Not by wine alone:
environmental impacts, risks and consequences of viticulture

L. Levitan, Environmental Risk Analysis Program, Center for the Environment, Cornell University, Ithaca, New York USA Email: lcl3@cornell.edu

Prepared for the 11th Australian Wine Industry Technical Conference, October 2001
Adelaide, South Australia

Abstract

In competing for a share of the global market, wine quality and sales panache account for a great deal. However, the perception and reality of environmentally benign production practices are of growing importance, especially to:

- An expanding niche market of consumers worldwide who are responsive to environmentally-sensitive eco-labels;
- Agricultural communities, which have become increasingly attuned to balancing creative tensions between maintaining livelihoods, sustaining health, and protecting the natural resources that undergird both;
- Regulators at all scales and levels of government, whose agencies are monitoring the non-target impacts of agricultural inputs and outflows to the waste stream; and
- Growers, who must make production decisions that take all of these issues and constraints into consideration.

This paper addresses key concepts and points to consider in identifying and evaluating environmental impacts of viticulture. It illustrates how to adapt recent work in the area of environmental and pesticide risk indicators to the specific case of Australian viticulture. The paper does not go into technical details about the use or construction of such assessment systems, because—as will be shown—details are particular to the situation. Rather, references are provided to access technical data and information and the focus here is instead on the role of environmental risk assessment in agriculture.

What is risk? When and why are there non-target impacts of agriculture production?

Viticulture—like any other agricultural system—generates non-target impacts. These impacts include all results and consequences of agricultural practices other than the production of grapes. Non-target effects are to be expected, since the fundamental objective of agriculture is to 're-arrange' the natural ecosystem in order to maximize energy surplus for human consumption. Shifting land cover, diverting water, overturning soil in order to produce the fruit of the vine all alter the status and equilibrium of natural resources and biota.. Some of the impacts are benign, some positive, but sometimes such consequences present risks, both within and beyond the agro-ecosystem, to human and non-human biota, and to the sustainability of natural resource flows.

'Risk' is the likelihood of harm or negative consequences. While the calculation of a precise estimate of risk can be complex, the concept is frequently boiled down to the simple equation *risk = hazard x exposure,* which can also be expressed more accurately as *risk = f(hazard, exposure)*. That is, *risk is a function of hazard and exposure*. What this means is that risk is a function of the ability to cause harm(*hazard*) and the chance of being in harm's way (*exposure*). Risk can be reduced by minimizing either hazard and/or exposure.

There are many types of hazards, differing in magnitude and severity, as well as in duration and reversibility. Hazards can range, from a minor, short-term skin irritation to death of an individual or an entire population (because, for example, habitat destruction is a hazard). Hazards can be minimized by selecting the most benign products and practices—those with potential to cause only the least severe types of harm, or having harmful effects that can be mitigated or reversed.

Exposure to hazards can also be reduced by a variety of management techniques, such as—drawing from the example of pesticides—use of protective clothing and equipment to minimize exposure of pesticide applicators to hazardous chemicals; setting spatial buffers between vineyards and homes to reduce residents' chance of exposure to pesticide spray drift; or choosing low dosage pesticides that minimize runoff and deposition on non-target biota or places.

Conceptually, reducing hazard and/or exposure is how one approaches risk reduction. However, despite the best intentions, we never arrive at a zero risk solution—because there are none. Reducing risks to one group or ecological endpoint typically introduces other risks. For example:

- frequent tillage to destroy weeds increases risk of erosion, while use of herbicides increases risk of chemical side effects;
- one class of pesticides creates toxic risk for fish while another chemical class of pesticides is riskier for mammals; and
- producers may face additional financial risk in applying management options that decrease environmental risks.

In acknowledging the lack of zero risk solutions it is tempting to throw up one's hands and say "Why try?" However, what we must do instead—as a society and as an industry—is to figure out how to prioritize among problems, work towards multi-attribute optimization, and recognize that solutions will draw from several dimensions of our resources and experience, involving management choices on the farm, policy decisions in local planning boards and regional commissions, and corporate research and development priorities. The remainder of this paper focuses on Environmental Risk Indicators as tools to inform the movement towards environmental risk reduction in viticulture.

