REVIEW OF POTENTIAL EDGE EFFECTS ON THE SAN FERNANDO VALLEY SPINEFLOWER

(*Chorizanthe parryi* var. *fernandina*)

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INTRODUCTION

The recent discovery of the San Fernando Valley spineflower (*Chorizanthe parryi* var. *fernandina*) on the Ahmanson Ranch project site in Ventura County, California prompted preliminary investigations into the biology of that taxon. The purpose of these studies is to develop a conservation strategy to protect, maintain, manage, and, possibly, reintroduce the spineflower into appropriate habitat. While the proposed development would remove a portion of the spineflower population, the majority of the known population is proposed to be conserved onsite. Residential development is planned adjacent to the proposed spineflower preserve area.

An effective conservation strategy should emphasize preserve design and habitat and species management. Accepted principles of preserve design include maximizing the width of the buffer between development and sensitive resources, minimizing habitat loss, fragmentation, and edge effects, maximizing genetic diversity and connectivity with other habitat patches, maintaining adequate habitat to allow for spatial and temporal population fluctuations, and maintaining a sustainable population size,[1] among others. Habitat and species management may be necessary to mitigate impacts from adjacent development and to maintain the functions and values of the population being conserved.

This paper assesses potential impacts to the conserved spineflower population from adjacent development based on a review of the scientific literature on edge effects (adverse effects of land uses on adjacent biological resource areas, such as weed invasions or changes in hydrology). A thorough literature search on edge effects has not been conducted for this paper due to time limitations. The summary presented herein is intended to (1) focus on potential impacts to sensitive plant species, and (2) address those risk factors associated with edge effects most likely to affect the spineflower, based on current knowledge of the species' biology. All identified risk factors have the potential to negatively impact some aspect of the species' biology or habitat; however, information is not yet available to definitively determine which factors pose the most serious threat to the species' persistence. This paper analyzes identified risk factors in relation to preserve design and proposes management actions and alternative scenarios to minimize or reduce the potential impacts of these risk factors.

SPINEFLOWER BIOLOGY

The biology of a species holds implications for preserve design and habitat management. Additional research is needed to assess the long-term viability of the spineflower population on Ahmanson Ranch and to identify specific management measures to ensure its persistence. This section summarizes our current knowledge of spineflower biology and limitations to our knowledge.

[1] Note that a sustainable population is not measured by species presence alone, but by the *effective* size of a population in contributing to future generations relative to an *ideal* population. The effective population size may be smaller than the census population number. Estimates of effective population size may be determined through demographic monitoring or genetic studies (Barrett and Kohn 1991).

San Fernando Valley spineflower is a small annual plant in the buckwheat family (Polygonaceae). This low-growing species is characterized by prostrate to ascending stems, small white flowers, and straight involucral awns (Hickman 1993). Historical habitat for the San Fernando Valley spineflower was apparently deep, low nutrient soils of sand benches, or soils with similar characteristics that occurred as mosaics within coastal sage scrub and, possibly, valley grassland (GLA 1999). Although soils on the property are generally well drained, acidic, and low in nitrogen and organics (GLA 1999), preliminary studies indicate that the spineflower population on Ahmanson Ranch occurs in open areas on compacted or recently disturbed soils that support few other plant species. It is unclear whether this association indicates that the spineflower prefers compacted soils, or if it is restricted to compacted soils by competition from other plant species that avoid the compacted soils. Also, it is not clear whether the density of spineflower plants differs between compacted and non-compacted soils. If spineflower densities are lower than normal on compacted soils, this may have long-term genetic consequences if spineflower populations are restricted to compacted areas in the future, either by preserve design or lack of effective management to reduce competition from other species. It has been suggested that lowered plant density has the same effect on reproductive success as small population size (Lamont et al. 1993; van Treuren et al. 1993; Groom 1998) for some insect-pollinated plants. Theoretical models that have included population density or size as input factors indicate that extinction rates increase dramatically as density declines, and extinction becomes almost inevitable below certain density thresholds (Dennis 1989 in Groom 1998; Kunin and Iwasa 1996 in Groom 1998). Groom (1998) documented that small patches of an annual herb suffered reproductive failure due to lack of effective pollination when critical thresholds of isolation were exceeded. In contrast, large patches attracted pollinators regardless of the level of isolation.

San Fernando Valley spineflower most likely forms a persistent seed bank in the soil, with seeds germinating under specific climatic conditions (e.g., appropriate temperature and amount and timing of rainfall). Seed banks typically contain multiple genotypes from various years, but years yielding large seed crops contribute disproportionately to the bank (Templeton and Levin 1979). In this respect, seed banks contain the "evolutionary memory" of a species (Del Castillo 1994). Seed banks buffer changes in population size, and help maintain genetic diversity and genetic spatial distribution (Del Castillo 1994). Seedling survival may depend on adequate rainfall, as well as light and nutrient conditions. These factors influence the degree of competition between the spineflower and other plant species.

Little is known about the reproductive biology of the spineflower (including whether the species is strictly outcrossing or can also self-pollinate). A wide range of insect visitors was observed on spineflower flowers during the 1999 field surveys (GLA 1999), but it has not yet been determined if any of these are effective pollinators. Insects observed on the spineflower included ants (mostly of the *Dorymyrex insanus* complex), ant-like spiders (possibly *Micaria* spp.), European honeybee (*Apis mellifera*), bee-flies (Bombyliidae), a small bumblebee (*Bombus* sp.), and tachnid flies (possibly *Archytas* spp.) (GLA 1999). Of these species, the ants appeared to be the most frequent flower visitors. Determination of the reproductive strategy is necessary to assess whether pollinators are important in maintaining the spineflower population. Determining the specific pollinator(s) is important in identifying the range and type of habitat(s) required for maintaining an effective pollinator population(s). Based on his work with a related taxon (*Eriogonum*) and on the spineflowers' floral morphology, Dr. James Reveal (pers. comm.) suggests that the San Fernando Valley spineflower may be capable of both cross-pollination and self-pollination. Outcrossing would likely be the primary means of reproduction, because of (1) the presumed differential timing between pollen release and stigma receptivity and (2) the spatial separation between anthers and stigma. Late in the pollination cycle, however, the still-receptive stigma may roll back and pick up any remaining pollen, thereby resulting in self-pollination. Although self-pollination may result in production of viable seed and ensure short-term persistence, it may also lead to reduced genetic diversity over time (Reveal pers. comm.). Dr. Eugene Jones (pers. comm.) is in the process of determining some of these reproductive characteristics for the San Fernando Valley spineflower (e.g., whether the flowers are protandrous versus protogynous, whether self-pollination is autogamous versus geitonogamous, etc.).[2] Dr. Jones notes that related taxa having similar floral structures may function differently from one another.

Reveal (pers. comm.) has observed other spineflower species being effectively pollinated by ants, but indicates that, in those cases, ants are incidental (secondary) rather than primary pollinators. Jones (pers. comm.) observed high densities of ants in and out of spineflower corollas in the field, and suggests that ants may play an important role in pollination of this species. Hickman (1974) demonstrated ant pollination as a specialized mutualistic system in another annual species within the buckwheat family, *Polygonum cascadense*. *Polygonum cascadense* shares several similarities with the spineflower, including habit (e.g., low, erect annual), habitat (e.g., open, dry slopes), and possibly, reproductive characteristics (e.g., stamens maturing before the stigma).

