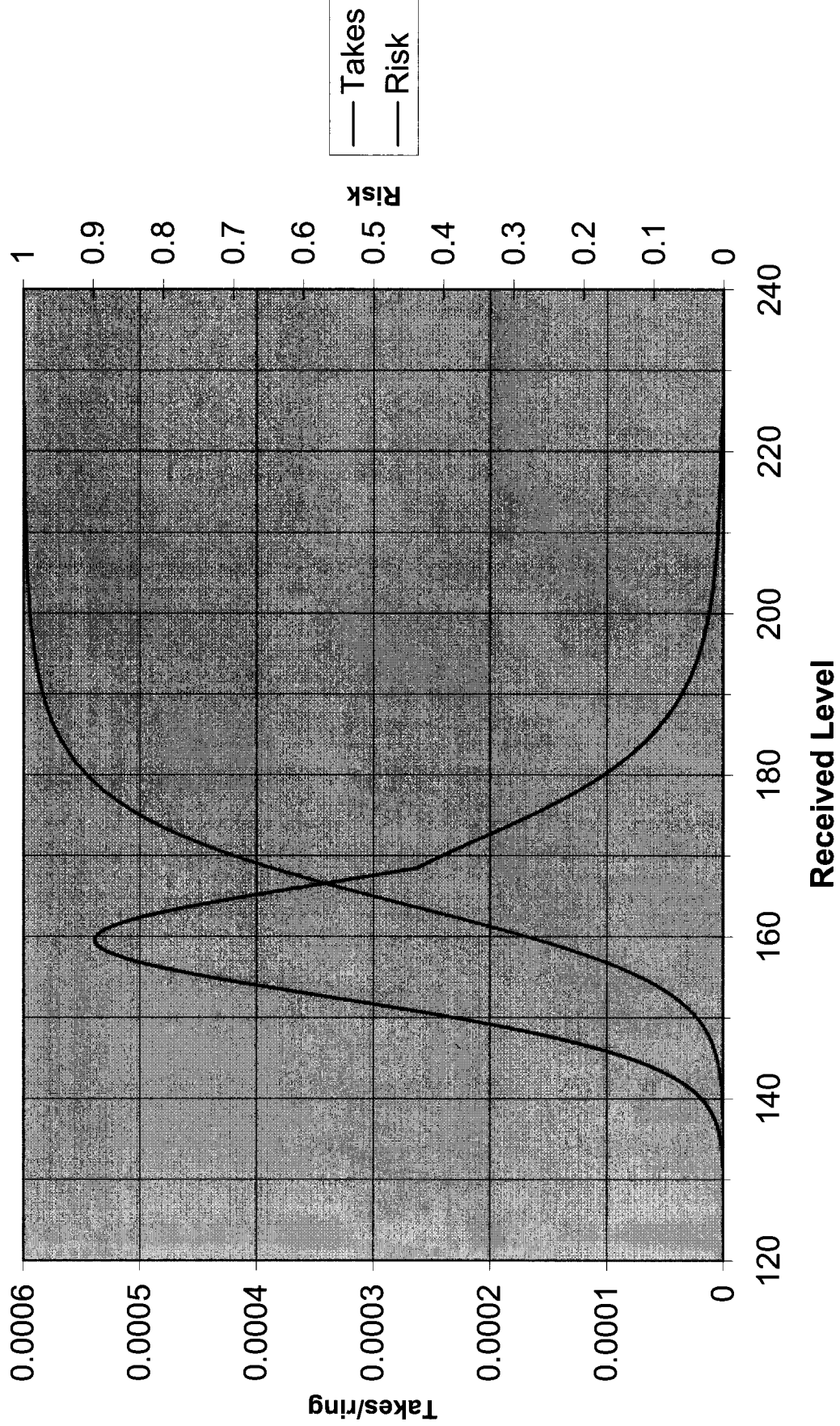
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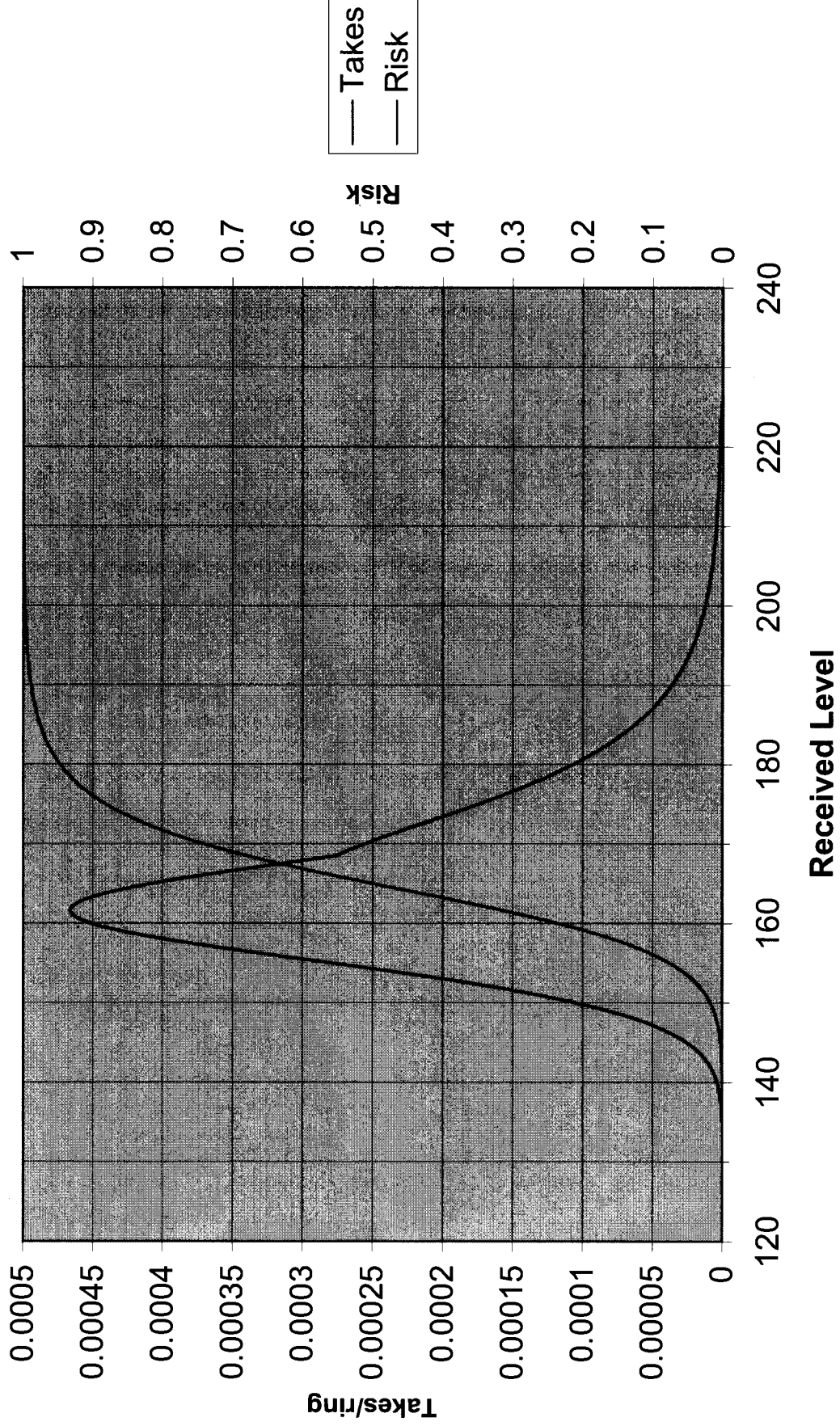
