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Hedging our bets : bees and biodiversity on California's central coast

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**HEDGING OUR BETS: BEES AND BIODIVERSITY ON CALIFORNIA'S
CENTRAL COAST**

A Thesis

Presented to

The Faculty of the Department of Environmental Studies

San Jose State University

In Partial Fulfillment

of the Requirements for the Degree

Master of Science

by

Joanna Johnson

August 2007

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ABSTRACT

HEDGING OUR BETS: BEES AND BIODIVERSITY ON CALIFORNIA'S CENTRAL COAST

by Joanna Johnson

Agroecosystems dominate the U.S. landscape, and conversion of natural land to crop fields is often cited as a leading threat to biodiversity. Agricultural practices designed to enhance biodiversity can provide farms with ecological services such as pollination. Insect pollinators, including bees, are widely considered to be indicators of ecosystem health as a result of their co-evolved relationships with the plants they pollinate.

This study explored whether native plant hedgerows have a positive effect on pollinator presence. Bees were collected from six organically managed farms in Central California, four of which had hedgerows and two that did not. On hedgerow sites, bees were sampled at varying distances out from the hedgerows. Native bee abundance and diversity were markedly greater on hedgerow sites. Additionally, native bee presence and species richness were higher in samples collected within the hedgerow than in those collected at further distances from these native plantings.

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INTRODUCTION

In recent history, one of the main strategies for biodiversity conservation has involved purchasing tracts of undeveloped land and setting them aside as protected areas. At present, 17,000 nature reserves comprise nearly ten percent of the planet's terrestrial surface area (Perfecto 2003). This choice of strategy reveals some underlying assumptions; namely that any conversion of natural land to human purposes will inevitably cause significant habitat loss and that human efforts to gain subsistence from their environment pose a major threat to biodiversity conservation. But recent work has called those assumptions into question, pointing out that biodiversity has continued to dwindle at an alarming rate despite the establishment of protected areas (Perfecto 2003).

Fewer acres are set aside for conservation in the United States. Roughly two thirds of the American landscape is currently devoted to agricultural purposes and about five percent of U.S. land is designated as wilderness (Foreman 2004). In keeping with the global pattern, wildlife species have continued in a general trend of decline throughout the lower 48 states. When projections for growth in human population enter the equation, it becomes clear that development pressure and habitat fragmentation are not likely to abate, painting a picture of an increasingly segmented and marginalized terrain for wildlife.

Considering the model of modern industrial agriculture, with its emphasis on ecologically simplified systems that require significant levels of external inputs ranging from mechanized planting to pesticide use to genetic engineering (Altieri 1999), it is easy to follow the argument that this method of farming has served as a major contributor to

environmental problems such as water pollution and species decline through agricultural runoff and habitat fragmentation. But a variety of other models for crop cultivation exist and authors are beginning to suggest that agriculture might actually play a critical supporting role in the conservation of biodiversity (Perfecto 2003). Might agricultural practices be modified to allow for healthy ecosystems as well as adequate food productivity?

Since the majority of remaining open space in the U.S. is farmland, determining how those places are managed will play an important role in overall ecosystem health. There are at least two potential solutions to the current decline of species diversity and abundance: 1. continue to divide the landscape into distinct farmed and non-farmed areas, making farmed areas as high yield as possible and leaving the remaining land as unmodified as possible, and 2. raise the levels of biodiversity on farms as a means of increasing the array of species and habitat capacity throughout the wider area.

Island biogeography theory provides some framework to help evaluate the merits of these alternatives. In summary, the theory states that the combined processes of extinction and migration determine the number of species on an island. Rate of extinction is proportional to island size (larger islands have lower extinction rates) and migration rate correlates to level of isolation (the further an island is located from the mainland or other islands, the lower the migration rate) (MacArthur and Wilson 1967). Concepts from this Theory have been extended to ecosystem fragments, including nature reserves, viewing them as habitat islands and using the theory to explain the loss of species from the protected area (Perfecto 2003). Basically, extinctions can occur within

fragments if migration to and from other remnant habitats is not feasible for the species under consideration. This theory has given rise to an ongoing "SLOSS" (Single Large versus Several Small) debate among figures in conservation biology regarding how many protected areas are necessary and how large they ought to be (Higgs 1981; Simberloff and Able 1982).

A key assumption of island biogeography theory is that the areas surrounding the island (whether an actual island or a habitat fragment) are inhospitable. But if the agricultural and other managed lands lying between unmodified areas are hospitable enough to act as corridors, allowing wildlife to move between reserves, then protected patches might not function as isolated islands of habitat. Extinction of a population in one reserve might thus eventually be reversed by re-colonization from a population persisting in another habitat fragment. A concept that takes these between reserve zones into account is that of metapopulations. The term refers to the entire collection of subpopulations found within various habitat fragments (Doak and Mills 1994). The role of migration in preventing extinction has great importance, as does the habitat condition in these surrounding areas (whether it encourages or inhibits wildlife movement).

If the matrix can help support biodiversity, diversifying farmland and incorporating native vegetation may facilitate migration of wildlife through areas that would otherwise be blocked to them, and thus lessen the overall chances of extinction. But these practices do not merely aid the conservationist's agenda; they are also good for the farmer. Benefits of biodiversity include protection from invasive organisms and pest outbreaks, nutrient recycling, detoxification, and the prevention of erosion (Altieri 1999).

Ecological agriculture affords opportunity for a middle ground between the contentious lines often drawn between farmer and conservationist. Advocating integration of native vegetation into agroecosystems, explicit imitation of nature in farm design, and a view of farms that considers landscape level ecological processes has revealed new alternatives in an ongoing debate.

Citing a variety of recent studies, Altieri states that regulatory functions within agroecosystems are largely related to the site's level of biodiversity (1999).

Agroecosystems may well be capable of providing more than production-related ecosystem services, but a good deal of knowledge has yet to be gained. Increasing pressure from population growth and development will only increase the importance of implementing ecologically sound agricultural practices.

California's central coast is home to a growing number of farmers who are practicing alternative agriculture. In many cases, their operations occur at a smaller scale than conventional growers do and their methods are primarily organic. These farms also tend to grow a diverse array of crops. Research is being carried out at a number of these sites to further determine and quantify the benefits of diversified farm systems that incorporate features such as native plant hedgerows. In their regional context, hedgerows are defined as "lines or groups of trees, shrubs, perennial forbs, and grasses that are planted along roadways, fences, field edges or other non-cropped areas" (Earnshaw 2004). Hedgerows have generally been understood to serve as habitat for beneficial insects, pollinators and other wildlife, as a means of erosion protection, in weed control, as a buffer to pesticide drift, and as an aesthetic resource.