The spineflower involucre (whorl of modified leaves adjoining each flower) is characterized by straight spines, which may be an adaptation for animal dispersal of seeds and may help anchor seeds to suitable substrate (GLA 1999). Seeds apparently remain in the involucre even after the plant disarticulates. Small mammals or even ants may play a role in seed dispersal; however, studies have not yet been conducted to determine whether any animals onsite effectively disperse spineflower seed. Reveal (pers. comm.) notes that gallinaceous birds that peck and scratch at the soil surface can be effective in planting seeds of chorizanthoid species as non-incidental dispersal agents, and that localized dispersal may also be accomplished by small mammals. The one season of data indicates relatively high seed production for the spineflower. It is not known whether seed predation by animals significantly affects the seed bank.

[2] Protandry refers to the condition in which flowers shed their pollen before the stigma becomes receptive. Protogyny refers to the opposite condition, i.e., the stigma matures and becomes receptive before the anthers dehisce and shed pollen. Autogamy refers to self-pollination that occurs when a flower is pollinated by its own pollen, whereas geitonogamy is the condition in which a flower is pollinated by pollen from another flower on the same plant. The latter is, in effect, self-pollination because the results are genetically identical to pollination by autogamy (Proctor et al. 1996).

Fire has been suggested as a possible management tool for maintaining or enhancing spineflower habitat. The effects of fire on germination of the San Fernando Valley spineflower have not yet been established. Studies on a closely related taxon (*Chorizanthe parryi* var. *parryi*) that occurs in similar habitat indicate that fire has at least a short-term inhibitory effect on seed germination (Ellstrand 1994; Ogden 1999).

BACKGROUND ON EDGE EFFECTS

In the context of conservation biology and preserve design, edge effects are defined as adverse changes to natural communities as a result of their proximity to human-modified areas (Lovejoy et al. 1986; Yahner 1988; Sauvajot and Buechner 1993) or, more simply, the adverse effects of development on adjacent biological resources. Examples of edge effects include increases in invasive, weedy species, increased trampling and soil compaction from human recreation, or increases in nonnative animal species. Edge effects have been documented within specified distances of developed lands, although the impacts may be species- or resource-specific and tempered by a host of site-specific factors, including microtopography (McEvoy and Cox 1987; Andersen 1991), distribution and size of gaps (Bergelson et al. 1993), and intactness of the natural community (Sauvajot and Buechner 1993). A number of empirical studies have concluded that detrimental effects to biological resources can occur at distances ranging from 150 to 600 feet from the edge of the urban-wildland interface (e.g., Gates and Gysel 1978; Brittingham and Temple 1983; Andren et al. 1985; Wilcove 1985; Angelstam 1986; Wilcove et al. 1986; Temple 1987; Andren and Angelstam 1988; Santos and Telleria 1992; Alberts et al. 1993; Scott 1993; Vissman 1993). The majority of these studies focus on impacts to wildlife habitat. Few studies that we reviewed focus specifically on edge effects to plant species.

Buffer Considerations

Kelly and Rotenberry (1993) provide guidelines for effective buffers around urban reserves that are useful in recommending buffer widths and assessing potential edge effects on the San Fernando Valley spineflower resulting from the proposed preserve design. Kelly and Rotenberry (1993) note that the effective size of an ecological preserve is almost always smaller than the area within the preserve boundary, or the total preserve size. The effective size is generally referred to as the *core area*. The preserve boundary or *edge* surrounds the core area. The width of the edge is a function of the permeability of the boundary to negative external influences or risk factors. Edge effects can be particularly significant for small reserves because of their relatively large perimeter to core ratios (Soulé et al. 1988; Bolger et al. 1991; Saunders et al. 1991). An effective buffer width can be determined on a site-specific basis by (1) identifying risk factors and potential impacts to the species of concern within the preserve and (2) determining the permeability of the urban-wildland boundary to vectors of those risk factors. Altering the boundary permeability through habitat management is a potential method for mitigating identified impacts (Kelly and Rotenberry 1993). However, this method may not be effective for all types of risk factors (e.g., wind-blown seed of invasive plant species). Incorporating appropriate site design measures and land use restrictions into the development abutting the preserve is an alternative method of avoiding and minimizing impacts to the preserve (i.e., designating a land use buffer outside the preserve).

RISK FACTORS AND POTENTIAL IMPACTS

Preliminary studies on the biology and ecology of the San Fernando Valley spineflower (GLA 1999) indicate that the following parameters may play a role in the persistence of this taxon on the Ahmanson Ranch and may be negatively influenced at the urban-wildland interface:

- gaps in vegetation cover (i.e., areas of bare soil)
- low nutrient soils
- pollinators
- seed dispersal agents
- extant seed bank

Risk factors at the urban-wildland boundary that may affect these parameters include the following:

- nonnative, invasive plant and animal species
- vegetation clearing for fuel management or creation of trails
- trampling
- increased water supply due to suburban irrigation and runoff
- chemicals (e.g., herbicides, pesticides, fertilizers)
- increased fire frequency

Some of these risk factors could affect more than one of the parameters. Potential effects of these risk factors on the spineflower population are discussed below.

Invasive Plant Species

San Fernando Valley spineflower appears to prefer open patches of bare ground, which are often invaded by exotic plant species, as well (Amor and Stevens 1976; Forcella and Harvey 1983; Bazzaz 1986; Alberts et al. 1993). Although the spineflower on the Ahmanson Ranch was noted on thin, compacted soils lacking nonnative grasses, it is not clear whether spineflower density is significantly lower on these soils versus on deeper soils or whether nonnative grasses may be more abundant in these areas in years with average or above-average rainfall. Brooks (1995) noted that with increased rainfall, annual grasses gradually gain dominance once they have colonized an area, regardless of management or other protective measures. Gordon-Reedy (pers. obs.) has also observed large fluctuations in nonnative grass density in open coastal sage scrub in Riverside County in years with variable rainfall amounts.

Direct competition between native and exotic plant species is well documented (Alberts et al. 1993). Furthermore, the successful invasion of exotic species may alter habitats and lead to displacement or extinction of native species over time. For example, exotic invasions have been shown to alter hydrological and biochemical cycles and disrupt natural fire regimes (MacDonald et al. 1988; Usher 1988; Vitousek 1990; D'Antonio and Vitousek 1992; Alberts et al. 1993). Vitousek and Walker (1989) noted that aggressive nonnative species might displace native species by altering soil fertility.

MacDonald et al. (1988) reported that reserves surrounded by development areas supporting populations of exotic species are most subject to invasion. However, in studies on the effects of urban encroachment into natural areas in the Santa Monica Mountains, Sauvajot and Buechner (1993) found that direct habitat alteration or disturbance within natural areas is a more significant factor in the extension of edge effects into those areas than proximity to urban development alone. Several other studies have also correlated invasions by alien plants into nature reserves with elevated levels of disturbance, high light conditions, and, in some cases, increased water availability (McConnaughay and Bazzaz 1987; Laurance 1991; Tyser and Worley 1992; Brothers and Spingarn 1992; Matlack 1993).

In a review of biological invasions of 24 nature reserves, Usher (1988) reported a positive correlation between the number of human visitors and the number of introduced species. Further, he cited circumstantial evidence that invasive plant species are most common near paths through the reserves. Tyser and Worley (1992) provided data indicating that alien plant species extend up to about 325 feet into natural habitat from primary roads, secondary roads, and backcountry trails. They found a gradual decline in species richness with distance from the edge, and effects along trails were less prominent (but still evident) than along roads. Ghersa and Roush (1993) noted that the number of propagules available rarely limits the abundance of weeds in a given setting; rather, one needs to consider both the dispersal strategies of the invading species and potential vehicles for dispersal. Well-known dispersal agents include humans (Usher 1988; Ghersa and Roush 1993), vehicles, and road construction (Amor and Stevens 1976; Amor and Piggin 1977; Lonsdale and Lane 1991 in Hobbs and Humphries 1995; Hobbs and Humphries 1995). In addition to promoting biological invasions by acting as dispersal vectors, humans can impact spineflower habitat by disturbing the soil surface, trampling individual plants, and increasing the fire frequency within or adjacent to reserves.

