

Community and Species-Specific Responses of Wild Bees to Insect Pest Control Programs Applied to a Pollinator-Dependent Crop

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ABSTRACT Wild bee conservation is regarded as essential for sustainable production of pollinator-dependent crops, yet little is known about the effects on wild bee communities of typical insect pest management programs used postbloom. We developed an insecticide program risk (IPR) index to quantify the relative risk to wild bees of insecticide programs applied to blueberry fields. This was used to determine the relationship between IPR and the abundance, diversity, and richness of wild bee communities sampled during three successive flowering seasons. In 2 of 3 yr, bee abundance and species richness declined with increasing IPR. Bee diversity declined with IPR in one of 3 yr. These results indicate that wild bee communities are negatively affected by increasingly intensive chemical pest management activities in crop fields and that interyear variability in bee populations has the potential to mask such effects in short-term studies. When several wild bee species were analyzed separately, two of three solitary and one of three social blueberry-foraging species declined with increasing IPR values, suggesting that different life histories and nesting habits may help some bee populations escape the negative effects of insecticides applied after bloom. Pollinator conservation programs aimed strictly at reducing insecticide use may have varying success, depending on the biology of the target bee species. The IPR index provides a standard method to compare pest management programs for their potential effect on wild bee communities, with broad application for use in other agricultural systems.

KEY WORDS Apoidea, insecticides, nontarget effects, *Vaccinium corymbosum*, wild bee conservation

A major challenge of sustainable crop production is responding to pest pressure while protecting beneficial organisms that provide important ecosystem services. This challenge is particularly relevant for producers of crops that require bee-mediated pollination to achieve economically viable yields, including many fruits and vegetables (Kevan et al. 1990, Williams 1994, Klein et al. 2007). Most of these crops can be pollinated by managed honey bees, *Apis mellifera* L., but with declining honey bee populations and fewer beekeepers (Cox-Foster et al. 2007, National Academy of Sciences 2007, Stokstad 2007), there is increasing interest in the conservation of wild bees to support sustainable crop pollination. In some crop systems, wild bees are the main pollinators (Winfree et al. 2007), can increase the effectiveness of managed pollinators (DeGrandi-Hoffman and Watkins, 2000, Greenleaf and Kremen 2006), or have been found to be more efficient pollinators than honey bees (Javorek et al. 2002).

Insecticides have been cited as a potential cause of bee declines (Kearns et al. 1998, National Academy of Sciences 2007), but evidence for this under typical agricultural conditions is limited. There is evidence

that applications of certain classes of insecticides when applied to crops where bees are foraging can cause acute bee mortality (Johansen 1977, Johansen and Mayer 1990) and that applications of persistent broad-spectrum insecticides to habitat adjacent to crop fields can cause rapid bee declines in adjacent crop fields (Kevan 1975, Kevan et al. 1990). Despite caution used by many growers when bees are active during bloom, insecticides applied at other times during the season may have chronic effects on bee communities that are more challenging to quantify (Desneux et al. 2007). This is partly because the lethal and sublethal effects of pesticides on wild bees are not as well understood as they are for honey bees (Tasei 2002, Morandin and Winston 2003). For example, of the >4,000 bee species known to occur across North America, only a handful have been included in toxicological studies that are equivalent to those conducted for honey bees (see Tasei 2002). Many species of wild bees are active in and around crop fields far longer than the bloom of the crop and often nest directly in crop fields, which has the potential to increase their chance of lethal pesticide exposure compared with honey bees that are only placed in crop fields during bloom (e.g., Thompson and Hunt 1999). It is also challenging to quantify the risk of pest

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management programs that include multiple insecticides with different modes of action and physical properties, applied at different rates per hectare, and applied at different times of the growing season over landscapes of varying complexity and bee community composition.