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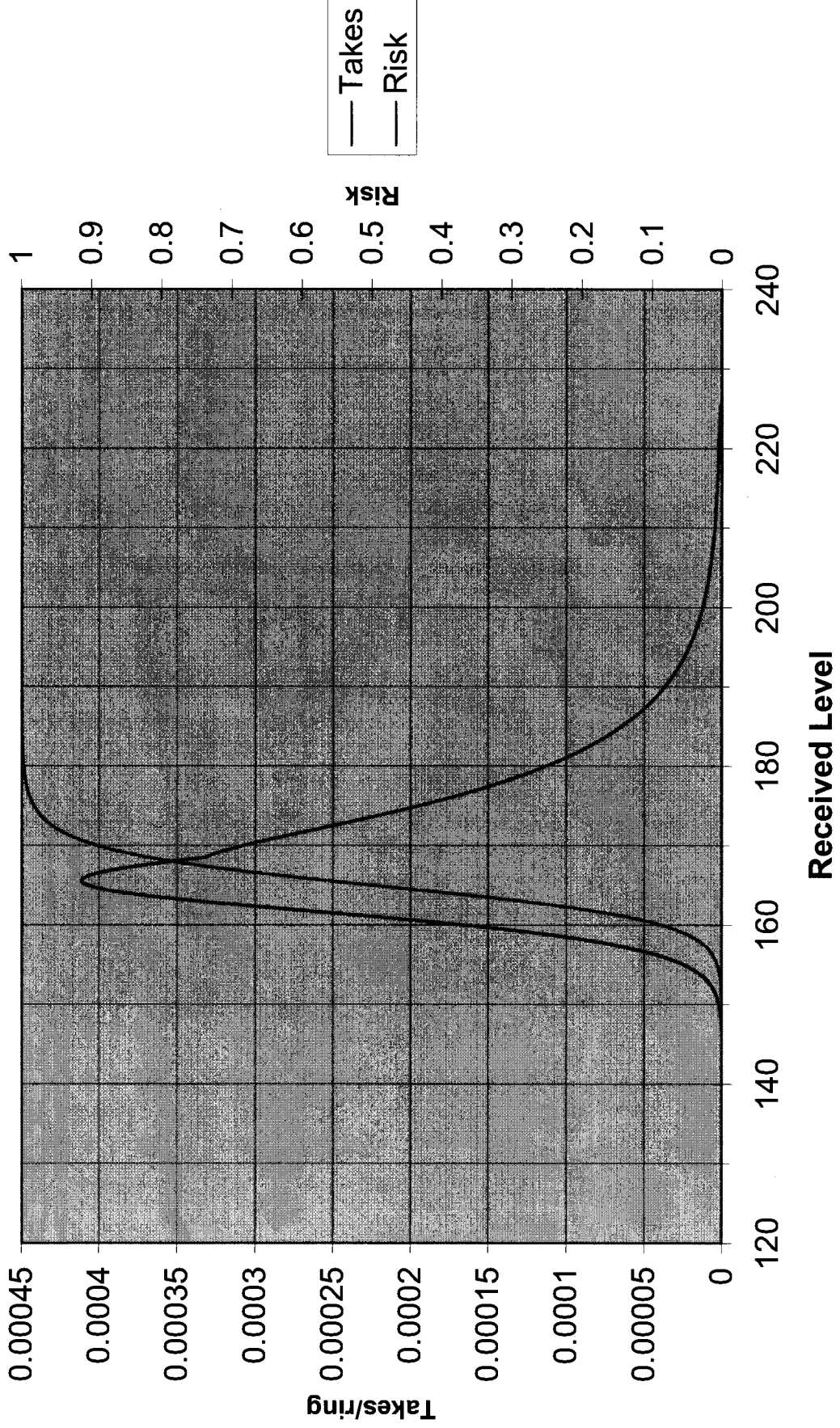
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