Factors that affect the success of invasions include dispersal ability of the invasive species, in conjunction with size and distribution of gaps in the vegetation (McConnaughay and Bazzaz 1987; Bergelson et al. 1993) and the timing of seed dispersal relative to environmental conditions or "invasion windows'" (Johnstone 1986). Bergelson et al. (1993) documented an *average* dispersal distance for the ruderal, wind-dispersed annual plant, *Senecio vulgaris,* of 1.1 feet; however, they also noted dispersal events for this same species of over 50 feet. McEvoy and Cox (1987) reported that 89% of seeds of another wind-dispersed species (*Senecio jacobaea*) traveled 16 feet or less, while no seeds were observed >45 feet from the source in a mark-recapture study. They noted, however, that secondary dispersal and animal dispersal may increase initial dispersal distances under some conditions. For example, in dry, open habitats, seeds may be moved along the ground or swept into the air by wind (McEvoy and Cox 1987).

Laurance (1991), in a study of edge effects in tropical forest fragments, found a striking abundance of invasive plants within 650 feet of forest edges, and lower (but still elevated) levels of invasive plants 1,640 feet from the edges. Tyser and Worley (1992), in a study in the intermountain region of western North America, found invasive plants extending over 325 feet from road and trail edges, although there was a gradual decline in invasive species richness beyond about 80 feet. Amor and Stevens (1976) also found a general decline in invasive plants with increasing distances from roads into sclerophyll forests in Australia. They reported that at 100 feet from a road edge, the majority of invasive species either dropped out altogether or occurred in lower percentages than at the road shoulder, particularly in drier plant communities. In the presence of artificial sources of water, however, the occurrence of some invasive species remained high regardless of distance from the edge (Amor and Stevens 1976). Where there is a large perimeter between the preserve and urban interface, larger numbers of colonizing propagules can be expected to enter the preserve (Alberts et al. 1993). In general, Alberts et al. (1993) found that ruderals tend to invade reserves quickly, given appropriate site conditions, whereas ornamental species invade reserves over a longer period of time, and their presence is correlated with increased sources of water.

Invasive Animal Species

The effect of nonnative animal species on biological resources within reserves has been well documented (e.g., Gates and Gysel 1978; Brittingham and Temple 1983; Wilcove 1985; Andren and Angelstam 1988; Langen et al. 1991; Donovan et al. 1997); however, most of this literature pertains to effects on wildlife species. For example, both domestic dogs and cats are known to adversely impact native wildlife, with effects ranging from harassment to disturbance of breeding activities to predation (Kelly and Rotenberry 1993; Spencer and Goldsmith 1994). Domestic dogs have been observed within reserves at a distance of greater than 325 feet from the edge, while cats have been observed within reserves more than 1 mile from human dwellings in Riverside County (Kelly and Rotenberry 1993). An increase in nonnative predators as a result of development adjacent to the spineflower preserve could potentially affect populations of rodents (e.g., kangaroo rats, pocket mice, pocket gophers) that may act as seed dispersal agents or play a role in bioturbation.[3] In a study of two populations of house cats on a suburban-desert interface near Tucson, Arizona, Spencer and Goldsmith (1994) found that most prey were diurnal species of rodents, birds, and reptiles. Radio-tracking studies indicated that the cats spent over 90% of their time within 100 feet of houses, although this may have been related to an abundant coyote population. Spencer and Goldsmith (1994) suggested that impacts of cats on native wildlife are concentrated within 100-200 feet of the urban-wildland interface in the presence of predators (e.g., coyotes), but may extend further in their absence.

If rodents consume spineflower seeds, then a reduction in the rodent population may reduce seed dispersal into sites suitable for germination. Perry and Gonzalez-Andujar (1993) developed a model to assess the role of seed dispersal on metapopulation growth

[3] Bioturbation is the aeration and mixing of soil by organisms.

and persistence of an annual plant, like the spineflower, that forms a seed bank and occurs in drought-like and disturbed environmental conditions. This model predicts that a strongly dispersing metapopulation is hardly affected by temporal environmental heterogeneity, while metapopulations with moderate or no dispersal capabilities suffered extinction in every replication. However, granivorous rodents tend to selectively harvest large seeds (Brown and Lieberman 1973; Brown et al. 1979; Samson et al. 1992; Brown and Harney 1993). Spineflower seeds are relatively small (ca. 2 mm), and may only be used by smaller rodents (e.g., pocket mice) that clip clusters of involucres. Even if rodents do not play a significant role in spineflower seed dispersal through seed predation, they may still effect some localized dispersal when the awn-tipped involucres (and seeds) become temporarily attached to their bodies. In addition, rodents may indirectly benefit the spineflower by suppressing populations of larger-seeded annual plants that compete with the spineflower (Davidson et al. 1984; Samson et al. 1992; Brown and Harney 1993).

Decreases in the rodent population may also reduce the amount of potentially high quality habitat for spineflower establishment. Rodent activities that result in bioturbation and bare soil patches have been associated with spineflower plants on Ahmanson Ranch (GLA 1999). Long-term studies in the Southwest have demonstrated that selective removal of kangaroo rats, for example, resulted in much less disruption of the soil surface, higher densities of tall perennial and annual grasses, increased accumulation of litter, decreased foraging by granivorous birds, and differential colonization by rodents typical of grassland habitats (Brown and Heske 1990; Thompson et al. 1991; Brown and Harney 1993).

Conversely, Mills (1996) demonstrated that edges could have higher populations of certain mammalian seed predators (e.g., deer mice [*Peromyscus* spp.]) than core areas, which may result in reduced plant recruitment. Deer mice are good edge specialists, and can reach high densities under appropriate conditions. Because they are generalists that can switch among food resources, they often exert a heavier toll on a certain food resource (like seeds) than specialists whose populations track the specific resource more closely. Jules and Rathcke (1999) found reduced recruitment of a native herbaceous perennial plant species (*Trillium ovatum*) within about 200 feet of a forest/clearcut edge, and demonstrated that this was significantly correlated, in part, with seed predation by rodents (species unspecified). To date, no studies have been conducted that define the role of rodent populations (if any) in spineflower seed dispersal or predation. In light of these uncertainties, it therefore seems important to maintain as natural a mix of native seed dispersers/predators as possible, and to minimize ecological imbalances due to abundant nonnative species.

One invasive species that has been documented on the Ahmanson Ranch and may potentially increase in dominance over time is the Argentine ant. Ant surveys indicated that the Argentine ant is abundant in some areas of the project site, but currently occurs in very low numbers in or near spineflower habitat, presumably due to xeric conditions (Hovore pers. comm.).

Disturbed habitats are often considered vulnerable to Argentine ant invasions. There is evidence that this exotic species rapidly invades disturbed areas within stands of native habitat (Erickson 1971; Ducote 1977 in Suarez et al. 1998; Ward 1987; DeKock and Giliomee 1989; Knight and Rust 1990; Suarez et al. 1998). Suarez et al. (1998) found Argentine ants most abundant along the edge of urban preserve areas, with densities of ants in the preserve decreasing with distance from the edge. They found that ant activity was highest within about 325 feet of the nearest urban edge, whereas areas sampled beyond 650 feet contained few or no Argentine ants. However, Argentine ants have also been found at distances of approximately 1,300 feet and 3,280 feet from the edge, respectively, in other urban reserves in southern California (Suarez et al. 1998). DeKock and Giliomee (1989) documented extensive penetration of this species into natural areas in South Africa along roads. Recent studies indicate that the Argentine ant may be capable of invading undisturbed habitat, as well (Cole et al. 1992; Human and Gordon 1996).

