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RED IMPORTED FIRE ANT IMPACT ON WILDLIFE: AN OVERVIEW

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ABSTRACT.—We reviewed the past 50 years of literature concerning red imported fire ant impacts on wildlife species. The red imported fire ant has had a negative impact on numerous species of wildlife. However, experimental evidence of population-level impacts is meager. We discuss potential fire ant perturbations and suggest research directions that may settle the long-standing controversy concerning the impact of imported fire ants on native vertebrates and the ecosystems they have invaded. *Key words*: exotic impacts; red imported fire ant; polygyne; *Solenopsis invicta*.

Red imported fire ants (*Solenopsis invicta*) (RIFA) were introduced from the South American Pantanal area into the United States through the port of Mobile, Alabama, in the early 1930s (Vinson and Sorensen, 1986). Despite federal control efforts initiated in 1958 (Hinkle, 1982), over 101,000,000 hectares in the United States are currently infested, and red imported fire ants are expected to infest almost one quarter of the Untied States before range limits are reached (Vinson and Sorensen, 1986).

Introductions of most exotic species fail (Simberloff, 1981), but some succeed and exert impacts ranging from relatively benign to destructive (Elton, 1958; Diamond and Case, 1986). Introduced predators have produced the greatest impact on native ecosystems (Diamond and Case, 1986; Vitousek, 1986). Perturbations by exotics on one trophic level of an ecosystem may indirectly cause changes at other levels (Carpenter et al., 1985).

A native fire ant (*S. geminata*) is considered a keystone predator, suppressing herbivores and predators in some habitats (Risch and Carrol, 1982). The red imported fire ant is an aggressive congener known to disrupt arthropod communities in the United States (Porter and Savignano, 1990).

The first reports of RIFA-caused mortality in vertebrates surfaced in the United States during the 1940s. Recent occurrences of RIFA colonies with multiple fertile queens (Glancey et al., 1973), and the high mound densities associated with these polygyne colonies, have renewed interest in RIFA impacts on native vertebrates and compounded the problems created by fire ants.

A mature colony of RIFAs may contain 200,000 or more individuals (Lofgren and Vander Meer, 1986), and mound densities may reach 1,400 colonies/hectare (Maxwell et al., 1982). When occurring at high densities, RIFAs dominate food sources (Vinson, 1990), other ants are outcompeted (Vargo and Porter, 1989; Porter et al., 1988; Vinson,

1990), native ant and other arthropod diversity decreased (Porter and Savignano, 1990), and vertebrates may be negatively affected (Maxwell et al., 1982).

A taxonomic history is beyond the scope of this paper, but it is necessary to clarify a point of confusion. Prior to the 1970s the taxonomic status of the imported fire ants was unclear (Buren, 1972). Two species of imported fire ant are now recognized, the red, and the black (*S. richteri*). Today *S. richteri*, outcompeted and displaced by *S. invicta*, occupies a limited range in northeast Mississippi and northwest Alabama, whereas *S. invicta* occurs throughout the remainder of the infested area. It is impossible to distinguish between the two species using references prior to the early 1970s.

Although current research focuses on many RIFA-vertebrate interactions, review and synthesis of literature on these interactions are lacking (Lutz and Demararis, 1990). Here we review and synthesize information published since 1948 concerning RIFA impacts on reptiles, birds, and mammals.

Impacts on Reptiles

Several reptile species in the southern United States are declining in all or parts of their range. RIFAs are often suggested as a contributing factor. For example, the Texas horned lizard (*Phrynosoma cornutum*) has been largely extirpated from eastern Texas (Price, 1990), in an area that coincides with the RIFA range in the state, and the RIFA has been suggested as a causal factor (Price, 1990; Goin, 1992). The invertebrate diet of many reptiles may overlap with the diet of RIFAs. Moreover, reptile life histories often include an egg stage vulnerable to RIFA predation.

Landers et al. (1980) reported RIFAs as predators on the hatchlings of the gopher tortoise (*Gopherus polyphemus*) in southwestern Georgia. Monitoring 38 nests in eight tortoise colonies, the authors discovered that mortality from various sources was high and only four nests (11%) survived incubation intact. While the majority of nest destruction occurred during incubation, RIFA-induced mortality occurred at the hatchling stage. Ten hatchling tortoises were killed by RIFAs.

Mount et al. (1981) presented convincing experimental evidence that RIFAs would attack and consume eggs of the six-lined racerunner (*Cnemidophorus sexlineatus*) in Alabama. Mount (1981) speculated that several reptile species were declining as a result of RIFA predation.