Environmental risk indicators: auditing and informing Environmental Management Systems (EMS)

Australian viticulturists comprise an industry generally perceived to be progressive in its approach to production and marketing, and cognizant of the environmental impacts of horticulture. [1] Such impacts include both those affecting the agroecosystem—and ultimately affecting vine quality and yields—as well as those affecting nearby populations and communities sharing the river basin and its resources. Indeed, it would be difficult to read the daily newspapers in Australia and be unaware of the tremendous concerns about salinization of ground and surface waters, water over-consumption, and erosion (*Australian Dryland Salinity Assessment 2000,* 2001; *Australia Water Resources Assessment 2000,* 2001).

Growing awareness about these environmental risk factors, and growing pressures to deal with them, has bred a commitment to land stewardship that is being translated for commodity groups in Australia and elsewhere into Environmental Management Systems (EMS) (Boller 1990, 1992; Carruthers and Murray 1999; Vasconcelos undated). These EMS are typically comprised of a set of designated 'best

---

[1] Winemaker flies the flag for clean 'n' green. *The Australian*. Feb 21, 2001. p.3. Amanda Hodge interview with Tony Sharley, Barossa Estates.

management practices' (BMPs) for all components of the production system, practices deemed to work towards solving or minimizing perceived problems. (See also Table 6 in Levitan 2001.)

While in some cases, a short-cycle feedback loop between carrying out the BMP and reaping its benefits makes evident the positive incentives to comply with the EMS, in other cases the feedback loop is longer or cycles benefits to beneficiaries other than the commodity producer. Thus there is need for compelling arguments for linking benefits for the common good with agricultural BMPs, the cost burden for which generally falls on industry shoulders.

One means for developing this argument is to substantiate the social, economic, environmental and/or agro-ecological benefits reaped by following BMPs. This can be done by auditing or assessing components of the EMS in order to clarify environmental health benefits and risks of a set of recommended practices as compared with other management options. Several methodologies are used to evaluate environmental impacts, including *field studies* involving sampling and surveillance of biophysical indicators; *laboratory simulations* in microcosms or mesocosms that simplify and/or mimic field conditions; mathematical—generally computerized—*simulations* based on quantitative and/or symbolic models, often designed to project from current conditions into the future; *qualitative evaluations* based on observation or surveys; and *indicators*. The latter is an approach that draws and synthesizes from information gathered by other methods in order to valuate, track and compare impacts across time or among options. As a tool for auditing and informing EMS, they are commonly called Environmental Risk Indicators (ERI). Those Indicators that focus most specifically on comparing risks of different pest control options are typically called Pesticide Risk Indicators.[2]

The relationship between EMS/BMPs and Environmental Risk Indicators (ERIs) can perhaps be metaphorically likened to the distinction between 'front of the house' and 'back of the house' operations in a winery, where the activity in the wine tasting room is at the 'front of the house'—relating and communicating with the public who respond to the décor, the pizazz, and the gestalt that consumers associate with a specific winery. The vineyard, the laboratory, the fermentation vats, etc. are at the 'back of the house'—comprising the behind-the-scenes hard work and technical details involved in growing grapes and making wine. Similarly, at the 'front of the house' of environmental assessment one might find the checklist of best management practices comprising the EMS, while at the 'back of the house' are the technical data, assumptions and calculations for estimating, auditing, measuring and validating the environmental impacts of the operations. In the same way that the typical consumer holds memories and expectations based on their front of the house experience, the goals of an EMS may be easier to aspire and relate to than the niggling details of a risk indicator. However, in the same way that no building is stronger than its foundation, the underlying assumptions, data, and construction of risk indicators and other auditing systems are critical in developing robust and valid recommendations for risk reduction and other aspects of the operation.