An element of biodiversity decline gaining heightened significance lately involves pollinators, especially bees. The issue is large in scope and significance. The United States Department of Agriculture has even referred to the last fifteen years as the worst "pollination crisis" in U.S. history (Nabhan 2001). Crop pollination is not a luxury. A conservative estimate puts one third of the U.S. food supply as directly dependant on plant/pollinator interactions, the most significant of which involve bees (Kremen et al. 2004). And 84% of 264 crop species studied either rely on or benefit from insect visits for seed production (Steffan-Dewenter and Tschamtkke 1999). A general decline in all pollinators has been linked primarily to pesticide use and misuse and loss of habitat due to conversion and fragmentation (Nabhan 2001). Honeybees, the primary vehicle of pollination in the U.S., are experiencing widespread devastation. Over the past fifty years, feral honeybee populations have dropped by 90% and managed colonies by two thirds (Kremen 2004). A portion of these heavy losses was due to infestations of parasitic (tracheal and *Varroa*) mites (Nabhan 2001). The risks of relying too heavily on a single species to meet our nation's pollination needs have received widespread media attention lately following the onset of Colony Collapse Disorder (CCD). Beekeepers reported unprecedented losses between 30% and 90% in the 2006-2007 winter (Holden 2007). Scientists have yet to identify a single or simple cause for this ailment, currently present in at least thirty-five U.S. states. One outcome of the honeybee crisis is that more attention is being given to the role of native pollinators in agriculture.

This situation presents a clear case where on-farm biodiversity enhancement may work directly in the grower's favor. In keeping with the notion that using land for human

food production need not occur to the exclusion of wildlife if it incorporates habitat features suitable for various species, attracting pollinators by providing them with habitat supplies the farm with an essential service in ensuring that flowers bear fruit. While past studies have led to the general conclusion that pollinators benefit from a farm's proximity to natural habitat, less is known concerning the significance of site level variables. Nor is there clear agreement as to whether the patchy resources typical of fragmented areas can support a stable population of pollinators.

Agriculture and Conservation

If a patchwork of managed systems between areas of unmodified habitat allows wildlife movement through means such as vegetation corridors, populations might not be limited exclusively to the boundaries of a reserve. Instead, they could move among a variety of protected areas. A study conducted around Mexico's El Triunfo Biosphere Reserve, an area surrounded primarily by shaded coffee plantations, demonstrated this possibility (Perfecto and Vandermeer 2002). Frugivorous forest bird species and ground-foraging ant species were observed to move throughout natural as well as human-managed areas, implying that the coffee plantations functioned as a sort of corridor allowing for movement outside the boundaries of the reserve. Work like this suggests that the habitat quality of agroecological landscapes should be viewed as integral to conservation strategy and policies (Perfecto 2003). This study is rather narrow in scope, however, for drawing general conclusions regarding the potential for agriculture to serve conservation purposes. While it shows the effectiveness of this habitat type for a handful

of species, that usefulness could vary widely depending on the species under consideration.

A common tendency in research has been to focus exclusively on the consequences of converting a natural system to an agricultural area. Agricultural land is often viewed as “lost” in terms of conservation purposes, or as though conversion mandates the creation of a biological desert. But now the suggestion is being offered that attention be given to the biodiversity impacts of different kinds of agroecosystems (Ramakrishnan 2000). Industrial, large-scale monoculture is not the only option. Many traditional societies have managed their systems in a landscape context that closely reflects the surrounding natural areas, allowing for movement and exchange between the two (Ramakrishnan 2000). And this is not merely a relic of the past. A recent estimate put the number of hectares surviving under traditional agriculture at more than three million (Heywood 1999). Examples of rural communities tending their productive lands in a manner that protects and even enhances biodiversity at a regional landscape level have been documented by a variety of authors (Halladay and Gilmour 1995; Collins and Qualset 1999; Vandermeer 2003). This argument, while inconclusive by nature since the baseline diversity prior to human habitation or intervention is not always readily available, still points an interesting direction. In summary, these authors point out that while conversion to agriculture inevitably entails modification and impacts on biodiversity, much might be learned from traditional societies regarding agroecosystem management and reducing the scale of impact.

Thus, the major problem may not be with agriculture itself, but with current practices. Modern agriculture tends to emphasize ecologically simplified systems that require high levels of external inputs (Altieri 1999) and place the focus primarily on high production. Managing agricultural systems with biodiversity in mind would mean granting equal importance to system resilience (Ramakrishnan 2000). It brings sustainability into the picture, and many studies have concluded that the high environmental cost of current mainstream agriculture threatens its viability in the long run (Altieri 1992). On this point, there is widespread agreement in the scientific community. Long-term experiments have clearly demonstrated that heavy reliance on ammonium or urea-based nitrogen fertilizers increases soil acidity, which in turn diminishes soil fertility, decreases plant and animal species diversity, and impedes nutrient cycling (Rasmussen et al. 1998). In some regions of California, improper field drainage and rising water tables are leading to alarmingly high levels of soil salinity (Altieri 1992). By implementing biodiversity-enhancing practices, agroecosystems might be able to play a beneficial role in conserving and contributing to the integrity of natural systems.

Van Elsen (2000) advocates a move from separation to integration of nature conservation and agriculture. Focusing primarily on the organic farming sector, he suggests establishing lists of criteria that support sustainable forms of agriculture. He goes on to claim that a division of the landscape in which a small percentage of “biotopes” persist where nature is given free reign (often as little as 5% on organic farms) and the other 95% is designated as productive area does not leave enough room for wild

species. His argument suggests that implementing organic practices is a step in the right direction, but the journey is far from complete.

Another school of thought advocates a different solution. Rather than attempting to farm less invasively, Green, et al. (2005) suggest that practicing high yield farming on smaller parcels of land might allow for the continuance of a greater number of species overall. Claiming a tendency for wildlife friendly farms to have lower production rates than conventional ones, the authors question whether conserving biodiversity and food demand from a growing human population are mutually exclusive goals in agroecosystems (Green et al. 2005). Their research did not address how varying levels of yield affect species found on farmed land or what impacts such a distinct process of fragmentation would have on the larger landscape. And using the concept of metapopulations, would this model restrict species to isolated areas of intact habitat? Without drawing any strong conclusions, they point to evidence from developing countries showing that farming is the largest threat to rare bird species and observations that biodiversity level decreases with increasing yield (Green et al. 2005). The conclusion they were confident to draw was that much more knowledge is needed. Information gaps they highlighted included determining the extent to which farming affects the wildlife of non-farmed areas, studying species population density across a spectrum of production models, and exploring further the outcome of land use when areas are spared from agricultural production (Green et al. 2005).