Many amphibian and reptile species may be vulnerable either to direct or indirect RIFA impact due to shared food habits with RIFAs and an egg stage vulnerable to RIFA predation. Difficulites with *in*

situ herpetological studies make assessment of RIFA impact on amphibian and reptile populations formidable.

Impacts on Wild Birds

The literature contains numerous references to RIFA predation on birds. Ridlehuber (1982) suggested that high RIFA foraging use of nesting cavities (70%) precluded use by wood ducks (*Aix sponsa*), and observed RIFA predation on wood duck nestlings and pipped eggs. Sikes and Arnold (1986) monitored cliff swallows (*Hirundo pyrrhonota*) in east-central Texas and found nest success was 41% with RIFAs present, compared with 75% success when RIFAs were controlled in nesting areas the following year. The authors considered the RIFA to be a major problem for cliff swallows in the vicinity of their study area.

Wilson and Silvy (1988) included a number of observations of predation on individual birds, including Attwater's prairie chicken (*Tympanuchus cupido attwateri*), northern bobwhite (*Colinus virginianus*), and scissor-tailed flycatcher (*Muscivora forficata*). Mount (1981) attributed the decline of the common nighthawk (*Chordeiles minor*), common ground-dove (*Columbina passerina*), and eastern meadowlark (*Sturnella magna*) to RIFA predation in southern Alabama. Hays (1980) observed nine cases of fire ant predation on least tern (*Sterna albifrons*) chicks. Interest in the impact of the RIFA on nesting least terns remains high, and additional research is underway (T. Lockely, U.S.D.A., APHIS, S&T, Imported Fire Ant Lab, Gulfport, MS, pers. comm.). Others mention incidental RIFA predation on various avian species (Rankin, 1957; U.S. Agric. Res. Serv., 1958; Hays and Hays, 1959; U.S. Dept. Agric., 1968; Ashdown, 1969; Lofgren, 1986; Lofgren and Vander Meer, 1986).

Red imported fire ants on offshore barrier islands in the ship canal of Houston, Texas, were manipulated to determine the impact on nesting shore birds (e.g., great egrets, *Casmerodius albus*) (B. Drees, Texas A&M Univ. Agric. Exp. Stn., pers. comm.). Preliminary results indicated that during times of low fire ant activity (cool weather), RIFA impact on nesting birds was minimal, but predation on nests of late-nesting individuals was high.

Interest and controversy concerning RIFA predation on northern bobwhite continues. Fire ants were first reported as a source of quail mortality by Stoddard (1931) and Travis (1938*a*,*b*). Travis (1938*a*) documented yearly (1924-1937) native fire ant predation on bobwhites ranging from one to nine percent ($\bar{X}$=6.5%), and speculated that the decline of the ground-dove in Florida was attributable to fire ants.

Other researchers have de-emphasized the role of fire ants as a predator of quail (Brennan, 1991; Komarek, 1980). Johnson (1961)

concluded, despite observing a six percent RIFA-induced quail chick mortality, that RIFAs had little, if any, effect on quail populations. Dewberry (1962) reported that 12% of hatching quail eggs were destroyed by fire ants, but noted that incubating birds were undisturbed by ants. Simpson (1976) observed only one of 1,725 quail nests in Georgia destroyed by fire ants. Rankin (1957), Wheeler (1958), Ashdown (1969), and Sterling and McDaniel (1978) mentioned imported fire ants as potential quail predators.

Impacts on Mammals

Deaths of wild mammals due to the RIFA first surfaced as anecdotal reports in the late 1940s. Biologists working with landowners in RIFA-infested areas regularly hear of blinded or dead white-tailed deer fawns (*Odocoileus virginianus*) (Rollins, 1989) or other mammals apparently harmed by the RIFA. These incidences, however, are rarely definitive evidence of RIFA-induced mortality in mammals.

Bruce et al. (1949) included the first report of mammal losses due to the RIFA, specifying the loss of newborn rabbits (*Sylvilagus* sp.) and squirrels (*Sciurus* sp.). Ten years later (U.S. Agric. Res. Serv., 1958) it was stated that there were no published reports of RIFA predation on birds or other animals, yet this report again cited the RIFA as a "destroyer" of newborn small mammals. A decade later RIFAs were still implicated in the deaths of newborn rabbits and squirrels (Ashdown (1969).

Ashdown (1969) included an account by a rural Florida couple who reported an increase in quail and wild rabbit populations following fire ant treatment with Mirex®. Reported wildlife kills increased in Georgia in the 1960s following RIFA population increases (Irvin, 1980). By 1970, public sentiment had turned against the use of Mirex® for RIFA control, and Coon and Fleet (1970) published a paper decrying the use of Mirex®. To support their stance, they de-empahsized the effect of the RIFA on mammals and other animals. At that time, little experimental evidence documented RIFA-wildlife interactions, and arguments concerning effects of RIFAs on wildlife populations were moot.