Native bees exhibit a wide range of responses to pesticide applications to crops (Tasei 2002), and a recent meta-analysis of anthropogenic factors affecting bee populations detected no significant link between typical crop pesticide applications and bee abundance (Winfree et al. 2009), although the number of studies that have investigated this issue is small. Comparisons of bee communities in organic and conventional agriculture have either detected no differences based on the pesticide program (Kremen et al. 2002) or have found that increasing organic agriculture at the landscape level is required to increase wild bee richness and density (Gabriel and Tschamtko 2007, Holzschuh et al. 2007). Kremen et al. (2004) grouped insecticides into four categories based on the LD₅₀ (the dose lethal to 50% of individuals) for honey bees and the known residual activity of each insecticide, and then used field area and the number of applications to create an index of insecticide use on watermelon farms, but they found no significant relationship between values of this index and pollination services to the crop. Using a binomial variable of pesticide/no pesticide application, Shuler et al. (2005) also found no significant relationship between wild bee abundance and pesticide use in cucurbit fields in the eastern United States. If typical insecticide programs applied for pest control have any effect on wild bee communities, this is most likely to be detected in systems with a high abundance of wild bees in fields receiving pesticide applications that vary widely in the amount and toxicity of the pesticides applied.

Highbush blueberry, *Vaccinium corymbosum* L., requires bee-mediated pollination for economically viable yields (Free 1993, Delaplane and Mayer 2000, Dogterom et al. 2000) and is grown in Michigan under a wide range of management intensity that may include from zero to eight insecticide applications per season depending on pest pressure (R.I., unpublished data). A rich wild bee community is present before, during, and after blueberry bloom, with >100 species of wild bees found in these fields, of which ≈10 species exhibit high abundance and/or fidelity to *Vaccinium* flowers (Tuell et al. 2009). This community includes species whose phenologies are tied closely to crop bloom and others that are found through the season, putting them at greater risk of exposure to the insecticides applied for insect pest control. This provides an opportunity to test the hypothesis that insecticide applications made when the crop is not in bloom affect the community of wild bees present during the bloom period.

In this study, we developed a new index to quantify the combined effect of insecticide toxicity to bees and the amount of insecticide applied throughout a growing season. This index was used to test the hypothesis that wild bee communities active during bloom are

negatively affected by increasing intensity of chemical insect pest management. By conducting the study over three growing seasons, we also tested the temporal stability of the bee community response to insecticide intensity. Bees with activity throughout the growing season were expected to be more affected by the insecticide intensity than those with activity restricted to bloom, and this hypothesis was tested by comparing the response of abundant bees from both groups.

Materials and Methods

This study was conducted at fifteen highbush blueberry fields within 20 km of Lake Michigan in the southwestern lower peninsula of Michigan. Sites were located <300 m above sea level, at least 3 km apart from one another, and ranged from 1.2 to 4 ha, embedded within a region of intensive blueberry production. Two of the farms were managed with minimal chemical or mechanical input and 13 were commercially managed, including a range of insecticide use. Pan trapping, a common method for monitoring bee populations (Aizen and Feinsinger 1994, Leong and Thorp 1999, Wilson et al. 2008, Droege et al. 2010), was used to compare bee communities among sites.

Due to varying early spring weather conditions from year to year, pan trapping was conducted twice in 2004 (16 May and 3 June), three times in 2005 (16–25 May), and twice in 2006 (17–31 May) at each field (Tuell et al. 2009). Pan traps consisted of 355-ml white and yellow plastic bowls (Amscan, Inc., Elmsford, NY) mounted onto 2.7-cm-diameter by 1.2-m polyvinyl chloride poles stabilized with rebar (Tuell and Isaacs 2009). Traps were filled halfway with a 2% soap solution (Dawn dish soap, Procter & Gamble, Cincinnati, OH). Pairs of white and yellow pan traps were placed 5 m apart along each of two transects running perpendicular to the orientation of the rows for a total of 20 traps. One transect was established within 1 m of the field edge and the other was established 25 m into the field. Traps were set out between 0800 and 1200 hours and were collected between 1600 and 2000 hours for a minimum trapping period of 6 h on days when wind velocity was <10 mph and ambient temperature was at least 13°C with clear or partly cloudy skies or 17°C with any sky condition other than rain. The number of bees (bee abundance) and the number of bee species (species richness) captured at each site were averaged across traps and samples each year. To determine the variance of the species abundance distribution, Simpson's index ($1/D$) was used to calculate bee diversity for each site in each year (Magurran 2004).