Agriculture and Biodiversity

The establishment of important links between biodiversity and ecosystem function remains one of the most compelling arguments for conservation (Hector et al. 2001). Agricultural biodiversity, or 'agrobiodiversity,' a term established in the 1990's, has been defined as "the diversity of genetic resources (varieties, breeds, species; cultivated, reared or wild) used directly or indirectly for food and agriculture; the diversity of species that support production (soil biota, pollinators, predators, etc.) and those in the wider environment that support agroecosystems (agricultural, pastoral, forest and aquatic), as well as the diversity of the agroecosystems themselves" (Pimbert 1999). An alternate definition supplied by Brookfield and Stocking is the "interactions between agricultural management practices, farmer's resource endowments, biophysical resources, and species" (1999). The International Federation of Organic Movements (IFOAM) uses the more general term biodiversity in their agricultural standards. They define it as "the variety of life forms and ecosystem types on earth, includes genetic diversity (i.e. diversity within species), species diversity (i.e. the number and variety of species) and ecosystem diversity (total number of ecosystem types)" (IFOAM 2005).

Ecological agriculture has been suggested as a biodiversity-friendly alternative to industrial, typically monoculture-based agriculture. A literature analysis that systematized the impacts of ecological and biodynamic farming methods on biological and landscape diversity found that low intensity agriculture practices significantly increased the diversity of organisms (primarily studied were vascular plants, birds, beetles, spiders, butterflies, bees, bumblebees, and earthworms) (Mander et al. 1999).

Vandermeer and Perfecto (1995) identified two components to agrobiodiversity: planned biodiversity such as intentionally cultivated crops and livestock, and associated biodiversity such as microfauna in soil and colonizers from the surrounding environment. Altieri (2001) concluded that there is a need to identify which kind of biodiversity is desirable for ecological services, determine what the best management practices would be based on that identification, and apply them in a cost-effective manner.

Increased biodiversity performs a variety of general ecological services. These include nutrient cycling, microclimate regulation, regulation of the water cycle, control of pests, and detoxification (Altieri 2001). Pimbert (1999) would add to this list production and yield efficiency, soil conservation, increased pollination and dispersal, and larger-scale climate regulation (as a source of atmospheric components, agriculture contributes significantly to the chemical makeup of the atmosphere). But as Altieri (2001) points out, there are many gaps in terms of biodiversity priorities for specific regions, and even more holes when it comes to the most effective methods of implementation.

An area where there is not clear agreement is with regards to assessing biodiversity. According to Duelli (1997), there is no established method in landscape ecology for an evaluation of biodiversity in agricultural landscapes. Although the means of assessing biodiversity are not tried and true, some methods for encouraging its re-establishment are being practiced. For example, Altieri (1999) references wide acceptance among agroecologists regarding the importance of field margins as havens for the natural enemies of crop pests, backing his statement with studies that demonstrate increased abundance of natural enemies in crops which are bordered by native vegetation.

The idea is to reintroduce a mosaic structure into the agricultural landscape with woodlots, fencerows, and hedgerows that serve as biological corridors allowing for movement and distribution as well as the creation of multiple habitats for reproduction, feeding, and sheltering of beneficial species (Altieri 1999).

Agriculture and Restoration

Some key strategies to restoring agricultural diversity have been suggested. Altieri (1992) describes increasing diversity in time through crop rotation and sequence and in space with practices such as cover cropping, intercropping, agroforestry, and mixing crops and livestock. Others have suggested the incorporation of "infield nature protection spots" (Berger et al. 2003). This entails changing the land use on specific parts of crop fields, typically areas with high habitat potential for wildlife, areas with extreme conditions, and portions of fields adjacent to ecotones such as a wood or pond edges. This particular study used the example of bird breeding habitat within fields and frog habitat within a crop that was next to a pond (Berger et al. 2003). The authors concluded that this was an efficient means of conserving biodiversity since only a small amount of productive arable land was lost, often without any measurable economic disadvantage to the farmer (Berger et al. 2003).

Landscape level restoration is often perceived as prohibitively costly because of lost agricultural production. A study in Southern Britain challenged this notion, though, when researchers found a positive relationship between grassland species diversity and hay production (Bullock et al. 2001). While more of these studies would need to be accomplished, and possibly at larger scales, before correlations could be made to

agriculture, these results are potentially significant. A study in a very different part of the world, eastern Amazonia, drew similar conclusions. Agricultural production on abandoned and degraded fields was most successful on sites that incorporated native trees into the farmscape (Nepstad et al. 1991).

Agriculture and Native Bees

One strand of diminishing biodiversity is that of native pollinators, namely bees. The diversity and geographic range of many wild bees has declined in recent decades (Westrich 1989). But native bees have only lately come into the spotlight, following the recent decline of the introduced European honey bee, *Apis mellifera*, due in part to parasitic tracheal and *Varroa* mites (Nabhan 2001). Batra (1995) points to historical evidence showing that honey bees became increasingly necessary in North American agriculture when intensive farming practices reduced local wild bee populations. Indeed, the leading threat to wild bees is listed as intensified agriculture (Osborne et al. 1999). Their absence would not go unnoticed, however. A conservative estimate puts a third of the U.S. food supply in direct dependence on plant/pollinator interactions, the most significant of which involve bees (Kremen et al. 2004). And eighty-four percent of 264 crop species studied rely on or benefit from insect visits for seed production and bees are among the most important pollinators in temperate regions such as the U.S. (Steffan-Dewenter and Tschamntke 1999). How much is this "free" service worth? Morse and Calderone put a \$14.6 billion price tag on the annual services of honeybees (2000). Following a similar model, Losey and Vaughan estimated a yearly \$3.07 billion contribution from native bees (2006). Driving the point even further, Allen-Wardell et al.

cite the management and protection of wild pollinators as critical to North American food supply systems (1998).

Might managed social bees fill the void? The answer is likely to be a no. Honeybees tend to visit only twenty to thirty percent of available plant species and consequently cannot guarantee overall pollination success (Roubik 1996). And for crops like blueberries and tomatoes that benefit from buzz pollination, honeybees simply are not ideal pollinators (Kevan 1999). A diversity of solitary bees with their variety of species-specific preferences are necessary in addition to social bees to keep the important interactions between plants and pollinators in working order (Steffan-Dewenter and Tschamtkke 1999). Increasingly, examples of this are coming to light. Chagnon, et al. (1993) found that strawberries produce the fullest, roundest fruits when both native and honeybees pollinate flowers. She and her team of researchers observed that smaller natives tend to pollinate the basal portions of the flowers, while larger-bodied honeybees focus their efforts on the apical portions.