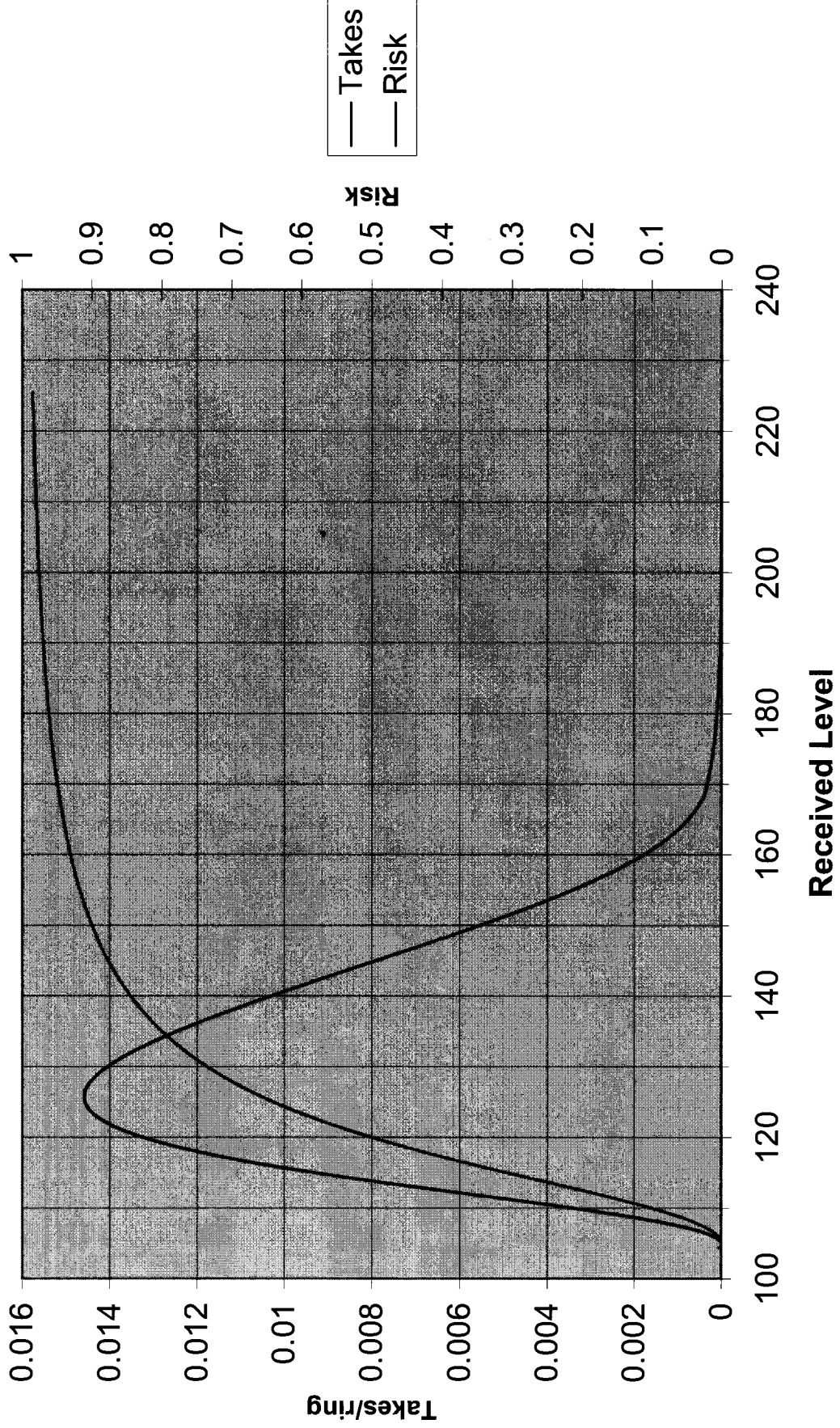
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