Argentine ants appear to be confined to low elevation areas with permanent soil moisture (Erickson 1971; Tremper 1976 in Suarez et al. 1998; Ward 1987; Knight and Rust 1990; Holway 1995, 1998). Tremper (1976) reported that Argentine ants desiccate more easily and are less tolerant of high temperatures than native ants. Suarez et al. (1998) indicated that the presence of the Argentine ants in urban reserves might be dependent on water runoff from developed areas. Holway (1998) found that the rate of Argentine ant invasion is primarily dependent on abiotic conditions (e.g., soil moisture), rather than on disturbance. He suggested that disturbed areas are often a point of introduction, but encourage invasions only if they increase the availability of a limiting resource such as water. Blachly and Forschler (1996) found Argentine ants thriving in areas disturbed by human activity, but indicated that their presence is also related to added ground cover, permanent water supplies, and a simplified native ant fauna.

Although the reproductive strategy of the San Fernando Valley spineflower is not yet known, field studies indicate that flowers are visited by a number of invertebrate species. Presumably, one or more of these species function as effective pollinators of the spineflower. Invasive faunal species (e.g., Argentine ants, parasites) have the potential to negatively impact pollinator populations. Loss or limitation of pollinators may adversely affect the long-term survivability of the spineflower by reducing seed output (e.g., reproductive failure) if there is no selfing (Jennersten 1988; Bawa 1990) or decreasing the effective population size through reduced gene flow (Bawa 1990; Menges 1991; Aizen and Feinsinger 1994). Some studies have shown that pollinator limitation can reduce seed output by 50-60% (Jennersten 1988; Pavlik et al. 1993; Bond 1995). Jules and Rathcke (1999) demonstrated that pollinator limitation was significantly related to reduced recruitment of a native plant species within 200 feet of a forest/clearcut edge.

It has been hypothesized that native ants may be a primary or secondary pollinator of the San Fernando Valley spineflower (GLA 1999). The Argentine ant is known to displace native ant species (Erickson 1971; Tremper 1976 in Suarez et al. 1998; Ward 1987; Holway 1995; Human and Gordon 1996; Suarez et al. 1998), although this apparently has not yet occurred in spineflower habitat on the Ahmanson Ranch. Nonetheless, potential negative interactions between native ant species or other insect pollinators and the Argentine ant would be a concern if the spineflower were insect-pollinated.

Ant pollination is considered relatively uncommon in plants (Proctor et al. 1996), although Jones (pers. comm.) indicates that ants may be a major pollinator of cushion plants in desert areas and Hickman (1974) has demonstrated effective ant pollination in a taxon related to the spineflower. Ant-pollinated plants tend to occur in hot, dry habitats and are further characterized by a prostrate or low-growing habit, small, inconspicuous flowers close to the stem, intertwining plants within a population, few seeds per flower, and small pollen volume and nectar quantity (Hickman 1974). The San Fernando Valley spineflower possesses many of these characteristics. In a study conducted in the South African fynbos,[4] Paton (1986 in Visser et al. 1996) correlated high densities of ants (species undetermined) in inflorescences of *Protea eximia* with lower numbers of other insects. Visser et al. (1996) investigated whether Argentine ants influenced the number of insect species and individuals present in the inflorescences of *Protea nitida*, and found that 10 of 11 insect taxa showed reduced numbers where Argentine ants were present and, in 5 cases, these reductions were highly significant. In addition, the total number of insects was significantly suppressed in inflorescences with high numbers of Argentine ants. Visser et al. (1996) speculated that a reduction in the diversity and abundance of insect visitors could result in reduced pollination and ultimately affect the reproductive capacity of the plant. In the species they studied, ants were not considered effective pollinators, and an increase in ant abundance was not expected to promote pollination.

Ants may also function as primary or secondary dispersers of seeds (Roberts and Heithaus 1986; Louda 1989). They have been reported to contribute to the spatial heterogeneity of seed distribution (Reichman 1984, 1979) and they decrease seed abundance of some numerically dominant ruderal species in relation to less dominant native annual species (Inouye et al. 1980). Displacement of native ant species by the Argentine ant could negatively affect spineflower persistence by reducing spineflower seed number and distribution. Bond and Slingsby (1984) investigated the effects of displacement of native ant species by the Argentine ant on a myrmecochorous plant[5] in South Africa, and found that the Argentine ant negatively affected seed dispersal and plant regeneration. Native ant species typically carry seeds to their nests, where they remain or are later discarded in nearby middens. While the ants derive nutritional benefits from the seeds, this process also increases seedling recruitment by minimizing competition near the parental plant, reducing seed predation at the soil surface, and enhancing plant growth in the nutrient-enriched soils of the nests or middens (Marshall et al. 1979; Heithaus et al. 1980; O'Dowd and Hay 1980; Bond and Slingsby 1984). In contrast, Argentine ants are slower to discover seeds, move them a shorter distance, and fail to store them in below-ground nests, thus resulting in decreased dispersal and increased seed predation (Bond and Slingsby 1984; Holway 1999). Bond and Slingsby (1984) reported significant decreases in seed germination and establishment in areas infested with Argentine ants compared with uninfested areas, and ascribed these differences primarily to increased seed

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[4] Fynbos is a chaparral-like vegetation community found in mediterranean climate regions of South Africa and Australia. It is dominated by evergreen shrubs with sclerophyllous (hard) leaves (Dallman 1998).

[5] A myrmecochorous plant is dependent on ants for seed dispersal.

predation. They further suggested that the negative effects of Argentine ants on myrmecochorous species with a persistent seed bank will only become apparent over relatively long time periods (e.g., decades) as the seed bank becomes depleted.

DeKock (1990) found that the first native ant species to be driven off by Argentine ants are those that are most effective in seed dispersal. She suggested that the effects of Argentine ant invasions on native plants would be indirect and related to a depleted seed bank. It should be noted that many ant-dispersed seeds have structural adaptations such as oily seed coats or fat-bearing appendages (elaiosomes) that provide nutritional rewards for the dispersing ants (Stebbins 1974; Marshall et al. 1979). Hughes and Westoby (1992) demonstrated that seed dispersal by ants was, in general, significantly higher for seeds with elaiosomes, although this effect was ant species-specific, and some dispersal did occur in the absence of these structures. It is not known whether spineflower seeds have any adaptations that would predispose them to ant-dispersal.

Vegetation Clearing

Disturbance of native vegetation communities can produce appropriate site conditions for germination of weedy species (Bazzaz 1986; Westman 1990; Alberts et al. 1993; Hobbs and Humphries 1995). In general, ruderal weedy species possess a number of characteristics that allow them to rapidly colonize gaps or bare areas. These include the production of abundant, typically wind-dispersed seeds that are quick to germinate, establish, and grow (Frenkel 1970; Amor and Piggin 1977; Bazzaz 1986). Thus, weedy exotics often out-compete native species that utilize similar habitats. Clearing of vegetation along the urban-wildland interface (e.g., firebreaks, roads) or within a preserve system (roads, trails) may provide opportunities for such weedy species to gain a foothold in the preserve (Amor and Stevens 1976; Amor and Piggin 1977; Lonsdale and Lane 1991 in Hobbs and Humphries 1995).