Hill (1969, 1972) quantified the loss of captive nesting cottontail rabbits (*S. floridanus*) to imported fire ants in Alabama from 1963-1967. Of 101 litters born in the 15 x 15 meter pens, 16% were destroyed by fire ants. In 61 x 61 meter pens, 81 litters were observed and 50% were destroyed. Of 43 litters born in large enclosures, 23% were destroyed. Hill observed that young cottontails and *Rattus norvegicus* were most vulnerable from birth to four days old. After this age the young mammals were more mobile and had enough fur to prevent stinging over large portions of their bodies.

Newsom et al. (1976) published the results of a study on fire ant-muskrat (*Ondatra zibethicus*) relationships in Louisiana coastal marshes. No significant association was discovered, but most muskrat houses was examined between reproductive peaks.

Fire ant-induced mortality of live-trapped small mammals has been documented by Masser and Grant (1986) and Flickinger (1989). One of 33 hispid cotton rats (*Sigmodon hispidus*), nine of 38 pygmy mice (*Baiomys taylori*), three of 23 white-footed mice (*Peromyscus leucopus*), and zero of eight fulvous harves mice (*Reithrodontomys fulvescens*) were killed by RIFAs, averaging 13% mortality overall (Masser and Grant, 1986). Checking traps at three-hour intervals did not reduce mortality. Flickinger (1989) recorded 20 to 225 (9%) captured hispid cotton rats killed by RIFAs despite attempts to reduce attractiveness of live traps. Additionally, 13 rats (6%) were covered with ants and partially devoured. Hill (1972), Masser and Grant (1986), and Flickinger (1989) indicated that RIFA-induced mortality was less during cool seasons and cold weather.

Smith (1988) and Smith et al. (1990) documented the avoidance of areas with high fire ant densities by northern pygmy mice (*Baiomys taylori*), and suggested similar avoidance patterns for some other rodent species. Grant and Killion (1988) included a number of observations of individual mammals killed by the RIFA. Killion et al. (1990) presented preliminary data which suggested that hispid cotton rats avoided areas of high fire ant density. The authors offered possible reasons for avoidance behavior, such as lowered reproductive success due to RIFA predation on nestlings, competition for food, or simple "harassment" caused by RIFA stings.

Implications and Research Needs

Sound management decisions are conditional on quantification of RIFA effects on wildlife at the population level, but the effect of RIFA on wildlife populations is poorly understood. Consequently, we can only speculate on management implications and suggest research needs that will lead to a better understanding of fire ant influences on wildlife.

The RIFA has the potential of impacting many vertebrate and invertebrate species. RIFAs favor open and semi-open habitats; preferences that are shared with many wildlife species. Foraging activity of the RIFA is temperature dependent, with most foraging activity occurring when soil temperatures are between 22-36° C at a depth of two centimeters (Porter and Tschinkel, 1987). This temperature range coincides with peak reproductive activity of many vertebrate species. Fire ants are attracted to mucous (Vinson and Sorensen, 1986), and newborn animals may be vulnerable to RIFAs.

Moreover, heavy RIFA foraging may preclude use of some habitats by vertebrates (and invertebrates) because of irritation from RIFA stings.

Red imported fire ants are efficient opportunist foragers and may outcompete both vertebrates and invertebrates for favored food sources. RIFAs simplify invertebrate communities (Porter et al., 1988; Porter and Savignano, 1990). Key native elements of the food chain may be supplanted by the RIFA, with potential negative effects on native species not directly impacted and on ecosystem properties.

While we know that individuals of many species may be killed by the RIFA, we do not know the effect of RIFAs on vertebrates at the population level. Additional passive anecdotal evidence and experiments of small scope will not provide an understanding of the complex interactions between vertebrate populations and RIFAs. To gain an understanding of the effects of the RIFA on vertebrates we need long-term comprehensive ecological studies that encompass populations of target vertebrates, and are conducted with controls and adequate temporal and spatial replication (Hurlburt, 1984). Small-scale studies do not adequately account for animal behavior in natural habitats where RIFA density is seldom homogeneous, but instead is a mosaic of areas having high and low density infestations. Consequently, a multi-disciplinary approach (e.g., entomologists and wildlife biologists) is needed to address potentially confounding effects, such as the reinfestation of treated monogyne areas with polygyne RIFA colonies. Although comprehensive ecological studies pose logistical problems (Eberhardt and Thomas, 1991), they are the most likely to generate useful data.

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