To evaluate the risk of insect pest management programs to bees, we developed an index that provides a value of insecticide program risk (IPR) for each field in each year (see equation 1). This is similar to the hazard ratio developed in risk assessment programs for honey bees (EPPO 1993, Mineau et al. 2008), but it integrates across all of the insecticides applied during a growing season. IPR values were calculated using insecticide application records for 2003, 2004, and 2005

Table 1. Insecticides applied during 2003–2005 at blueberry fields sampled for bees in 2004–2006 (insecticides were used one or more times during this study)

Trade name	(AI)	Chemical class	Targeted pests ^a	kg (AI)/ha ^b	LD ₅₀ (μg/bee) ^c	Relative residue duration ^d
Diazinon 50WP	Diazinon	Organophosphate	BBA	1.13	0.37 ^e	0.75
Guthion 50WP	Azinphos-methyl	Organophosphate	FW, OBLR	0.56–0.84	0.43 ^e	0.75
Imidan 70WP	Phosmet	Organophosphate	BBM, JB, OBLR	0.39–1.05	1.13 ^e	0.75
Malathion 8F, 8EC, LV	Malathion	Organophosphate	BBM, JB	0.93–2.3	0.73 ^f	0.25
Lannate 2.4LV, 90SP	Methomyl	Carbamate	FW, BBA	0.51–1.02	1.29 ^f	0.25
Sevin 50W	Carbaryl	Carbamate	BBM, JB, OBLR	0.88–1.76	1.50 ^e	0.5
Sevin 80WSP	Carbaryl	Carbamate	BBM, JB, OBLR	1.62–1.8	1.50 ^e	0.5
Sevin XLR+	Carbaryl	Carbamate	BBM, JB, OBLR	1.04–2.07	26.5 ^e	0.5
Asana XL 0.66EC	Esfenvalerate	Pyrethroid	FW, BBA	0.03–0.05	0.06 ^e	0.5
Provado 1.6F	Imidacloprid	Neonicotinoid	BBM, JB, BBA	0.08–0.1	0.0179 ^h	0.25
Spintor 2SC	Spinosad	Naturalyte	BBM	0.1	0.058 ⁱ	0.5
Confirm 2F	Tebufenozide	Growth regulator	FW, OBLR	0.24–0.27	234 ^e	1.0
Dipel	<i>B. thuringiensis</i>	Biological	FW	0.06–0.12	nd	0.1

^a FW, cranberry and cherry fruitworms; JB, Japanese beetle; BBM, blueberry maggot; OBLR, obliquebanded leafroller; BBA, blueberry aphid.

^b Based on application rates reported by the grower cooperators.

^c Topical LD₅₀ for honey bees; nd, no data, but considered to be nontoxic to bees.

^d Ranking of the duration of relative residual activity on the plant surface based on expert opinion (0–1 scale).

^e University of California (1981), ^f Atkins and Kellum (1986), ^g PPDB (2009), ^h Iwasa et al. (2004), ⁱ Mayes et al. (2001).

supplied by the owners of each field because the effects of insecticide applications, almost all of which occur after bloom, are expected to be observed in the bee community that emerges the following spring. All grower decisions regarding pesticide applications were independent of the authors. The amount (kilograms) of active ingredient (AI) applied per hectare was first determined for each insecticide application from the product of the application rate and the proportion of AI for each product applied. This value was divided by the contact LD₅₀ value for honey bees for each insecticide used (Table 1). The values for all insecticide applications applied to each field during each year were then summed to provide unique IPR scores for each field per season (equation 1).

$$IPR = \sum \frac{kgAI/ha}{HoneybeeLD_{50}} \quad [1]$$

The LD₅₀ for *A. mellifera* was used because values for wild bees are not available for all insecticides, and this is the most complete data set. In addition, the relative magnitude of the values is expected to reflect the relative toxicity of these products to other bee species. Details of the insecticides applied to blueberry fields in this study and their LD₅₀ values for *A. mellifera* are listed in Table 1.