In its simplest form, a Pesticide or Environmental Risk Indicator compares data from different inputs—*e.g.* the different pesticides that could be used to control the same pest—in terms of their impact on the same measurement endpoint, *e.g.* lethal toxicity to fish. Results can be *categorized*, *e.g.* the pesticides labeled as having 'high,' 'medium,' or 'low' toxicity to fish, or *ranked*, *e.g.* from the most toxic to least toxic to fish. The indicator is a pesticide *hazard* indicator if it is based solely upon the way that fish respond to the same dose of each pesticide tested. It is a pesticide *risk* indicator if it also incorporates an estimate of exposure, *e.g.* by comparing the effects on fish of a recommended dosage for each of the pesticide tested. Or by comparing effects of the amount of pesticide that might be consumed by the fish.[3]

[2] However, the nomenclature is not consistent. See Levitan (2000) for history and discussion of this nomenclature and Levitan (2001),Table 3, for a listing of synonyms and near-synonyms for Environmental Risk Indicators.
[3] Even in this simplified indicator of toxic risk to fish, there are numerous 'behind the scenes' assumptions, that could affect results. For example: Is effect on the fish measured in a field or laboratory situation? Is effect measured in a container of still water, or in water moving in a way that simulates a river current (since this affects quantity of

It is important to recognize that the risk indicator described here evaluates only the relative toxicity of pesticides to fish; not other risks of the pesticide, *e.g*. to birds, to people, to the likelihood that the pesticides will leach into waterways or drift onto adjacent land. The critical point is that while Environmental Risk Indicators can be constructed to gauge any set of costs, benefits, impacts or risks on the farm and/or to the broader society and environment, an indicator can only generate information based on the input data. Thus an indicator of fish toxicity cannot be used to base claims about relative impacts of pesticides on beneficial organisms or on people, nor should such a single endpoint indicator be the sole basis for recommending choice of one pest control product over another.

Indicators of 'clean' vs. indicators of 'green'

Australian viticulture is seeking indicators to substantiate and validate its reputation as 'clean and green.' The term 'clean' has come to refer to agricultural products free of pesticide residue, an assessment endpoint that is relatively straightforward to measure. In Victoria, for example, where more than 25% of Australian horticulture is produced, the Chemical Standards Branch of the DNRE uses a targeted approach, performing laboratory tests to monitor for those chemicals most likely to be found because they are the ones most frequently used. Nevertheless, in 1999/2000 they found MRLs (maximum permitted residue levels) exceeded in just 1% of samples (Victorian Produce 2001). In general, Australian horticulture ranks well on indicators of 'clean.'

Pesticide residues on finished products are typically a function of (a) the length of time between application and consumption and (b) pesticide properties. On grapes as with other horticultural products, residues are affected by spray concentrations, number of applications, and climate, as well as length of the pre-harvest interval. Typically longer pre-harvest intervals are required for pesticides that degrade slowly and are, therefore, more likely to leave residues. So the regulation of pre-harvest intervals is a straightforward means for managing pesticide risks to consumers. In summarizing their study of residues from 31 insecticides and fungicides most widely used on grapes in Italian vineyards, Cabras et al. (1995) conclude that "taking into account all the factors affecting the decrease of residues—degradation, fruit growth [during the period between application and harvest], fermentation, vinification technology—the probability of finding detectable residues in wine is quite modest."[4] In an earlier review, Cabras et al. (1987) evaluated the fate of fungicides used in vineyards at every stage leading to the finished wine, and noted that any residues remaining are considerably reduced during wine making by use of clarifying substances.[5] Thus 'clean' does not appear to be the challenge in monitoring and reducing environmental health effects of viticulture.

Both conceptually and functionally it is far more challenging to identify the key components of 'green' production, to follow 'green practices,' and to validate risk reductions resulting from this presumably more benign agriculture. It may not be clear and is certainly not uniformly equal which environmental endpoints are most critical in generating risk and/or most sensitive to viticultural practices. In New York State (NYS), for example, the use of fungicides on grapes presents a significant environmental risk to groundwater in a region where sandy soils overlay a shallow aquifer, but is a less significant water quality

---

pesticide ingested by the fish)? Are the tests on young or adult fish? Of what species? At what density of fish in the test water body? Should acute lethality (i.e. quick death) be the measured effect? Or are changes in reproductive behavior of similar importance? etc.