Trampling

Trampling can affect the spineflower either by damaging individual plants or altering the ecosystem. Maschinski et al. (1997) demonstrated that the combination of trampling and poor climatic conditions resulted in an accelerated extinction probability for a native plant species. In this case, trampling directly affected plant fitness, resulting in significantly lower fruit production. Trampling can also create gaps in vegetation that provide opportunities for exotic plant establishment (Hobbs and Huenneke 1992). Cole (1987) reported that even low levels of trampling caused a substantial loss of vegetation cover and species diversity, and resulted in an increase in soil compaction, whereas soil erosion occurred with higher levels of trampling. In other studies (see Dale and Weaver 1974; Bright 1986), species diversity increased in areas subject to trampling, but species composition shifted to those plants that are resistant to trampling. In general, plants with tough, wiry leaves or thick leaves and a tufted growth form (e.g., grasses) are more resistant to trampling than herbaceous plants, such as the spineflower, whose branches or stems could be easily crushed or broken (Cole 1987; Hall and Kuss 1989). Refer to the literature cited above (plant invasions, vegetation clearing) for discussions on invasion of gaps or vegetation disturbances by weedy versus native species.

Harrison (1981) found that the season or timing of trampling influences the effects on native species and their recovery. The ability to recover from trampling is also dependent on environmental conditions (temperature, moisture) and growth form characteristics (Cole 1987). Some adverse effects of trampling (soil compaction, erosion) are less easily reversed than others. For these factors, recovery may be difficult after only a few years of trampling at relatively high intensities (Cole 1987).

Increased Water Supply

Changes in surface and subsurface hydrological conditions at or near the urban-wildland boundary could occur as a result of removal of native vegetation, increased runoff from roads or other paved surfaces, and residential or commercial irrigation. Increased surface water flows may result in increased erosion and transport of particulate matter (Saunders et al. 1991). Altered patterns of erosion may deposit new substrates for plant colonization, although such areas are often quickly colonized by weedy species that require both disturbance and nutrient-rich substrates for establishment (Hobbs and Atkins 1988). Increased surface flows may also be a conduit for introducing invasive species into the preserve. Holway (1998) indicated that Argentine ant colonies are often dispersed into new areas by jump-dispersal events such as floods, and that these types of dispersal events are an important component of the large-scale dynamics of Argentine ant invasions.

Increased surface moisture or underground seepage that results in increased soil moisture levels may also promote the establishment of exotic plant species (Alberts et al. 1993; McIntyre and Lavorel 1994; Amor and Stevens 1976) or wetland-dependent native plant species, facilitate invasion by Argentine ants (Suarez et al. 1998), alter seed bank characteristics, and modify habitat for ground-dwelling fauna (Saunders et al. 1991). Seepage is expected to be minimal in most areas along the urban-wildland interface due to the underlying substrate. However, the current project design includes a few hundred feet of man-made slopes between two stands of the spineflower, and there is the potential for some seepage on these fill soils (Barker pers. comm.).

Chemicals (Herbicides, Insecticides, Fertilizers)

Chemical pollutants can adversely affect biological resource areas in many ways, including decreases in pollinators, increases in weedy exotic species, or damage to or direct killing of native plants. The use of herbicides to maintain open areas within or adjacent to the preserve can result in chemical habitat fragmentation and consequent reductions in pollinator populations (Buchmann and Nabhan 1996). Insecticide spraying in adjacent residential areas can result in pollution drift that kills pollinators in reserve areas (Kelly and Rotenberry 1993; Allen-Wardell et al. 1998). Boutin and Jobin (1998) reported that chemical pesticide drift using ground equipment has been estimated at 1-10% of the application rate within about 30 feet of the target. In a study on the effects of various herbicides on native plant species in a nature reserve, Marrs et al. (1989) demonstrated that the maximum safe distance (i.e., no lethal effects) was about 20 feet from the spray source, although the average safe distance was 6.5 feet or less. They also found that adverse but non-lethal effects of spraying (e.g., plant damage, flower suppression) occurred at slightly greater distances than lethal effects, and showed seasonal variability. For example, no damage was detected beyond about 8 feet for most of the species they tested in fall. A few species, however, appeared to be particularly sensitive to herbicides during this time period, and showed damage between 33 and 65 feet from the spray source. In spring, the maximum distance at which damage effects were apparent was about 25 feet from the spray source. However, most damaged plants recovered completely by the end of the growing season. Based on these results, Marrs et al. (1989) advocated the use of a 16 to 33-foot buffer zone to minimize lethal effects to herbaceous plants from herbicide drift, and noted that wider buffers (e.g., 50 feet) would reduce risks even further.

Other chemicals, such as are included in fertilizers, may enhance growth of weedy species and, thus, should not be used adjacent to the preserve. For example, nitrogen is a limiting factor in plant growth, and the addition of nitrogen fertilizers enhances the growth of many plant species. Many native plant species, however, are adapted to low-nitrogen systems (Vitousek et al. 1997; Zink and Allen 1998). Vitousek et al. (1997) stated that the addition of nitrogen to such systems, through direct fertilization or runoff from adjacent areas, could cause shifts in species dominance and reduce overall species diversity. Furthermore, nitrogen-rich systems may promote exotic weedy species to the detriment of native species (Zink and Allen 1998).

Aerial fallout of nitrogenous compounds from automobiles may also contribute to increased nitrogen in the soil. Allen (1996) has observed high mortality of coastal sage scrub shrubs in areas with high soil nitrogen levels, and hypothesizes that nitrogen deposition from air pollution may be responsible for this mortality (Allen et al. 1996). Vegetation and soils are known to be important sinks for other atmospheric pollutants from automobiles, as well, although a number of biological and environmental factors may affect the actual absorption or accumulation of such compounds. The level of pollutants in roadside plants has been positively correlated with traffic density. Singh et al. (1995) reported the most significant effects where traffic volume was high (e.g., >4,000 vehicles per 2 hours).

Increased Fire Frequency

The effects of fire on the San Fernando Valley spineflower are not yet known. Seed germination of a closely related taxon, Parry's spineflower (*Chorizanthe parryi* var. *parryi*) appears to be inhibited by fire in both greenhouse and natural settings (Ellstrand 1994; Ogden 1999). Despite the inhibitory effect of direct scorching, fire may also prove beneficial to the spineflower by creating openings and temporarily reducing competition.

San Fernando Valley spineflower occurs primarily in openings in coastal sage scrub, although much of its habitat on the Ahmanson Ranch appears to have been invaded by nonnative grasses. The coastal sage scrub community is adapted to fire, but not completely dependent on it for continued viability. In general, it is considered a relatively stable vegetation community over a broad range of fire frequencies, particularly if detrimental factors such as fragmentation and exotic weed species invasions are minimized. However, excessively long or short fire intervals may result in (1) shifts in the composition of the dominant species of this community (Westman 1987, 1981; Keeley 1991) or (2) displacement of native species by nonnative species, such as annual grasses. Nonnative grasses exert a number of undesirable effects on native plant communities, including altering fire regimes. Colonization of an area by nonnative grasses provides the fine fuel needed to start and maintain fires. This can lead to increased fire frequency, extent, and intensity. Nonnative grasses typically recover more quickly than native species following grass-fueled fires, thereby initiating a cycle of increasing fire susceptibility (D'Antonio and Vitousek 1992; Hobbs and Huenneke 1992). Changes in fire regimes due to invasive species can result in a wide range of ecosystem changes, including nutrient loss, altered local microclimate, and prevention of succession (D'Antonio and Vitousek 1992).

The use of fire has been suggested as one method for controlling nonnative grasses. Controlled burns have been used with some success to control nonnative grasses, particularly in grassland communities (Zavon 1982 in Pollack and Kan 1998; Ahmed 1983 in Pollack and Kan 1998; Keeley 1990; George et al. 1992; Pollack and Kan 1998). Pollack and Kan (1998) and others (see Menke 1992) found that late-spring fires were an effective method of controlling annual species that do not have well-developed seed banks, or of reducing the size of the seed bank in those alien species that do form a seed bank. Pollack and Kan (1998) suggested that knowledge of the target species' phenology is critical in effective timing of burns. In their study, late-spring burns were associated with more intense fire behavior and the need for fire suppression equipment (Pollack and Kan 1998). Controlled or prescribed burns are often suggested as a management tool to improve habitat characteristics, and a recent report of the Wildland/Urban Interface Task Force (1994) included a wildland fire management-planning model designed to facilitate prescribed burning and post-fire management. However, recent attempts to incorporate burns (or even "let-burn" policies) into habitat management plans in southern California have met with resistance from local fire control agencies, particularly near urban areas.