The exposure component of the index given in equation 1 [kg(AI)/ha] gives equal weight to insecticides applied at different times of the season. However, because bee species vary in the length of their adult activity periods, we also calculated an adjusted IPR score (IPR_t) that uses the same equation with the addition of a weighting factor based on the timing of each spray application (see equation 2). Thus, May/June applications were weighted by 1, July applications were weighted by 2, and August/September applications weighted by 3, to correspond with the increased abundance of bees on flowering plants observed throughout the season (Tuell et al. 2008).

$$IPR_t = \sum \frac{(kgAI/ha) * (time\ of\ application)}{HoneybeeLD_{50}} \quad [2]$$

Insecticides also have varying residual activity on plant surfaces, with variable decay rates under UV exposure, or absorption by plant tissue via translaminar movement. The duration of residual activity after application adds to the risk of exposure. To determine whether insecticide residual activity helped to explain the response of bees to IPR, a second adjusted index (IPR_r) was calculated that multiplied each application's risk score by a value between 0.1 and 1, with 0 assigned to chemicals with the shortest relative residue duration, and 1 assigned to chemicals with the longest relative residue duration, based on expert opinion (Table 1).

$$IPR_r = \sum \frac{(kgAI/ha) * (residual\ activity)}{HoneybeeLD_{50}} \quad [3]$$

Mantel tests (calculated with the “vegan” package for R 2.3.1) were used to compare pairwise bee community similarity indices (Jaccard, Bray-Curtis, and Morisita-Horn) with pairwise geographic distances between each of the 15 blueberry fields in each year. The degree of spatial autocorrelation in nonpairwise variables (bee abundance, species richness, and Simpson's 1/D diversity index) was assessed using Moran's I test (using the “ape” package for R 2.3.1) for each year. Data from each year were analyzed separately because the bee community composition and trends in response to management intensity were not similar among years.

To determine whether wild bee abundance, species richness, or diversity (Simpson's 1/D) during bloom varied with the IPR, IPR_t, or IPR_r indices, we performed linear regression analysis (JMP 8.0.2), using $\alpha = 0.05$. To test whether individual species responded differently to insecticide programs applied to blueberry fields, the three most abundant solitary bee

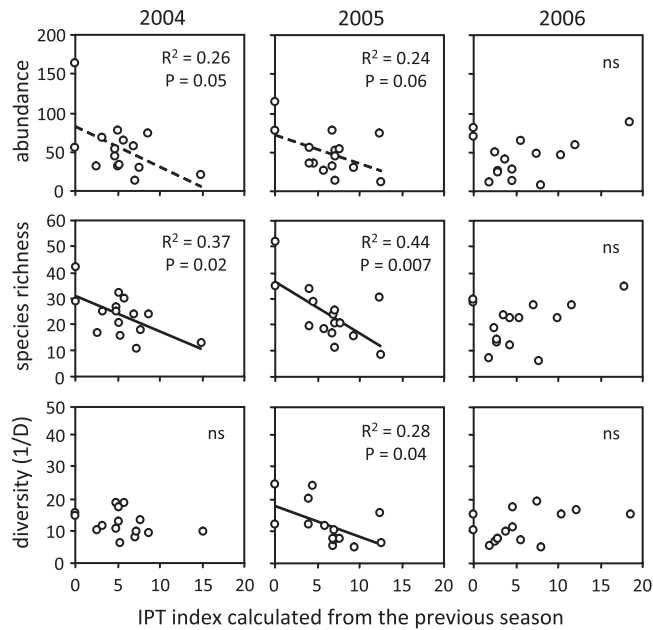


Fig. 1. Wild bee abundance, bee species richness, and bee diversity (1/D) as a function of IPR, based on insecticide applications made in the year before bee sampling. Best-fit linear regression lines are solid for regressions where $P < 0.05$ and are dashed for regressions where P is between 0.05 and 0.1.

species that are abundant during bloom (*Andrena carlini* Cockerell [4% of all bees trapped by Tuell et al. 2009], *Andrena carolina* Viereck (14%), and *Andrena vicina* Smith [4%]), and the three most abundant non-*Apis* social or semisocial bee species that are active through the postbloom period [*Augochlorella aurata* (Smith) (5%), *Lasioglossum imitatum* (Smith) (4%), and *Lasioglossum pilosum* (Smith) (8%)], were each analyzed separately using the same methods. These species are all known to forage on *Vaccinium* (Mitchell 1960, 1962; Hurd 1979; MacKenzie and Eickwort 1996; Tuell et al. 2009).