4 Cabras et al. (1995) found low concentrations [their categorization] of residues of five of the 31 most widely used fungicide and insecticide active ingredients applied to grapes in vineyards in Italy: benomyl (0.05 ppm) in 1 of 4 samples, dimethoate (0.02-0.06 ppm) in 3 of 5 samples, iprodione (0.02-0.07 ppm) in 8 of 10 samples, metalaxyl (0.04-0.14ppm) in all samples, and vinclozolin (0.02) in 1 of 20 samples. Number of samples refers to the number of grape samples tested from vineyards known to have received applications of the named pesticide.

5 They do note, however, that some of the fungicides (dichlofluanid, phthalimides, triadimefon) may delay fermentation or block the final fermentation stage (Cabras et al. 1987).

issue in another NYS wine region with loamy glacial till. In most of the rainy, humid eastern United States, water consumption for agriculture does not present the same environmental threat that it does in Australia, where one-quarter to one-third of water management areas are at or beyond sustainable extraction limits and 65% more water is used now than 20 years ago (*Australia Water Resources Assessment 2000*, 2001). Conversely, the humid growing seasons in the eastern US present greater obstacles to production of grapes and other soft fruits without fungicides than is the case in the drier western wine regions of the US.

The next sections describe steps that can be taken to identify the key components of 'green agriculture' in a particular environment and to assess progress towards reaching goals in greening of production practices.

The 'criteria and indicator matrix'

A 'criteria and indicator matrix' is a framework for organizing relevant information for use in an agricultural assessment or auditing system (Levitan in preparation). It is a useful means for getting an overview of environmental health and risk factors pertinent to a given agricultural system, as well as an overview of both hazard and exposure indicators that could be used to measure non-target effects of the system. The rows of the matrix list the environmental health and risk issues (*e.g.* pesticide impacts on beneficial organisms, pesticide impacts on neurological function in people, etc.). Data in the cells articulate the specific measures that could be taken to make such an assessment and the criteria for categorizing impacts of different pesticides and other management variables on the given indicators, as these have been determined by subject-area experts referenced in the last columns (Table 1).

There are often impacts that one would like to evaluate in a system, but for which there is no reasonable measurement endpoint, or for which data have not been collected. For example, while a healthy soil microflora is a critical component of the agro-ecosystem, it is unclear what should be measured to assess the impact of pesticides on microflora—*e.g.,* which is the most relevant indicator of impact: diversity of microflora species, mass of a single species, total mass of microflora, respiration rate, etc..

The underpinning of a risk indicator is having data collected under a similar set of circumstances for the full array of possible management options (*e.g.* for all pesticides which could be used in a given situation). Thus, although pesticide interactive effects may be of considerable importance in evaluating the relative risk of different pesticides, the number of potential mixtures is infinite and it is unlikely there will ever be a sufficient dataset of such impacts. The criteria and indicator matrix provides a transparent means for evaluating feasibility and data availability as well as:

(a) which factors (**types of impacts** and risks) to take into consideration in assessing a particular agricultural management system (*e.g.* pesticide acute and carcinogenic effects on people, pesticide impacts on pollinators, water consumption impacts on salinization, tillage impacts on erosion, etc.;

(b) which **assessment endpoints** to use to register impacts on the selected variables;

(c) how the impacts can be measured (**measurement endpoints**); and

(d) the **criteria** or thresholds for assigning a measurement endpoint to a given category of impact (positive, neutral, or negative).

(e) The matrix can also include the **source of information** for setting categorical limits; and

(f) the **source and reliability of supporting datasets** (*e.g.* dose-response relationships for a set of pesticides).

The most relevant factors (types of impacts) very much depends on where system boundaries are set for the assessment—*i.e.* around the crop, the farm, the commodity, the rural community, the river basin, the nation, or the earth. While there is no one correct or morally superior approach to setting system boundaries, it may be the case that benefits and risks will fall out differently in looking at systems defined by each of these parameters (Levitan 1997).