Not only are native bees significant for crop pollination services, but they may also serve as general indicators of agroecosystem health. Relatively little is known regarding pollination services by wild bees or their vulnerability to environmental changes such as habitat loss or degradation (Kremen et al. 2002). Should they prove to be significant, these “free” pollination services could make conservation and restoration of bee habitat a viable economic alternative to dependence on managed honey bees (Kremen et al. 2002). And at a larger scale, findings could be such that on the subject of

crop pollination, ecosystem service arguments may align with debates in favor of biodiversity conservation (Kremen et al. 2002).

A study of crop pollination by native, unmanaged bee communities in California examining how pollination services varied on organic and conventional farms along a gradient of isolation from natural habitat found a significant and positive correlation between proportion of natural habitat area and pollination services (Kremen et al. 2004). The authors stated that this relationship existed independently to the factors of farm type, insecticide use, field size, and honeybee abundance. They concluded that stability and predictability of pollination services rises with a corresponding increase in natural habitat area (Kremen et al. 2004). Along similar lines, Steffan-Dewenter and Tscharntke found that frequency of flowers visits by all bee species decreased with increasing isolation from species rich grasslands (1999). And frequency and diversity of flower-visiting insects to grapefruit plantations in Argentina decreased with increasing distance from forest edge (Chacoff and Aizen 2006).

Ricketts' (2004) study in Costa Rica exploring the role of tropical forest remnants as sources of pollinators to adjacent coffee crops also reached similar conclusions. Bee richness, visitation rate, and pollen deposition rate was substantially higher in sites that were within 100 meters of forest fragments. In addition, following a decline in *Apis mellifera* populations during the second year of the study, he found a corresponding decline of pollination in isolated sites where *Apis* was the dominant pollinator. This led him to further surmise that these forest fragments provided surrounding crops with a

diversity of bees that increased the frequency as well as the stability of pollination services by reducing reliance on a single introduced pollinator (Ricketts 2004).

A three year study in Ontario, Canada of native and domestic insect pollinators isolated the factors of cultivar, east vs. west facing slope, temperature, humidity, and light intensity fluctuations and argued that temperature is the factor largely responsible for variation in numbers of honey bees and cultivar, slope position, and year are more significant for natives (Boyle-Makowski 1985). The author further concluded that in general, native pollinators are less affected by adverse conditions and appeared to compensate for reduced honey bee activity in less favorable times (Boyle-Makowski 1985). Another study arrived at similar conclusions regarding diversity's importance for sustainability due to year to year variation in populations (Kremen et al. 2002).

Findings are less agreed upon concerning the effect of agricultural intensification upon native bee populations. Contrary to common expectations, Klein (2002), in a study of the impacts of land-use intensity on bee pollination of coffee plants and community structure of trap-nesting bees and wasps, found that the number and abundance of all species of coffee-visiting bees has no significant correlation with land-use intensity. The abundance of solitary bees increased with land-use intensity while the abundance of social bees decreased (Klein 2002). The abundance and number of trap-nesting species also increased with intensification. The author used these results to support the idea that many benign and beneficial insects may stand to gain from agricultural land use (Klein 2002). A different study came up with contrasting results. An examination of various farms in Yolo County, California, that ranged from organic near natural habitat to organic

isolated from natural habitat to conventional concluded that agricultural intensification reduced the diversity and abundance of native bees (Kremen et al. 2002). But in order to make a real comparison here, it would probably be necessary to incorporate the quality of surrounding habitat as a significant factor. Kremen et al. (2002) also argued that isolation from important floral and nesting resources present in natural habitat was likely to be the key factor explaining the decline in native bees. Thus, proximity or incorporation of natural habitat may be the primary issue rather than agricultural intensification. Indeed, these authors went on to argue that only on organic farms near natural habitat were native bee communities capable of providing full pollination services for a crop with heavy pollination requirements such as watermelon (Kremen et al. 2002). Other farms, ones that were either managed organically but isolated from natural habitat or those managed conventionally, had greatly reduced diversity and abundance of native bees, and therefore insufficient pollination services from natives alone (Kremen et al. 2002).

The effect of fragmentation on native bees is also less than clear. As Cane (2001) points out, various authors, while citing habitat fragmentation as significantly harmful, have data which show that some native species do remarkably well in sampled fragments. He references other studies that report greater or comparable densities and diversity of native bees in fragments than in undisturbed habitats (Cane 2001). He includes a note that dramatic shifts in species composition can also accompany these results (Cane 2001). While there is no clear standard for what counts as a “fragment” and what size area constitutes “natural habitat,” recent work does seem to show that even fragments of native habitat can significantly enhance pollinator populations. In reference to restoration

efforts and strategies, Kremen et al. (2002), suggested that “patchy” resources can be effective since bees appear able to persist within small fragments of habitat. Other researchers have stated that bee populations seem more connected to the quality of patches than to their size or isolation (Gathermann et al. 1994; Tscharntke et al. 1998). Cane suggested that their mobility may make them less susceptible to fragmentation than some other insect groups (2001). But Steffan-Dewenter and Tscharntke, testing the hypothesis that habitat fragmentation affects bee populations and thereby disrupts plant-pollinator interactions, provide a sobering backdrop with the conclusion that habitat connectivity is crucial to maintaining abundant and diverse bee communities as well as plant-pollinator interactions in economically important crops and threatened wild plants (1999). Another area of potential fallout could follow. Diminished pollinator guilds could in turn change foraging behavior of flower-visiting insects, thereby further disturbing plant-pollinator interactions. There is the additional matter of which pollinators would cope best in a fragmented environment. The mean body size of flower-visiting bees was larger (these larger bees tend to have better flight capabilities) on more isolated habitat islands, insinuating that diversity would automatically decrease in fragmented habitats (Steffan-Dewenter and Tscharntke 1999).

Connectivity, defined as “the degree to which the landscape facilitates or impedes movement among resource patches,” (Baudry 2004) might be seen as fragmentation’s counterpart. A study tracking pollen transfer by bees and butterflies compared rates of transfer between patches connected by a corridor and others that were not to test the hypothesis that corridors increase the movement of pollinators into patches (Townsend

and Levey 2004). Their results supported the hypothesis. Another study by Gathmann and Tschamtkke in 2002 determined the maximum foraging distance between nest site and food patch to be 150-600 m for the sixteen solitary bee species examined. In this case, local habitat structure was determined to be more important than overall landscape structure, therefore necessitating the creation or preservation of a thick network of habitat patches throughout the landscape to provide for long-term sustainability of wild bee populations (Gathmann and Tschamtkke 2002).