Results

Over 3 yr and seven sampling events at each of the 15 farms during bloom, 3,228 non-*Apis* bees were captured, with 70–86 species captured each year and a total of 112 species captured over the 3 yr (Tuell et al. 2009). Captures of wild bees varied widely across farms, with traps collecting between 5 and 260 bees at different farms during bloom each year. Site proximity was not related to bee community similarity (Bray-Curtis index; Mantel $Z = 1729.3$, $df = 14$, $P = 0.143$) or bee abundance, species richness, or diversity (Moran's $I < 0.514$, $df = 13$, $P > 0.05$), indicating no spatial autocorrelation among sites. For subsequent analyses, we therefore assumed that sites represented independent samples.

Bee abundance and species richness declined with increasing values of the IPR index in 2 of the 3 yr of this study (Fig. 1), with up to 44% of the variation being explained by the magnitude of IPR. Regression analysis revealed less abundant wild bee communities

in 2004 and 2005 at farms with higher IPR values; 2004: $F_{1,13} = 4.58$, $P = 0.05$; 2005: $F_{1,13} = 4.12$, $P = 0.06$; and 2006: $F_{1,13} = 1.16$, $P = 0.30$ (Fig. 1). The species richness of wild bees trapped at the fifteen fields revealed a similar pattern of declining richness as IPR increased; 2004: $F_{1,13} = 7.76$, $P = 0.02$; 2005: $F_{1,13} = 10.08$, $P = 0.007$; and 2006: $F_{1,13} = 1.95$, $P = 0.19$ (Fig. 1). Wild bee species diversity was negatively correlated with the insecticide program risk only in the middle year of this study; 2004: $F_{1,13} = 1.83$, $P = 0.20$; 2005: $F_{1,13} = 5.06$, $P = 0.04$; and 2006: $F_{1,13} = 2.79$, $P = 0.12$ (Fig. 1).

The relationships between IPR and wild bee abundance, richness, and diversity became stronger with the addition of the insecticide application timing factor, IPR_t (see equation 2). Regression analysis using the IPR_t improved the regression coefficient values in most instances, particularly for the bee data collected in 2005. Thus, the regression coefficient values for the decline in each measure of the bee community in 2005 with IPR_t were 0.38 ($P = 0.02$) compared with 0.24 for bee abundance, 0.53 ($P = 0.002$) compared with 0.44 for bee richness, and 0.36 ($P = 0.02$) compared with 0.28 for diversity. The base model (IPR) was also improved for bee abundance and species richness when it was weighted by the relative residue duration for each chemical, IPR_r (see equation 3; Table 1). Regression coefficients for both bee abundance and species richness increased in 2004 (abundance: $R^2 = 0.34$, $P = 0.02$; richness: $R^2 = 0.41$, $P = 0.01$) and 2005 (abundance: $R^2 = 0.47$, $P = 0.005$; richness: $R^2 = 0.47$, $P = 0.005$) over the base model.