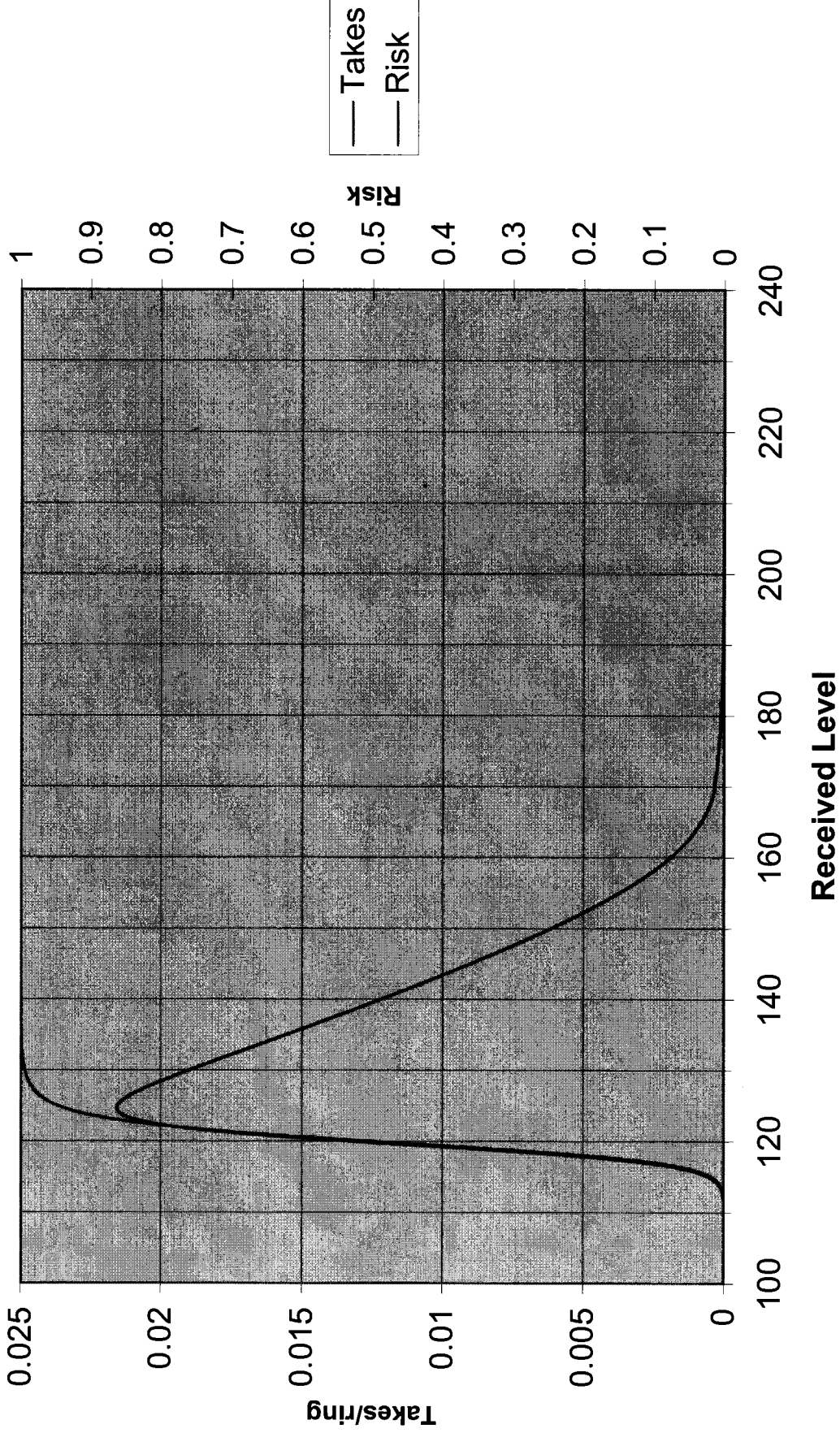
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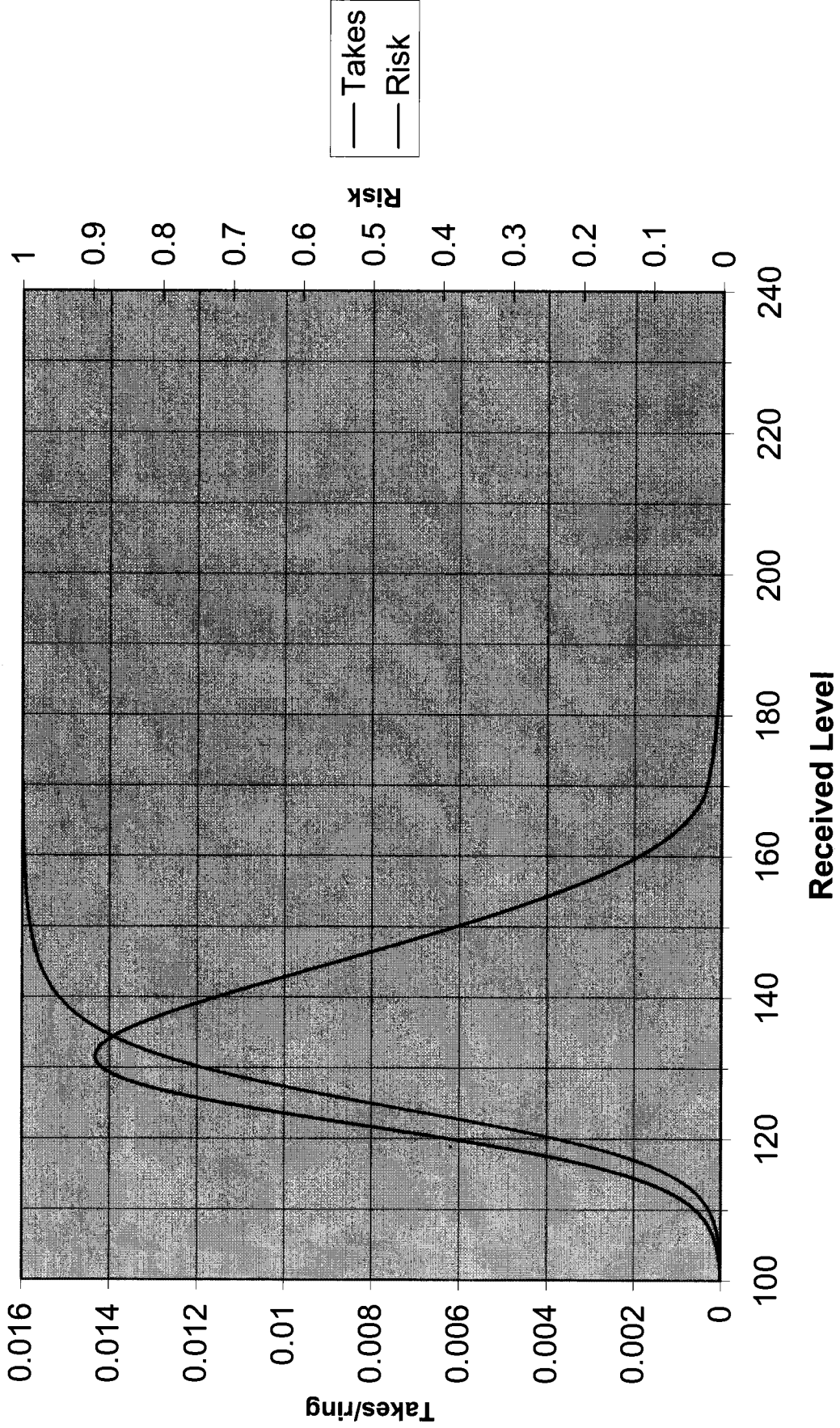