In addition to the fire-inducing effects of nonnative grasses, fire frequency near urban-wildland boundaries may increase due to other human-related activities (e.g., construction or utility maintenance activities, children playing with matches).

ANALYSIS OF RISK FACTORS

The objectives of this analysis are to (1) determine how risk factors can be reduced through buffers and management actions; (2) provide a relative ranking of risk factors that pose the greatest threat to spineflower persistence, based on boundary permeability; and (3) recommend buffer/management scenarios that effectively address risk factors. This analysis utilizes a step-wise approach by first considering buffer widths alone as a means of reducing risk factors, then overlaying buffers with proposed management actions[6] to reduce potential negative effects from risk factors. Ranking of risk factors is based on the literature review, field observations, and professional judgment. Risk factors that can be least controlled by management are considered to present the highest risk to spineflower persistence.

Buffer Widths

Buffers are an important component of preserve design. Here, the buffer is defined as the distance between the edge of the current spineflower population within the preserve and the edge of the preserve. Various buffer widths were assessed to determine their effectiveness in minimizing identified risk factors. The five buffer widths included in this analysis range from a minimum width (15 feet) to greater widths shown to be effective in the edge effect literature for specific risk factors. Table 1 presents the relative assessment of varying buffer widths in minimizing risk factors.

Table 1
ESTIMATED BUFFER EFFECTIVENESS FOR MINIMIZING EDGE EFFECTS OF SELECTED RISK FACTORS ON THE SPINEFLOWER

RISK FACTORS	BUFFER WIDTHS (FEET) [1]				
	15	30-50	80-100	200	300
Invasive Animals	L	L	L	M	M
Increased Fire Frequency	L	L	L	M	M
Invasive Plants	L	L	M	H	H
Vegetation Clearing	L	L	M	H	H
Increased Water Supply	L	L	M	H	H
Trampling	L	L	M	H	H
Chemicals	L	M	H	H	H

[1] Estimated effectiveness rankings: Low (L) = Unlikely to be effective; Moderate (M) = moderately effective; High (H) = highly likely to be effective.

Table 1 indicates that ranking of risk factors (i.e., from highest risk to the spineflower to lowest risk), based on buffer widths, can be grouped as follows:

- **Invasive Animals and Increased Fire Frequency** -- Literature on invasive animals indicates that most impacts that could affect the spineflower are concentrated within about 100-325 feet of the edge. Nonetheless, both cats and dogs have the ability to disperse much further into preserve areas. Argentine ants also have the ability to disperse further into preserve areas, but apparently only in

[6] For the purpose of this analysis, other preserve design elements, land use restrictions, and engineering designs are included under management actions.

the presence of adequate water supplies. Buffer width alone is not expected to be highly effective in reducing fire frequency.

- **Invasive Plants, Vegetation Clearing, Increased Water Supply, and Trampling** --Invasive plant species and vegetation clearing are closely related risk factors. Literature reviewed on invasive plants in temperate systems indicates that they may extend up to 325 feet into preserve areas, with a gradual decline in invasive species beyond about 80-100 feet. Further, the effectiveness of invasions is related to suitable substrates (e.g., gaps or disturbances, which may be created by vegetation clearing) and dispersal ability of the invasive species, among other factors.

  Surface runoff on the project site will be controlled through engineering designs. There is the potential for underground seepage, however, which may have a zone of influence that extends up to about 200 feet, depending on the substrate. The effects of trampling are primarily direct and limited to the area of impact, although associated trespass by humans can be an effective means of introducing nonnative species into the preserve.

- **Chemicals** -- Literature indicates that the majority of pesticide drift from chemicals will extend less than 35 feet from the source. Although the effects of fertilizers are typically localized, these compounds may be more widely dispersed through surface runoff or seepage. Atmospheric pollutants from cars can adversely affect plants, particularly where traffic density is very high; however, this may not be a factor in a residential development.

Management Actions

Management actions are expected to have varying degrees of effectiveness in reducing negative effects of identified spineflower risk factors. For example, the project proposes to control alterations in surface and subsurface hydrology through engineering designs. Restrictions on landscaping palettes, irrigation, and habitat disturbance adjacent to the preserve will reduce the potential for ornamental, invasive species in the preserve by limiting both the source material and appropriate site conditions for colonization. However, these restrictions do not address nonnative, weedy species that are already present in the area, and which have also been identified as major risk factors to spineflower persistence.

Table 2 overlays various management measures and buffer widths for each risk factor to assess their combined effectiveness in controlling edge effects. This analysis considers a wide range of management measures, not just those considered to be the most effective in controlling edge effects. These recommendations may not be comprehensive, and their effectiveness can only be roughly estimated at this time, based on the known biology of the species and conditions on the Ahmanson Ranch. Ranking of these measures also does not consider implementation or enforcement feasibility for each measure.

Table 2
ESTIMATED MANAGEMENT AND BUFFER EFFECTIVENESS FOR REDUCING EDGE EFFECTS

RISK FACTORS/MANAGEMENT MEASURES	BUFFER WIDTHS (FEET)[1]				
	15	**30-50**	**80-100**	**200**	**300**
Invasive Animals					
• No Specific Management Measures[2]	L	L	L	M	M
• Restrict landscaping palettes adjacent to the preserve to exclude use of invasive exotic species	L	L	M	H	H
• Restrict irrigation in and adjacent to the preserve	L	L	M	H	H
• Maintain current surface and subsurface hydrological conditions within the preserve through engineering design of adjacent areas	M	M	M	M	H
• Utilize french drains to minimize seepage on fill slopes, as determined necessary	H	H	H	H	H
• Inspect plants used in revegetation efforts in or adjacent to the preserve for pest species (e.g., Argentine ants)	L	L	M	H	H
• Avoid use of barriers (e.g., walls) with subsurface footings within or adjacent to the preserve	H	H	H	H	H
• Implement a bait control program for Argentine ants, as determined necessary through monitoring	L	L	M	M	M
• Bell cats in residential areas adjacent to the preserve and educate homeowners on the danger of coyotes to free-roaming cats	L	L	M	M	M
• Maintain habitat connectivity between preserve areas to encourage native predators in the preserve (thereby reducing populations of nonnative predators) and allow for recolonization of edge areas by native mammals	L	M	M	H	H
• Minimize internal fragmentation (e.g., roads, trails) and close unnecessary existing dirt roads	M	H	H	H	H
• Construct barriers to exclude nonnative animals (e.g., dogs)	M	M	M	M	M
Increased Fire Frequency					
• No Specific Management Measures[2]	L	L	L	M	M
• Implement a weed control program to reduce fine fuel capacity in fire-susceptible habitats	L	L	M	M	M