The three most abundant solitary social wild bee species collected during this study known to forage on

Table 2. Regression coefficients for linear regressions of six *Vaccinium* pollinating wild bee species and the IPR index modified by time of application (IPR_t) and by residual activity of the insecticides (IPR_r) from blueberry farms in southwest Michigan (regression model: \log_{10} of bee abundance = IPR)^a

Bee species	Seasonality	Sociality	Avg. bees per farm (total bees) ^b	Index	R ²		
					2004	2005	2006
<i>Andrena carlini</i>	Spring	Solitary	0–7.0 (133)	IPR	0.33		
				IPR _t	0.27		
				IPR _r	0.39		
<i>Andrena carolina</i>	Spring	Solitary	0–17.3 (533)	IPR			
				IPR _t			
				IPR _r			
<i>Andrena vicina</i>	Spring	Solitary	0–3.1 (132)	IPR	0.29		
				IPR _t			
				IPR _r			
<i>Augochlorella aurata</i>	Summer	Social	0–12.8 (174)	IPR	0.32		
				IPR _t	0.28		
				IPR _r	0.39		
<i>Lasioglossum imitatum</i>	Summer	Social	0–32.0 (124)	IPR			
				IPR _t			
				IPR _r			
<i>Lasioglossum pilosum</i>	Summer	Social	0–10.3 (264)	IPR			
				IPR _t			
				IPR _r			

^a These are the three most abundant solitary and social bee species captured during blueberry bloom. Only significant R² values ($P < 0.05$) are shown.

^b Across all sites over 3 yr.

Vaccinium exhibited varying responses to the intensity of insect pest management programs within and among years. Of the three spring active solitary bee species analyzed, *Andrena carlini* Cockerell declined with increasing IPR using all three indices and *A. vicina* declined with increasing IPR and IPR_r, but this was seen only in 2004 (Table 2). Similarly, one of the three summer-active social species, *Augochlorella aurata* Smith, declined with IPR and IPR_r, but only in 2004 (Table 2). Abundance of *Andrena carolina* Verrick, a *Vaccinium* specialist (LaBerge 1980), and *Lasioglossum imitatum* Svensson and *Lasioglossum pilosum* (Smith), two social species, did not vary significantly with the indices in any of the years (Table 2).

Discussion

Within agricultural systems, it is expected that individual growers will vary the intensity of their insect pest management practices which in turn will affect the abundance of wild bees that inhabit their farms. Our results show that fields treated with plant protection programs having high overall risk (high IPR values) are less likely to support abundant and diverse wild bee communities, although this relationship was variable from year to year. For pollinator dependent crops, reduced abundance and diversity of wild bees are expected to reduce the stability of pollination services (Kevan 1977, Ricketts 2004, Ricketts et al. 2004; Winfree and Kremen 2009) and cause growers of insect pollination-dependent crops to become increasingly reliant on managed bees such as *Apis mellifera* (Williams 1994).

This study provides evidence for negative impacts of season-long pest management programs on the wild bee community in blueberry fields. The IPR index

developed here revealed negative relationships between the intensity of insecticide programs applied to blueberry fields and the wild bee community present during bloom the following year. This novel index combines the quantity and toxicity of insecticides applied to agricultural fields throughout a growing season. It may be used by land managers to compare the risk of their management practices to bees, and to identify pesticide applications that could be removed or replaced with less bee-toxic alternatives while maintaining pest control. It may also be a useful tool for other researchers studying the effects of farming intensity on wild bee ecology in crops that receive applications of pesticides.

Risk indices have been developed to quantify the impact of pest management programs on human health and the environment, with toxicity to pollinators as one of many parameters (e.g., Kovach et al. 1992). Some more specific indices have been developed to determine the effect of typical chemical crop management inputs on bees (e.g., Kremen et al. 2004), but these have rarely detected strong responses. More recently, analysis of honey bee poisoning records by Mineau et al. (2008) found that application rate, contact toxicity, and area treated were predictors of poisonings, but the relevance of their analysis to wild bees in crop fields having different biology and sensitivity to toxicants remains unknown.