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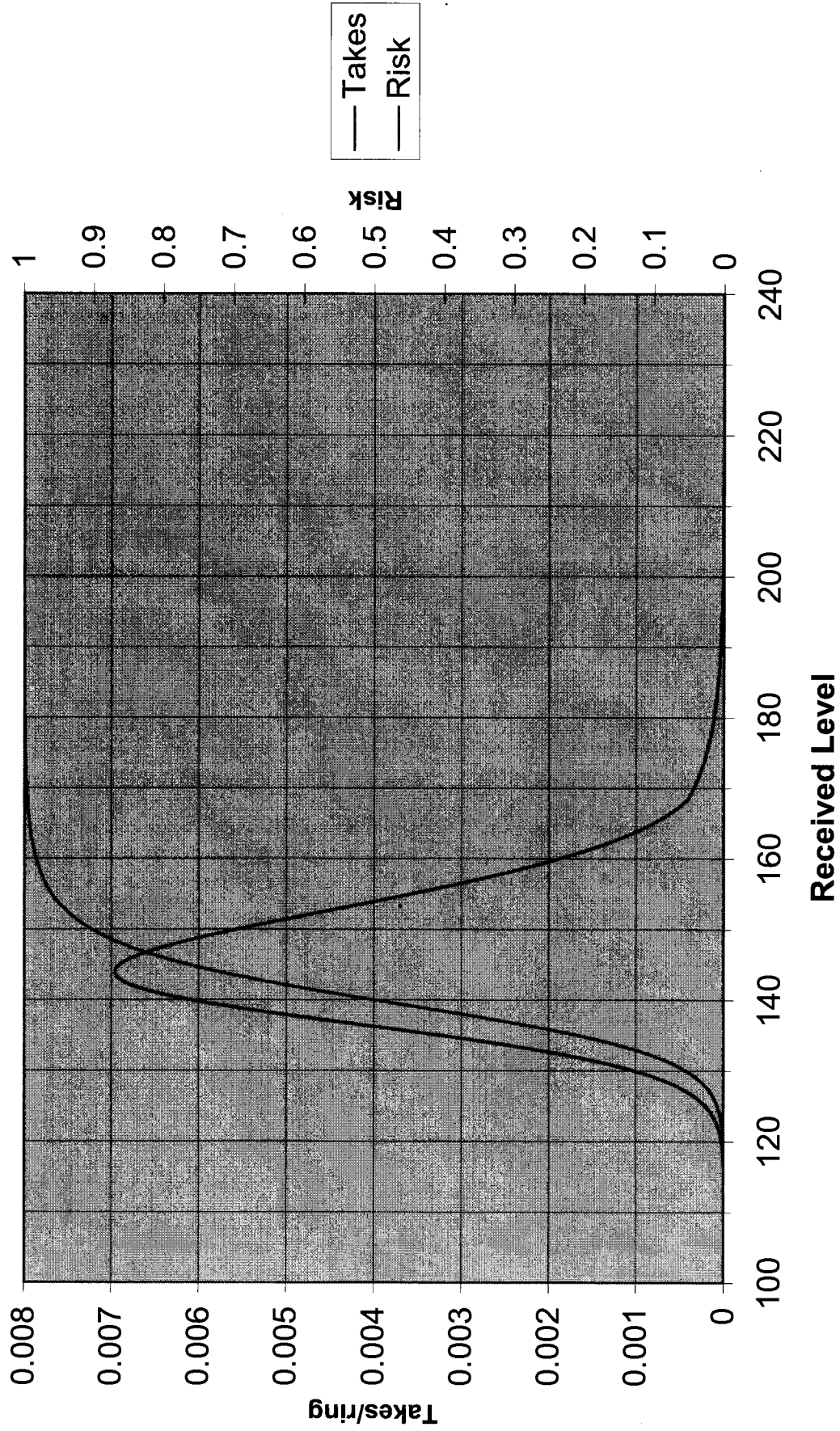
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