Even a cursory examination of pollinator studies brings up questions regarding the scale at which they can be properly carried out. Results from a bumblebee foraging study supporting the hypothesis that they do not necessarily forage close to their nest demonstrate that study designs on a landscape scale are necessary in order to make decisions about the management of bees as pollinators and conservation of bee and plant biodiversity (Osborne et al. 1999). To complicate matters further, pollinator guilds appear to be affected differently at different spatial scales. For instance, Steffan-Dwenter et al. found that species richness and abundance of solitary wild bees showed a strong correlation with percentage of seminatural habitats at small scales (up to 750 m) while honeybees were correlated with a landscape context only at large scales (2002). These findings led them to conclude that local landscape destruction affected solitary wild bees to a greater extent than it did social bees. Their study also highlighted the fact that next to nothing is known regarding the relation between landscape composition and pollinator diversity.

Another study focusing on invertebrates sheds light on the issue of scale. A test of local vs. landscape controls on diversity of surface-dwelling soil macroinvertebrates found that local species richness may be strongly influenced by ecological processes operating at larger spatial scales (Dauber et al. 2005). In other words, information at both scales was necessary to explain local variation of species richness to a great extent. Researchers undertaking a bird study used aerial photographs to determine the relationship of bird abundance and landscape variables at a wider range of spatial scales than had previously been accomplished. They found scale to be a critical factor when examining effects of the environment on a species because species respond at different scales (Brennan and Schnell 2005). Consequently, analysis at multiple scales can offer insight into the spatial scales at which species respond. These findings lead to a conclusion that models and policies for the conservation of biodiversity in cultivated landscapes must address management of both habitat quality and the overall landscape mosaic (Dauber et al. 2005). Put differently, sound management involves combining both landscape design and land use practices (Baudry 2004). Here Baudry addresses the “scale gap between fine scale farm practices and coarse scale regional planning” (2004).

As the field of agroecology continues to emerge, it is obvious that many dots await connecting before a clear picture of the relationship between biodiversity and agroecosystems can be seen. Further study is needed to determine which farming practices benefit wildlife the most and how many species can realistically coexist with lower intensity agriculture. The use of biotic indicators in assessing overall ecosystem

health and relative biodiversity level is promising, but again, further study is needed for major conclusions to be drawn.

OBJECTIVES

The goals of this study were twofold. First, to develop an overall picture of habitat availability for pollinators on California's Central Coast. I compiled a map of the vegetation and land use in southern Santa Cruz County to serve as a visual representation of the farm/wildlife interaction in the region. This map can be used to inform regional conservation and restoration planning and serve as a decision-making tool in prioritizing future projects. The work was a cooperative effort with the Open Space Alliance of Santa Cruz and the Wild Farm Alliance, non-profit organizations dedicated in part to promoting sustainable agriculture and wildlife linkages between protected lands. The second goal of this research was to assess the benefits to bees of incorporating native species hedgerows into sustainable farming. This objective required a focused study involving strawberry crops, hedgerows, and native bees to explore whether a correlation exists between on-farm habitat restoration efforts (i.e., hedgerows) and native bee abundance and diversity.

Research Questions

The first goal was designed to address the following questions: (1) Do a substantial number of farms in the Pajaro watershed lie near or adjacent to natural habitat? (2) How might decision-makers prioritize conservation and/or restoration projects in the Central Coast region? The second objective addressed these questions: (1)

To what extent do native bees use patchy resources (i.e., hedgerows) in the Pajaro River and Elkhorn watersheds? (2) Do crops on farms without hedgerows receive fewer visits from native bees? (3) Do crops lying adjacent to hedgerows receive a higher number of visits from native bees than those lying at a distance from hedgerows?

STUDY SITE

The Pajaro River watershed, an area of approximately 1,300 square miles located in the southern portion of Santa Cruz County, contains over 75% of the region's farmlands (USDA 1997). Ranked fifth in agricultural production in California, with approximately 55,000 acres dedicated to cultivation, Santa Cruz County supports more than sixty crop species thanks to a growing season that averages 245 days. The Santa Cruz area is also notable for its number of organic farms, for which it currently ranks number one in the state. In 1996, over 102 farmers were registered as organic (Santa Cruz County Farm Bureau).

The Pajaro watershed lies adjacent to the Elkhorn Slough, one of the few remaining coastal wetlands in California and home to at least six threatened or endangered species. Historically, the Elkhorn Slough was part of a much larger wetland system covering the mouth of the Pajaro Valley. Following the Reclamation Period of the early 1800's, much of what had formerly been coastal wetland was drained to provide additional land for crops and livestock. At present, a wide variety of habitat types persist in the area, including oak woodland, grassland, maritime chaparral, salt marsh, tidal flats, and riparian areas.

METHODS

For Objective 1, I created a map of farms in the Pajaro Valley that compiled information from several sources: the Santa Cruz County Agricultural Commission, Santa Cruz County Geographic Information System (GIS), California Certified Organic Farmers (CCOF), and the California Natural Diversity Database (CNDDB). The Agricultural Commission granted me access to their database containing permit and site identification numbers for farmlands along with corresponding print outs showing the designation of agricultural parcels. Matching the site identification numbers to assessor's information from Santa Cruz County's GIS provided land ownership information as well as the specific locations of farms within the 1:24,000 scale quad sized files. The system's hydrology layer allowed me to determine which farms bordered riparian areas. I was able to make a further distinction as to which farms were managed organically by looking up farms in CCOF's directory database.

The CNDDB provided data on the locations of species allowing analysis of where farm and natural habitat overlapped. The CNDDB provides up to date information on the status and locations of rare and endangered plants, animals, and vegetation types throughout California digitally and in the form of map overlays with accompanying explanatory texts. Only actual sightings of rare species and natural communities are recorded in the database. The electronic CNDDB information was added to the Santa Cruz County's GIS map using ESRI ArcMap 9.1.0.750, showing farm location in relation to rare and sensitive plant and animal communities. To preserve this information in an

accessible and useful manner, I created a database using a File Maker program that included parcel number, farm name, whether it bordered a riparian area, and whether it was managed organically or conventionally.