Table 2 (continued)
ESTIMATED MANAGEMENT AND BUFFER EFFECTIVENESS FOR REDUCING EDGE EFFECTS

RISK FACTORS/MANAGEMENT MEASURES	BUFFER WIDTHS (FEET) [1]				
	15	30-50	80-100	200	300
Increased Fire Frequency (continued)					
• Implement prescribed burning if shown to be advantageous to spineflower persistence and if allowed within the preserve by fire control agencies	M	M	H	H	H
• Restrict the use of construction or utility maintenance equipment in or adjacent to the preserve to avoid or minimize potential fires due to sparking (e.g., metal blades from bulldozers or other construction equipment striking rocks) or downed electrical lines	M	M	M	M	M
Invasive Plants					
• No Specific Management Measures[2]	L	L	M	H	H
• Restrict landscaping palettes adjacent to the preserve to exclude use on invasive exotic species	L	L	M	H	H
• Restrict irrigation adjacent to the preserve	L	L	M	H	H
• Maintain fuel breaks outside preserve boundary	L	L/M	M	H	H
• Minimize or prohibit vegetation clearing within the preserve (e.g., roads, trails)	H	H	H	H	H
• Restrict vegetation clearing immediately adjacent to the preserve	L	L	M	H	H
• Restore cleared areas with native species as soon as possible, subject to other conservation objectives	M	M	H	H	H
• Maintain current surface and subsurface hydrological conditions within the preserve through engineering design of adjacent developed areas	M	M	M	H	H
• Utilize french drains to minimize seepage on fill slopes, as determined necessary	H	H	H	H	H
• Control invasive weeds within the preserve and adjacent to the preserve (most appropriate method[s] to be determined)	L	L	M	H	H
• Reduce potential for invasion by weedy species by restoring selected disturbed areas within the preserve and adjacent to the urban boundary to reduce disturbance gaps	M	M	H	H	H

Table 2 (continued)
ESTIMATED MANAGEMENT AND BUFFER EFFECTIVENESS FOR REDUCING EDGE EFFECTS

RISK FACTORS/MANAGEMENT MEASURES	BUFFER WIDTHS (FEET) [1]				
	15	30-50	80-100	200	300
Invasive Plants (continued)					
• Reduce potential for invasion by weedy species by selecting sites for habitat enhancement or species reintroduction that minimize the potential for weed invasion	M	M	M	H	H
Vegetation Clearing					
• No Specific Management Measures[2]	L	L	M	H	H
• Site fire or fuel breaks outside preserve boundaries	L	L	M	H	H
• Minimize or prohibit vegetation clearing within the preserve (e.g., roads, trails)	H	H	H	H	H
• Restore cleared areas with native species as soon as possible, subject to other conservation objectives	M	M	H	H	H
Increased Water Supply					
• No Specific Management Measures[2]	L	L	M	H	H
• Maintain current surface and subsurface hydrological conditions within the preserve through engineering design of adjacent developed areas	M	M	M	H	H
• Utilize french drains to minimize seepage on fill slopes, as determined necessary	H	H	H	H	H
• Divert runoff from roads away from the preserve	M	M	M	H	H
• Restrict irrigation adjacent to the preserve	L	L	M	H	H
Trampling					
• No Specific Management Measures[2]	L	L	M	H	H
• Construct solid barriers to exclude or restrict pedestrian traffic	H	H	H	H	H
• Prohibit motorized vehicles, bicycles, and equestrian uses within the preserve	H	H	H	H	H
• Eliminate or reroute trails through the preserve to avoid sensitive biological resources	M	M	H	H	H
• Erect signs denoting boundary of the preserve and permitted uses	M	M	H	H	H
• Initiate an educational program (kiosks, information brochures, school programs, docent program)	M	M	H	H	H

Table 2 (continued)
ESTIMATED MANAGEMENT AND BUFFER EFFECTIVENESS FOR REDUCING EDGE EFFECTS

RISK FACTORS/MANAGEMENT MEASURES	BUFFER WIDTHS (FEET) [1]				
	15	30-50	80-100	200	300
Chemicals					
• No Specific Management Measures[2]	L	M	H	H	H
• Restrict use of herbicides within the preserve, and avoid use of pesticides within and adjacent to the preserve; herbicides must have no toxic effects on invertebrates	M	H	H	H	H
• Avoid use of herbicides and pesticides under conditions that would promote pollution drift (e.g., windy conditions)	L	M	H	H	H
• Avoid use of fertilizers within and adjacent to the preserve	M	M	H	H	H

1 Estimated effectiveness rankings: Low (L) = Unlikely to be effective; Moderate (M) = moderately effective; High (H) = highly likely to be effective.

2 Rankings indicate buffer effectiveness only (see Table 1), and are provided for comparison purposes.

Depending on buffer width and proposed land uses adjacent to the preserve, many of the recommended land use restrictions will require cooperation from homeowners. In addition, management measures in Table 2 are not weighted. It may be that some measures ranked as low are highly effective when combined with other measures. Conversely, some measures ranked high may be less important in minimizing risk factors than other measures with lower rankings (e.g., inspecting plants used in revegetation efforts versus restricting irrigation adjacent to the preserve). In some cases, there may be conflicts between various management measures. For example, a solid barrier would be highly effective in restricting human access and associated trampling effects. However, if the barrier includes subsurface footings, it may encourage nesting of Argentine ants. Some of the measures presented below may conflict with other objectives of spineflower protection, as well (e.g., habitat restoration). It is presumed that these measures will be refined during development of a detailed conservation strategy and management program for the spineflower. Finally, rankings in Table 2 consider individual effects only, and do not address the potential benefits of cumulative management measures. Combinations of certain management actions may have an enhanced capacity to address certain risk factors, as discussed in a later section of this document.

Table 2 indicates that individual management measures do, in fact, vary in their effectiveness for a specific risk factor. This makes it difficult to easily discern which buffer width would be expected to reduce a given risk factor to an adequate or acceptable level. Using a lowest common denominator approach (i.e., grouping risk factors according to the *least* effective management measure) results in the following ranking of risk factors, based on both management actions and buffer widths:

- **Invasive Animals and Increased Fire Frequency --** Based on this analysis, invasive animals and fire frequency are considered the highest risk factors to the spineflower because they require the largest buffer width (>300 feet) in order for *all* management measures to be highly effective. Management measures for both risk factors are considered moderately effective at 80-100 feet.

- **Invasive Plants, Vegetation Clearing, and Increased Water Supply --** Management measures for these three factors are all considered moderately effective at a buffer width of 80-100 feet and highly effective at widths of 200 feet or greater. Because control of these factors can presumably be achieved at narrower buffer widths than the factors above, they are given a lower ranking in terms of risk to the spineflower than either invasive animals or fire frequency.

- **Chemicals and Trampling --** All management measures for these risk factors are considered moderately effective at buffer widths of 30-50 feet and highly effective at buffer widths of 80 feet or greater. Therefore, these factors are given the lowest ranking in terms of risk to the spineflower, assuming management measures are implemented.

DISCUSSION

The analyses above assume that (1) risk factors are equivalent in their potential detrimental effects on spineflower persistence and (2) management measures are equally effective in ameliorating edge effects to the spineflower. Neither of these assumptions is likely to be valid, although the information needed to verify this is not available. Ranking of risk factors as a result of the combined effect of buffer width and management actions focused on individual management measures, and did not consider the interaction between different measures. For example, different levels of effectiveness may be achieved when management measures are combined. Even though some measures may be ranked low in effectiveness, they could increase in value when combined with other measures. For this reason, measures with low rankings are generally still considered important. Some management measures may not be as effective as others. They could override the positive effects of more effective measures or at least result in situations where management measures are effective for one component of a risk factor and less effective for others. Finally, it should be noted that there is no descriptive model for the spineflower or related taxa to demonstrate how this species may respond to either the risk factors or management measures. Risk factors are discussed below with respect to expected management effectiveness as a result of either management measure interactions or shortcomings.

1. *Invasive Animals*. Eleven management actions have been recommended to reduce edge effects due to invasive animal species. Invasive animals have a high potential to adversely affect the spineflower, although no such effects have yet been documented.