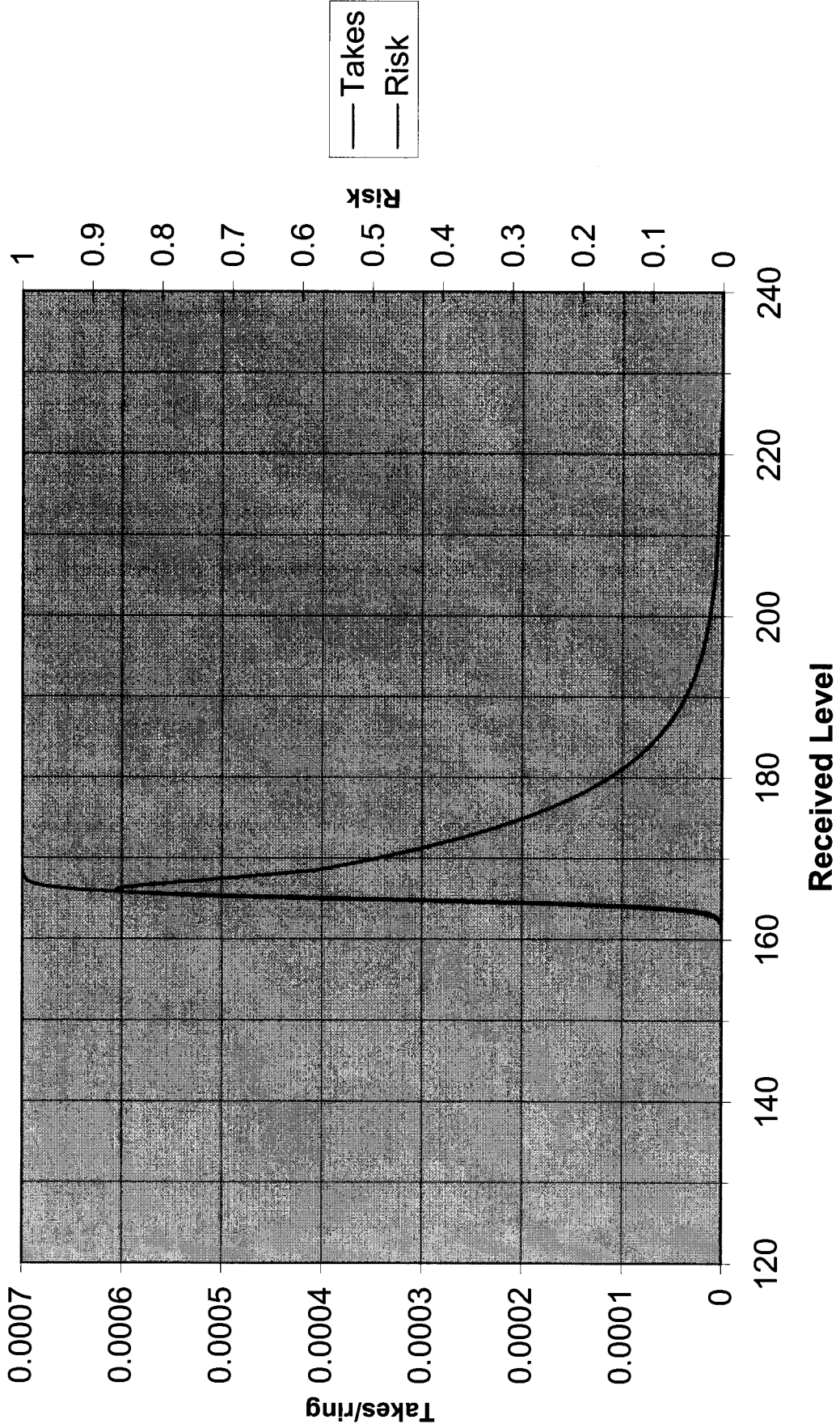
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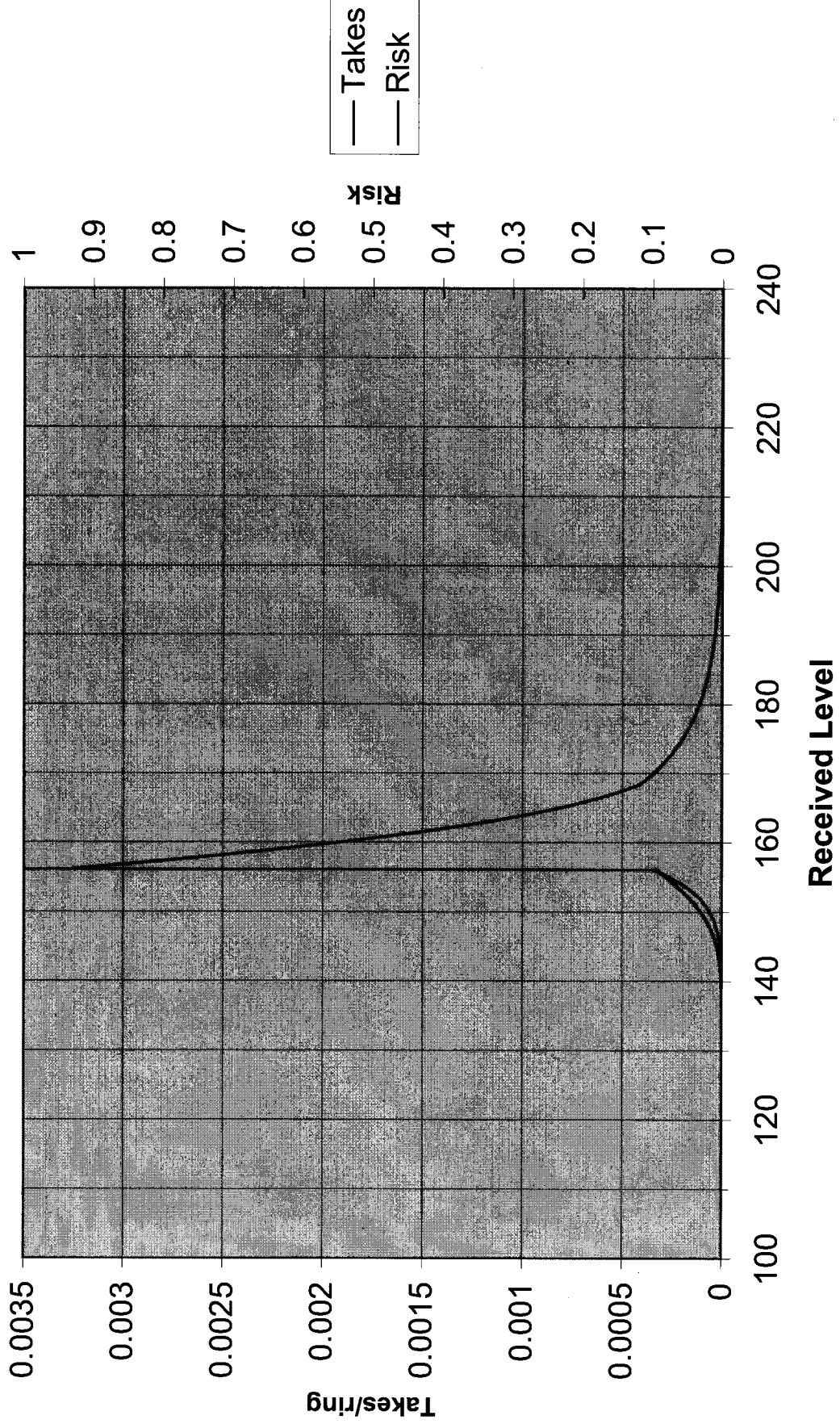