For the second objective, I selected six farm sites located in relative proximity to the city of Watsonville, and lying within the Pajaro River and Elkhorn watersheds (see Figure1). All of the farms were managed with organic principles, and all of the selected sites were cultivating strawberries. Strawberries were targeted for several reasons. First, they are one of the more stable elements of a typical farmscape since they continue to flower and fruit throughout most of the growing season. They are also a significant component of the local economy. Strawberries continue to be ranked as the number one crop in the county, bringing in approximately \$107,408,000 in 2002 (Santa Cruz County Agricultural Crop and Livestock Report). In addition, they usually receive between 10% and 40% benefit in yields from pollinators (Klein et al. 2006).

The field sites reflected a spectrum of natural habitats on and/or surrounding the farm. Two of the sites possessed both hedgerows and connectivity to relatively undisturbed natural habitat. These sites were located on the Triple M Ranch in Las Lomas, a Farm Training and Research Center owned and operated by The Agriculture and Land Based Training Association (ALBA). Two other sites had hedgerows and little else in the way of native vegetation within the immediate vicinity. These two were Crystal Bay Farms on Zils Road and High Ground Organics on Lee Road. The final two sites were isolated from any kind of native habitat. Both were owned by Driscoll Strawberries and located on West Beach St., near Watsonville.

Figure 1. Map of study area. All sites were located in southern Santa Cruz County, CA.

Data were collected in July 2006 as the majority of crop pollination occurs during summer. Four sites with native plant hedgerows were contrasted with two sites that did not have hedgerows. The four-two ratio of hedgerow to non-hedgerow sites reflects the limited number of organic farms in the region.

Two primary sampling methods were employed. The first was visual assessment along 50 meter transects within strawberry rows. I followed standard protocol for a method known as the "bee walk," in which the observer keeps a slow and even pace along a transect line, carefully observing out from the mid-line and recording all flower visits by bees, which flower species was visited, and whether nectar, pollen or both were collected (Dramstad and Fry 1995). To supplement the visual assessment method, I also conducted passive sampling using a modified version of pan traps known as "bee bowls." Six-inch wide colored plastic bowls are filled with a mixture of water and unscented dish soap to disrupt surface tension and cause bees that land to drown. Bowls were dark blue and white, colors that are typical of bee-pollinated flowers (Russell et al. 2005). Two bowls, one of each color, were placed along strawberry row transects and left out during daylight hours on six different occasions during the sampling period. The dates were July 7, 8, 13, 14, 21, and 22, 2006.

Weather and time of day are significant factors influencing pollinator activity. To reduce bias due to weather conditions, sampling was limited to daylight hours and clear skies (Croxtton et al. 2002, Walther-Hellwig and Frankl 2000). To examine native bee presence on hedgerows, two bee bowls were randomly placed along the length of each hedgerow on the four hedgerow sites. To determine the significance of crop proximity to

hedgerows, I placed two of the bowl traps along strawberry rows at three different distances from the hedgerows: 15, 25, and 50 meters (see Figure 2). This design had to be modified at one site because the hedgerow was located at a distance greater than 15 meters from the strawberries. On that site, traps were placed at 50, 75, and 88 meters from the hedgerow. To further determine the influence of hedgerows, I sampled bees from strawberry rows on the farms that do not have hedgerows. Similar distancing and crop row placement was replicated on the non-hedgerow sites.

All bees present in the traps were removed with tweezers at the end of each sampling day, placed inside clear plastic film canisters, and then frozen. In November 2006, the specimens were thawed and pinned according to standard entomological guidelines for the purposes of identification. Initial identifications were confirmed at the University of California at Davis with the assistance of Dr. Robbin Thorp, Professor Emeritus.

One-way repeated measures ANOVA was used to determine species abundance and richness difference between hedgerow versus non-hedgerow sites over time and difference with respect to distance from hedgerows over time using SYSTAT 11.0. Between-date effects were tested using data from the six collection periods. Analysis occurred at two levels. Overall numbers of bees collected from all six sites were tested for the hedgerow's effect on bee abundance and diversity. At the four hedgerow sites, bee presence was analyzed at varying distances out from the hedgerow. The numbers of bees collected at the various distances were lumped into a field total for each site. Bowl

color was nested within treatment for all sites and within distance from edge for hedgerow sites.

Figure 2. Diagram of study design. Modified pan traps (bee bowls) were randomly placed within hedgerows and at varying distances out from the hedgerows.

RESULTS

Objective 1: Mapping Study

A substantial amount of overlap existed between agricultural land and habitat for rare, threatened, and endangered species in the southern portion of Santa Cruz County. Over 40 organic farms (outlined in green in Appendix) lay within or adjacent to areas identified as sensitive habitat by the CNDDDB or known to host rare or threatened species. The same was true for over 50 conventionally managed farmlands (outlined in black in Appendix). More than 300 agricultural parcels were located adjacent to riparian areas. The completed map (Appendix) shows a good deal of interface between local agriculture and biodiversity. For instance, approximately 455 of the 700 hectares identified as habitat for the threatened Santa Cruz tarplant (*Holocarpha macradenia*) shown in Appendix is agricultural land. About 221 out of 245 hectares identified as coastal and valley fresh marsh, a unique natural community known to support nesting tricolored blackbirds (*Agelaius tricolor*) and threatened California tiger salamanders (*Ambystoma californiense*), fall within farmland (see Appendix). Nearly all 184 hectares of the bank swallow's (*Riparia riparia*) habitat, (approximately 174), are designated as agricultural and 221 of 245 hectares of southern Santa Cruz County's central dune scrube vegetation community which supports the threatened Monterey spineflower (*Chorizanthe pungens* var. *pungens*) and the endangered sand gilia (*Gilia tenuiflora* ssp. *arenaria*) lie within agricultural borders (see Appendix). Farms also overlap with habitat of the endangered California red-legged frog (*Rana draytonii*) and the Western snowy plover (*Charadrius alexandrinus*) (see Appendix). A total of eleven plant and animal species classified as

rare, threatened, or endangered are known to inhabit this portion of Santa Cruz County (see Table 1).

Scientific Name	Common Name	Federal Status
<i>Ambystoma macrodactylum croceum</i>	Santa Cruz long-toed salamander	Endangered
<i>Ambystoma californiense</i>	California tiger salamander	Threatened
<i>Rana aurora draytonii</i>	California red-legged frog	Threatened
<i>Charadrius alexandrinus nivosus</i>	Western snowy plover	Threatened
<i>Oncorhynchus mykiss irideus</i>	Steelhead- Central California Coast	Threatened
<i>Oncorhynchus mykiss irideus</i>	Steelhead- South/Central California Coast	Threatened
<i>Holocarpha macradenia</i>	Santa Cruz tarplant	Threatened
<i>Chorizanthe pungens</i> var. <i>pungens</i>	Monterey spineflower	Threatened
<i>Chorizanthe robusta</i> var. <i>robusta</i>	Robust spineflower	Endangered
<i>Gilia tenuiflora</i> ssp. <i>arenaria</i>	Sand gilia	Endangered
<i>Pedicularis dudleyi</i>	Dudley's lousewort	Rare

Table 1: Federally listed species present in southern Santa Cruz County. A variety of rare, threatened, and endangered plants and animals persist in the agricultural landscape surrounding Watsonville, California.