The higher regression coefficient values found after the addition of temporal components in the two adjusted IPR indices indicate that the timing of application and the duration of residual insecticide toxicity are important components of the risk experienced by bees in insecticide-treated crops. Although the basic IPR index presented here can be calculated knowing only the amount, formulation, and active ingredient of insecticides applied to fields, more refined IPR values

that incorporate timing of application, residue aging profiles, wild bee toxicity components, and spatial extent of applications may find utility for accurately estimating risk of pest management activities to bees. Foraging ranges of many bees exceed the scale of a single crop field (Gathmann and Tschamtko 2002, Greenleaf et al. 2007) and the bee community captured in one field likely experiences insecticides applied in other fields not included in this index. Incorporating the IPR index into landscape analyses such as that of Lonsdorf et al. (2009), rather than our single field approach, may improve the utility of this index because this will better reflect the area over which bees are known to forage. As these data and approaches become increasingly available, land managers aiming to reduce the risk of their farming practices to wild bees may be able to employ software to include risk to wild bees in their comparison of alternative pest management programs. The increasing availability of wild bee-specific data are expected to greatly aid in studies of pesticide risk to bees (Tasei 2002) as well as allow for development of tools to help land managers balance the demands for pest management with their interests in bee conservation.

The IPR index is based on LD₅₀ values from topical applications to honey bees, which do not take into account the routes of exposure that may be important for wild bee communities foraging in crop fields. For example, neonicotinoid insecticides have extremely low topical LD₅₀ values, indicating high bee toxicity, but because this class of insecticides is applied at very low rates per hectare and absorbed into plant tissues after foliar applications, the risk of contact exposure to bees is expected to be much lower than with insecticides whose residues remain on the surface of the plant for many days after application. The disparity between toxicity ranking and true risk may help explain the difference between the 2006 results and those in 2004 and 2005. Several grower cooperators adopted the neonicotinoid insecticide imidacloprid during 2005; yet, there was little apparent negative impact on the wild bee community on their farms in 2006 (Fig. 1). There were no applications of systemic insecticides to the roots of blueberry during this study, minimizing the risk of nectar-borne residue uptake by bees during crop bloom.

For *A. carolina*, which is the dominant wild bee pollinating blueberry in Michigan (Tuell et al. 2009), our results support the prediction that insecticides applied outside the bloom period have little effect on fitness of bees that are active only during crop bloom when bee-safe insecticides are used. Such temporal escape of insecticide residues may have important implications for survival of wild bees that are specialists of crop plants with short bloom periods when bee toxic insecticides are avoided by farmers. However, the abundance of two other solitary species that are dominant in the bee community during blueberry bloom, *A. carlini* and *A. vicina*, did decline with increasing IPR, at least in the first year. Although these two species are known to forage on blueberry, their activity period may be less synchronized with blue-

berry bloom and their risk of exposure to insecticides greater than that of *A. carolina*. The annual variation in responses of wild bee communities found here reflects the typical variation in this group of animals reported in previous studies and reviews (Cane and Payne 1993, Williams et al. 2001), and it emphasizes the importance of multi-year studies to detect effects of chronic disturbance factors on wild bee populations.

We expected to see a greater impact of insecticide applications on the social species than on the solitary species, because generations that emerge later in the season would have full exposure to the more toxic insect pest management practices that are initiated after crop bloom has ceased. One of three species supported this expectation in 2004, but the other two did not (Table 2). Different species have different nesting preferences, with some nesting directly in blueberry fields under the crop canopy while others nest in field margins. Species nesting in field margins would be expected to have much less exposure to the insect pest management practices within a field, although pesticide drift may play a role. Further research into the nesting biology of dominant wild bee crop pollinators and the level of insecticide exposure experienced by wild bees in and around crop fields is needed before such potential links between pesticide exposure and preferred nesting location can be explored.

The data presented here suggest that reducing the risk of pest insect control programs to bees will help conserve populations of these beneficial insects that are active during crop bloom. Other factors such as landscape complexity, inter-annual weather variation, noncrop floral density, and nesting availability are also expected to play a role in bee community variation in agricultural landscapes. The magnitude of the response to reducing the toxicity of insect pest control programs is expected to vary because of different nesting requirements and phenologies. Long-term studies of bee populations in farms that implement such changes will provide valuable insight into the magnitude and speed of the change in wild pollinator communities, and the value of the additional pollination expected. To help refine such conservation strategies, it will be important to understand where bees nest in agricultural lands, the level of insecticide exposure to wild bees active in crop fields, and the toxicity of these pesticides to wild bees that are important for supporting pollination of crop plants.

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