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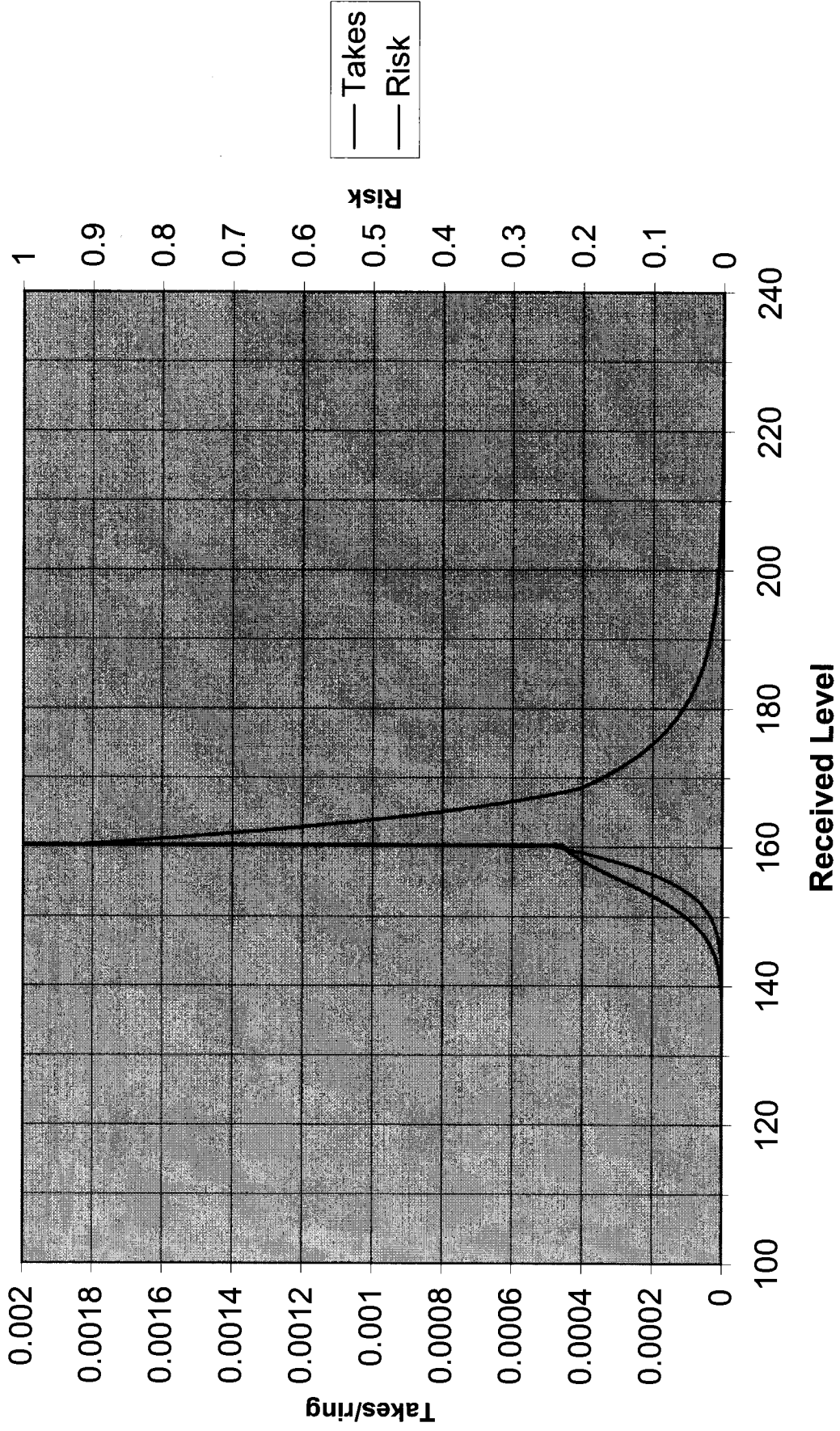