Objective 2: Bee Study

Three-hundred-seventy-five bees were pinned and identified to genus or species. Four families were represented, thirteen genera, fifteen species, and three or more morphospecies. Figure 3 shows a summary of all bees that were collected. Members of the *Lasioglossum* and *Halictus* genera, both belonging to the Halictidae family, a group commonly known as “sweat bees,” were present in the greatest numbers. A total of seven species from the Halictidae family were collected. While *Lasioglossum* (*Dialictus*) spp. and *Halictus tripartitus* were the most common species, *Apis mellifera* (honeybees) and *Bombus vosnesenskii* (yellow faced bumblebees) ranked third and fourth in abundance, respectively (see Figure 3). The *Lasioglossum* were not identified to species.

Data collected from bee walks were far from conclusive. Some sites had more natives; others had more European honeybees (see Figure 4). No bees, native or otherwise, were observed at the Rancho Delfino site. There was a significant difference in both the number of species and abundance of solitary bees between sites with and without hedgerows (see Table 2). Total bee number and the three most abundant species were analyzed individually. Total bee number showed a significant difference between hedgerow and non-hedgerow sites (see Figure 5; Table 2; $P=0.006$). Species richness showed a similar pattern to overall bee abundance in that diversity was highest at hedgerow sites (see Figure 6; Table 3; $P=0.001$). *Lasioglossum* (*Dialictus*) spp. were present at all sites, but their abundance was greater on hedgerow sites (see Figure 7; Table 4; $P=0.007$). The difference in abundance was stark in the case of *Halictus tripartitus*, as no individuals were collected from non-hedgerow sites (see Figure 8; Table

5; $P=0.003$). *Apis mellifera* showed a different pattern. While they were collected in greater numbers from hedgerow sites, there was no significant difference in the number of individuals between hedgerow and non-hedgerow sites (see Figure 9; Table 6; $P=0.245$). Bowl color was not significant in determining species present ($P=0.872$).

Figure 3. Species collected in order of decreasing abundance. Halictids were most common, followed by honeybees and bumblebees. Sites Triple M 1, Triple M 2, High Ground, and Crystal Bay had hedgerows. Sites Rancho Delfino and Rancho Ring did not.

Figure 4. Native vs. honey bees recorded on bee walks. No clear trends were apparent in the number of native vs. honey bees recorded.

Table 2. One-way repeated measures ANOVA table for hedgerows vs. non-hedgerows over time for all bees.

Source	Sums of Squares	Degrees of Freedom	Mean Squares	F-Statistic	P-Value
Hedgerow	600.250	1	600.250	12.074	0.006
Error	497.125	10	49.712

Figure 5. Bee abundance and hedgerows. The mean number of bees collected at hedgerow sites was significantly higher than at non-hedgerow sites ($P=0.006$).

Table 3. One-way repeated measures ANOVA table for hedgerows vs. non-hedgerows over time for total species richness.

Source	Sums of Squares	Degrees of Freedom	Mean Squares	F-Statistic	P-Value
Hedgerow	84.028	1	84.028	19.888	0.001
Error	42.250	10	4.225

Figure 6. Species richness and hedgerows. The mean total of species was significantly greater on hedgerow sites ($P=0.001$).

Table 4. One-way repeated measures ANOVA table for hedgerows vs. non-hedgerows over time for *Lasioglossum (Dialictus)* spp.

Source	Sums of Squares	Degrees of Freedom	Mean Squares	F-Statistic	P-Value
Hedgerow	30.250	1	30.250	12.626	0.007
Error	19.167	8	2.396

Figure 7. *Lasioglossum (Dialictus)* spp. and hedgerows. Abundance was significantly higher on hedgerow sites (P=0.007)

Table 5. One-way repeated measures ANOVA table for hedgerows vs. non-hedgerows over time for *Halictus tripartitus*.

Source	Sums of Squares	Degrees of Freedom	Mean Squares	F-Statistic	P-Value
Hedgerow	30.250	1	30.250	15.714	0.003
Error	19.167	10	1.925

Figure 8. *Halictus tripartitus* and hedgerows. This halictid was only collected on hedgerow sites.

Table 6. One-way repeated measures ANOVA table for hedgerows vs. non-hedgerows over time for *Apis mellifera*.

Source	Sums of Squares	Degrees of Freedom	Mean Squares	F-Statistic	P-Value
Hedgerow	6.533	1	6.533	1.525	0.245
Error	42.850	10	4.285

Figure 9. *Apis mellifera* and hedgerows. There was no significant difference in honey bee presence on hedgerow vs. non-hedgerow sites.

At the four hedgerow sites, bee presence was analyzed at varying distances out from the hedgerow. Overall abundance was significantly greater within the hedgerow than out in the strawberry rows, but distance from the hedgerow did not appear to be a significant factor (see Figure 10; Table 7; $P=0.034$). In general, species richness data ran along similar lines to abundance with the highest level of diversity present in the hedgerow itself and little to no effect detectable at distances greater than 15 meters from native vegetation (see Figure 11; Table 8; $P=0.014$). Distance had no statistically significant effect on the presence of *Lasioglossum (Dialictus)* spp. (see Figure 12; Table 9; $P=0.107$). Likewise, distance from the hedgerow had no visible effect on *Halictus tripartitus* presence (see Figure 13; Table 10; $P=0.539$). Nor was distance a critical factor in honeybee (*Apis mellifera*) presence, and they did not appear to be any more present within the hedgerow (see Figure 14; Table 11; $P=0.353$).

DISCUSSION

Species function as members of larger communities, all contributing in various ways to overall ecosystem health. Bees have a specific set of habitat needs, as do snowy plovers and tiger salamanders, but these needs often intersect in shared geographic areas. Hence, conservation efforts aimed at rare, threatened, and endangered species can indirectly benefit pollinators, as well. The map produced in this study shows opportunities to develop habitat for a number of important species that are all part of the natural communities that bees need. Agriculture profits from pollination by wild bees,

bees cannot exist in an ecosystem devoid of other species, and taken together all of these species need habitat and natural communities.