Of particular concern are (a) changes in soil moisture conditions that could alter habitat for rodents (potential seed dispersers) or encourage invasion of spineflower habitat by Argentine ants; (b) introduction of nonnative animal species (e.g., Argentine ants) on plant materials or along roads; and (c) habitat fragmentation that could lead to reduced levels of native predators (e.g., coyotes) and concomitant increases in nonnative predators (e.g., cats) that could affect rodent populations. Controlling irrigation and maintaining habitat connectivity between the spineflower preserve and other open space areas in order to encourage native predators in the preserve will be key issues in management effectiveness for this risk factor. Despite the potential seriousness of invasive animals on spineflower persistence, it appears that management measures are available to control the most detrimental aspects of animal invasions, given adequate buffer widths and appropriate preserve design.

2. *Increased Fire Frequency*. None of the buffer widths considered in this analysis would be effective in stopping the spread of fire into the preserve from adjacent areas, but three management measures have been recommended to reduce the frequency and intensity of fires within the preserve. At this time, the effect of fire on the spineflower is not known. It can be assumed, however, that frequent or intense fires would be detrimental to individual spineflowers and spineflower habitat. Changes in natural fire cycles are related, in part, to the presence of fine fuels (especially nonnative grasses) within the preserve. While complete removal of grasses within the preserve is highly unlikely, a weed control program can potentially reduce nonnative grass cover and inhibit the spread of grasses into currently unoccupied areas of the preserve. Despite weed control measures within the preserve, reinvasions may occur from sources outside the preserve, and the probability of such reinvasions increases with narrow buffer widths (<80 feet).

3. *Invasive Plants*. Eleven management actions have been recommended to reduce edge effects due to invasive plant species. While some of these measures were ranked as having low effectiveness at narrow buffer widths, they are still important in reducing overall invasiveness, particularly in combination with other measures. For example, restrictions on landscaping and irrigation adjacent to the preserve, in conjunction with revegetation of disturbed areas, are expected to reduce opportunities for invasion of nonnative ornamental plant species. The same combination of measures is not expected to be as effective in reducing either the invasion or increasing dominance of nonnative weedy species already present in the area. Field studies have indicated that competition with these weedy species may already play a major role in limiting spineflower distribution. Because of the uncertainty of controlling additional weed invasions into the preserve, invasive plants may pose the highest risk factor to the spineflower.

4. *Vegetation Clearing*. Three management actions have been recommended to reduce edge effects from this risk factor, and two of these are expected to be moderately to highly effective even at relatively narrow buffer widths. Vegetation clearing is of concern because it provides gaps that facilitate invasions by nonnative plant species. This risk factor is considered relatively high because of its relationship to invasive

plants and the uncertainty of controlling this factor outside the preserve. For example, vegetation clearing will occur adjacent to the preserve during the development process, and may be a long-term condition, depending on fuel break requirements. While weed control will likely occur within the preserve, there is a lesser chance of effective controls outside the preserve; thus, cleared areas outside the preserve may provide a constant source of propagules (seeds) for invasions into the preserve. At narrower buffer widths (<80 feet), the potential for dispersal of invasive species into the preserve is relatively high.

5. *Increased Water Supply*. This risk factor plays a key role in the success of nonnative plant and animal species invasions. Control of surface and soil moisture alone may be adequate to reduce invasions of nonnative ornamental plant species and the Argentine ant into the spineflower preserve. The ranking of this risk factor assumes that all recommended management measures (including irrigation restrictions) would be implemented.

6. *Chemicals*. As with vegetation clearing, the greatest uncertainty in controlling this risk factor is expected to be the use of chemicals adjacent to the preserve. Edge effects from chemicals do not appear to have as wide a zone of influence as other risk factors, as evidenced by a high level of management/buffer effectiveness at 80-100 feet, and at least moderate levels at 30-50 feet. The effects of chemicals on the spineflower are not known; however, they may affect both vegetation and pollinator populations. Any application of herbicides within the preserve (e.g., for weed control purposes) should be experimental in nature to determine the effects on both vegetation and pollinator populations. Placement of heavily traveled roads adjacent to the preserve should be evaluated relative to contribution to increased nitrogen levels in the soil or atmospheric pollutants that could be detrimental to native plant species or enhance growth of weedy species.

7. *Trampling*. Trampling has the potential to directly damage spineflower plants, resulting in lowered reproductive success. Other potential trampling effects include the loss of vegetation cover and species diversity, and an increase in soil compaction or erosion. Some of these potential effects (loss of vegetation cover, soil compaction) might appear beneficial to the spineflower. However, they may also promote invasion of spineflower habitat by trampling-resistant plant species that may outcompete the spineflower and further alter site conditions. There is a high potential for effective control of this risk factor, however, with all recommended management measures having a moderate or high effectiveness at a buffer width of 30-50 feet. This effectiveness ranking assumes a solid barrier to inhibit trespass into the preserve. The use of subsurface footings for such a barrier should be discouraged, however, since they may provide suitable nesting habitat for Argentine ants.

CONCLUSIONS

In designing and managing effective buffers for preserves, it is useful to consider both potential risk factors to biological resources from urban areas and the permeability of the urban-wildland boundary to those factors (Stamps et al. 1987; Kelly and Rotenberry 1993). The analysis and discussion above focused on (1) identifying potential risk factors and the ways they may negatively influence the spineflower population, (2) assessing the permeability of the boundary to those risk factors, and (3) identifying methods of changing or managing the boundary permeability to reduce potential impacts. In cases where boundary permeability cannot be managed effectively, an increased setback or buffer between sensitive biological resources and the development boundary, coupled with intensive management efforts and land use restrictions near the preserve, may be required to conserve the spineflower population.

Table 3 summarizes the overall effectiveness of management measures for each risk factor (based on the lowest common denominator) at each buffer width. Ranking of risk factors in Table 3 reflects the increased effectiveness in controlling risk factors when all management measures are combined for a given factor. For example, it appears that management measures, if implemented, may be more effective in controlling invasive animals than invasive plants.

Table 3
SUMMARY OF COMBINED BUFFER WIDTH AND MANAGEMENT EFFECTIVENESS[1] FOR REDUCING RISK FACTORS FOR THE SPINEFLOWER ON THE AHMANSON RANCH PROJECT

RISK FACTORS[2]	BUFFER WIDTHS (FEET)[3]				
	15	**30-50**	**80-100**	**200**	**300**
Invasive Plants	L	L	M	H	H
Vegetation Clearing	L	L	M	H	H
Increased Fire Frequency	L	L	M	M	M
Invasive Animals	L	L	M	M	M
Increased Water Supply	L	L	M	H	H
Chemicals	L	M	H	H	H
Trampling	M	M	H	H	H

1 Effectiveness rankings in Table 3 reflect the lowest common denominator for each risk factor, or the least effective management measure.
2 Risk factors are listed according to the level of threat they present to the spineflower (i.e., highest threat to lowest threat), assuming all management measures in Table 2 are implemented.
3 Estimated effectiveness rankings: Low (L) = Unlikely to be effective; Moderate (M) = moderately effective; High (H) = highly likely to be effective.

Based on this analysis, it is estimated that a buffer width of 15 feet, in combination with specific management measures, would be moderately effectively in controlling 1 risk factor (trampling) and unlikely to be effective in controlling the remaining 6 factors. A buffer width of 30-50 feet, in combination with management, would be moderately effective in controlling 2 risk factors (trampling and chemicals) and unlikely to control 5 factors. A buffer width of 80-100 feet, in combination with management measures, would be moderately effective in reducing the 5 greatest risk factors to the spineflower and highly effective in reducing the remaining risk factors. There appear to be no detectable differences in buffer effectiveness between 200 and 300 feet based on the literature reviewed. At both distances, management measures would be highly effective for 5 risk factors and moderately effective for the remaining 2 risk factors. Selection of an appropriate buffer/management package should focus on achieving an acceptable level of effectiveness in reducing the highest risk factors.

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