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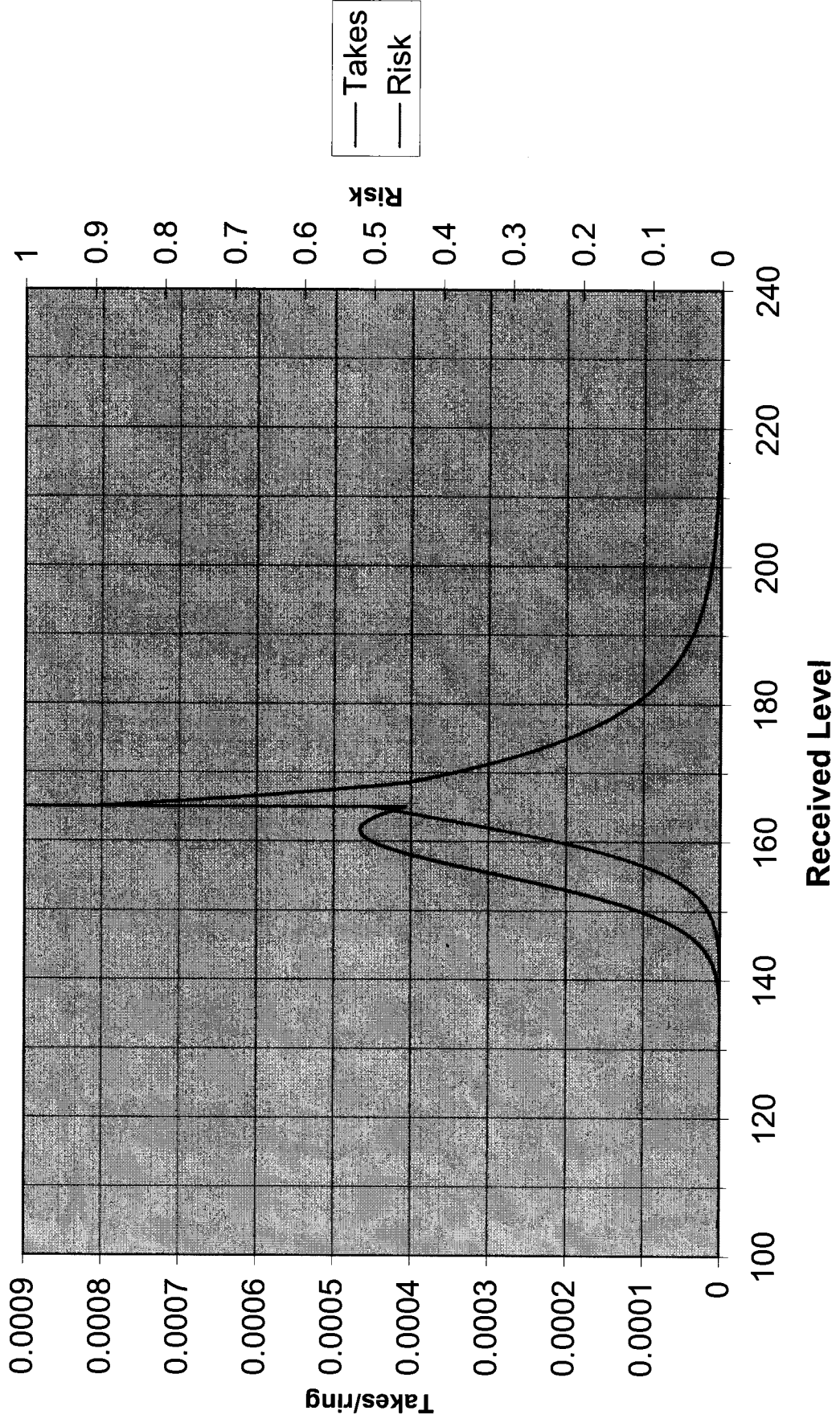
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