Table 7. One-way repeated measures ANOVA table with distance as the factor and bee abundance as the dependent variable.

Source	Sums of Squares	Degrees of Freedom	Mean Squares	F-Statistic	P-Value
Distance	184.191	4	46.048	3.287	0.034
Error	252.153	18	14.008

Figure 10. Bee abundance and distance. Though total bee numbers were higher within the hedgerow, abundance was not greatly affected by varying distance out from the hedgerow ($P=0.034$).

Table 8. One-way repeated measures ANOVA table with distance as the factor and species richness as the dependent variable.

Source	Sums of Squares	Degrees of Freedom	Mean Squares	F-Statistic	P-Value
Distance	40.397	4	10.099	4.074	0.014
Error	49.583	20	2.479

Figure 11. Species richness and distance. Total number of species was significantly higher within hedgerows ($P=0.014$). Differences diminish at varying distances out from the hedgerow, however.

Table 9. One-way repeated measures ANOVA table with distance as the factor and *Lasioglossum (Dialictus)* spp. abundance as the dependent variable.

Source	Sums of Squares	Degrees of Freedom	Mean Squares	F-Statistic	P-Value
Distance	45.727	4	11.432	2.221	0.107
Error	92.639	18	5.147

Figure 12. *Lasioglossum (Dialictus)* spp. and distance. Distance had no statistically significant effect on the presence of this species ($P=0.107$).

Table 10. One-way repeated measures ANOVA table with distance as the factor and *Halictus tripartitus* abundance as the dependent variable.

Source	Sums of Squares	Degrees of Freedom	Mean Squares	F-Statistic	P-Value
Distance	4.980	4	1.245	.804	0.539
Error	27.875	18	1.549

Figure 13. *Halictus tripartitus* and distance. Distance from hedgerows had no measurable effect on this species (P=0.539).

Table 11. One-way repeated measures ANOVA table with distance as the factor and *Apis mellifera* abundance as the dependent variable.

Source	Sums of Squares	Degrees of Freedom	Mean Squares	F-Statistic	P-Value
Distance	5.285	4	1.321	1.176	0.353
Error	21.350	19	1.124

Figure 14. *Apis mellifera* and distance. There was no statistically significant effect of distance from hedgerows on honey bees ($P=0.353$).

The heightened number of species and greater abundance of bees on sites with hedgerows adds another argument for managing farms to support biodiversity. On each of the four hedgerow sites, native plantings were located along uncultivated field margins or fence lines. Consequently, there was no need for farmers to take land out of production in order to provide additional habitat and forage for pollinators and other beneficial insects. The results of this study suggest that investing in native vegetation can have a valuable return in the form of pollination service. Additionally, the presence of native bees 50 meters away from the hedgerow implies that the benefits of hedgerows, rather than being a localized effect, extend well out into the fields.

The conspicuous absence of bees from the two non-hedgerow sites, as shown by this study, was due in part to a corresponding lack of native vegetation. It is also interesting to note that both of these sites were blocks of organically managed strawberries surrounded by fields of conventionally managed crops. The close proximity to pesticides could also have contributed to a diminished pollinator population.

The finding that bees within the *Halictidae* family were most abundant in the samples came as no surprise since halictids, or sweat bees, are often found in association with strawberries. Their presence is notable because they are significant contributors to strawberry pollination (Chagnon, et al. 1993). Relative to other native bee species collected, *Lasioglossum (Dialictus)* spp. presence showed a fairly weak correlation with hedgerows. There are several possible explanations for this. Perhaps these bees rely less on the presence of native vegetation than other species do. It could also mean that

Dialictus tend to travel greater distances from their nests than some of the other halictids. This would be surprising however, given their small size.

Peponapis pruinosa, a bee that obligately forages on squash for pollen (Kremen 2004), was collected from two of the sites, suggesting that this bee visits other crop species, as well. Two more specialists, *Diadasia bituberculata* and *Diadasia rinconis*, were also collected at those sites. Bees of this genus are known to be associated with cactus, further suggesting that these specialists may forage on a variety of flowering plants.

Apis mellifera, the European honeybee, was the only species whose presence was not significantly greater on hedgerow sites. A true generalist, the non-native honeybee is unlikely to have close ties to native vegetation. The data suggest that native hedgerows neither attract nor discourage honeybees. More data may show native plants actually increase their abundance since a greater number of honeybees were collected from hedgerow sites.

It is evident from this study that native bees in the Pajaro and Elkhorn watersheds use patchy resources. It is also clear that hedgerows benefit native bees to a greater extent than non-native honeybees. Hedgerows help attract a diverse pollinator population to farms. In the wake of Colony Collapse Disorder and the realization that relying too heavily on any single species for pollination is an unwise policy, measures should be taken to draw a wide array of pollinators to farmland. This research supports conservation plantings such as hedgerows as a means of accomplishing that goal.

Recommendations

In terms of management implications, hedgerows, as a form of native habitat restoration, are a potentially viable means of maintaining and increasing a diverse native pollinator presence on farms. Thus, installing more hedgerows on farms in the Central Coast area should encourage regional pollinator abundance and diversity. Hedgerows are compatible with food safety guidelines and a variety of incentive programs are available through local Resource Conservation Districts to assist farmers in obtaining the resources necessary for installing conservation plantings.

For maximum pollination effectiveness, hedgerows should be located adjacent or within close proximity to crop fields. The primary components of hedgerows designed to serve as windbreaks and attract beneficial insects including pollinators are shrubs that achieve a height of between four and eight feet. Species recommended for use as the backbone of hedgerows in the central coast region of California include *Ceanothus* spp. (California lilac), *Rhamnus californica* (coffeeberry), *Baccharis pilularis* (coyote brush), *Heteromoles arbutifolia* (toyon), *Eriogonum* spp. *fasciculatum*, *giganteum*, *parvifolium*, and *latifolium* (buckwheat), and *Atriplex lentiformis* (saltbush). Due to their size at maturity, four-foot spacing is recommended when planting these shrubs. A variety of other forbs, grasses, shrubs, and trees can be added to the list to further increase the biodiversity benefits to farms (Earnshaw 2004).

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Open Space Alliance Pajaro Valley Farms and Wetlands

The California Natural Diversity Database (CNDDB) is a program that inventories the status and locations of rare plants and animals in California. CNDDB staff work with partners to maintain current lists of rare plants and animals, and to identify and protect their habitats. CNDDB provides information for these agencies. For more information see <http://www.dfg.ca.gov/nature/conservation.html>.