Technical inputs to the matrix are drawn from all corners of the scientific literature and from expert judgments, but decisions about which of the factors (which impacts) are most critical for the system under review should be within the purview of those most affected. Stakeholders should decide on both the system boundaries and on the set of issues to examine. Technical expertise is needed to identify the most meaningful (*i.e*. the most sensitive or leveraging) indicators of impact on that system (*i.e*. the assessment endpoints), and the most appropriate measures of impact (*i.e*. the measurement endpoints).

Ideally, classification criteria reflect environmentally- or biologically-relevant thresholds, but often they do not because such information is not known. Instead arbitrary thresholds are set at obvious arithmetic breakpoints (*e.g*. $LD_{50} < 10$ = very hazardous, < 100 = moderately hazardous, < 1000 = slightly hazardous, > 1000 = not hazardous). Thus the 'criteria and indicator matrix' performs an additional valuable function in making transparent the underlying assumptions and data upon which prescriptive and descriptive judgments are made about impact and risks of different management options.

Typology of environmental risk indicators

Different sets of assessment endpoints are pertinent to ERIs that are intended for different purposes and target audiences. Some ERIs are *descriptive*—intended to evaluate and/or communicate about impacts that already occurred. For example, environmental protection agencies want to know about trends in pesticide use and risk. These and other policy research tools are typically *descriptive*[6], after-the-fact environmental risk descriptors. Eco-labels—which are essentially indicators of approval of some aspect of production, intended to influence consumer opinion and behavior in the marketplace—are also *descriptive*. Conversely, indicators that are intended to inform and influence future farm management and policy decisions are called *prescriptive*.

To relate this distinction between *descriptive* and *prescriptive* indicators more specifically to viticulture and environmental management systems, one can think about *prescriptive* indicators as providing the 'back of the house' technical information and guidance used to develop an EMS, specify a BMP and inform particular management choices. A prescriptive indicator could show which pest control option has least potential for negative human health effects or is most benign to the biological control organisms active in the vineyard. Descriptive indicators, on the other hand, are often used by the industry and individual production units as 'front of the house' communication tools to evaluate or substantiate claims of 'clean and green.'

Descriptive assessments have the luxury of selectively evaluating whatever factors in a system are of interest to whomever is asking the question. They need not describe everything about the system, nor be holistic. They can be as generalized or as specific as needed to serve their purpose—*e.g.,* as large scale as a trend analysis of water consumption on each of the earth's continents or as localized as a comparison of pesticide impacts on population of a fish species in a particular catchment.

Similarly, eco-labels are not specific to any one set of environmental standards or any one certification program. The term is generically applied to this market mechanism. Indeed, eco-labels are the 'front end' for many different assessment criteria and auditing systems. Eco-labels are a key market mechanism to communicate environmental standards to the public. If successful, they provide feedback and incentives to producers to adopt more environmentally benign and sustainable production practices.[7] The criteria

---

[6] Economists call descriptive analysis *ex poste* and prescriptive analysis, *ex ante*.

[7] In describing findings of the recent report *Repairing the Country* (2001), the chair of Southcorp said: "farmers need a commercial incentive to adopt more sustainable practices."

used to designate an eco-label are idiosyncratic and frequently based upon social or geographic criteria (*e.g.* such as whether the farmworkers were adequately paid or whether a product was locally produced), rather than on criteria such as toxicity of agro-chemical inputs, impact of irrigation on salinization, etc.

Because of the uses to which policy and consumer tools are put, their assessment results always stand separate from the decision process. That is, the eco-labeled product stands on a shelf in the supermarket, the results of a trend analysis are printed in a book—so the results can not change with particular mitigating circumstances. The better prescriptive indicators, however, depend upon input from the production manager or other decision-maker, such that the decision-process is intertwined with output results. This is one of the keys factors differentiating policy tools and other descriptive indicators from prescriptive indicators used to guide pragmatic production practices. Other typological characteristics of environmental and pesticide risk indicators constructed and used for different purposes are more fully described in previous publications (Levitan 1997, 2000).[8] Here we will focus on why it is important to build interactive functionality into farm-scale decision tools and how to construct such indicators.

Constraints, concerns and difficulties to consider[9]

Exposure Factors

Certain inputs to an indicator can be drawn from the generic scientific literature. Hazard endpoints, for example, are for the most part consistent across all venues[10], while—in contrast—exposure endpoints are affected by situation-specific management, climate and site variables. For this reason, a prescriptive ERI that is intended to guide (or audit) farm management practices should be constructed as an interactive decision-tree, rather than as a simple risk ranking based upon an immutable algebraic equation or algorithm (Levitan 1997, 2000).

To illustrate: the widely used viticultural herbicide trifluralin is hazardous to fish (AWRI 2000; Meister 1999). If a pesticide risk indicator were to base its risk ranking of trifluralin on hazard data drawn from standard toxicological sources using application dosage as a proxy for exposure, the result would be misleading because trifluralin poses a risk to fish only if it runs off or drifts into surface waters containing fish. A more realistic risk estimate might consider a combination of dosage and distance to surface water as proxies for exposure to fish.

Pesticide fate and transport properties are often used as exposure endpoints. However, they also are somewhat situation specific. For example, in Australia, trifluralin has been found to be phytotoxic to certain crops planted the year following application, more so than would be inferred from North American studies showing a reference half life of 60 days and less than 10% residues remaining into the following season (Hornsby et al. 1996; Jolley and Johnstone 1994; Wauchope et al. 1992). However, perhaps because the climate conditions in North America are different from those in Australia, the results of similar studies in Victoria showed half-life exceeding 180 days, and with as much as one quarter of the trifluralin remaining in the field the following year (Jolley and Johnstone 1994). Several points jump out from these findings, among them that:

(1) physico-chemical 'properties' of agrochemicals are not standard across soils and climates (Kookana et al. 1998). While this may be obvious to those with hands-on field experience, such insights also must be incorporated into environmental management audits and indicators;

---

8 A number of environmental risk indicators are described in *Integrated Pest Management Measurement Systems Workshop* (1998), Levitan et al. (1995), Levitan (1997) and Reus et al. (1999). A listing of ERI constructed for different purposes is maintained at www.cfe.cornell.edu/ERAP/PRI (Levitan 2001).

9 See also Riha et al. (1997) and Levitan (1997).

10 For simplicity sake we are ignoring here the variability among sub-populations that results in different susceptibilities to hazards at different life stages and among individuals with different genetic make-up. However, there will be a considerable rethinking of environmental health effects in the nascent era of genomics.

(2) trifluralin phytotoxicity is not an important risk factor unless sensitive crops are to be grown in the same soil within the residual period. Phytotoxicity should not, therefore, be part of a tally of trifluralin risks unless this factor is relevant for a particular crop rotation. Such knowledge about the crop rotation or crops in neighboring areas can only come from the property manager via an interactive 'expert system' or decision-tree type of assessment;

(3) first half-life may not be the best measurement endpoint or indicator of persistence, especially when degradation does not follow first order kinetics (Jolley and Johnstone 1994). To extrapolate this concept more broadly—it is critical, albeit sometimes quite difficult, to use measurement endpoints that accurately reflect on the specified risk parameter or assessment endpoint; and

(4) many differences can be noted between Australian pesticide persistence data and standard US references (Wauchope et al. 1992; Jolley and Johnstone 1994; Hornsby et al. 1996). These could reflect real differences due to soils and climate across the world, or could reflect the fact that any point estimate of behavioral parameters such as persistence and soil sorption is likely to be off the mark under many conditions found within either vast continent. Risk indicators must be able to incorporate such variability.

Recent work by Ying and Williams (2000) on dissipation of herbicides from soil and grapes in the Barossa Valley of South Australia substantiate this point. [11] They found that ordered rankings of four herbicides by dissipation rate were different under different soil and climate conditions, and different on grapes than in the soil.

Hazards

In selecting assessment endpoints for an environmental risk indicator, the impacts of pesticides and other factors of production can not be measured for all groups of biota. In fact, with few exceptions, it is typical (though hardly ideal) to only assess acute impacts on people and perhaps on fish, and to use carcinogenicity potential as the proxy for all other long-term effects of pesticides. As will be discussed in more depth below, this parsimony is a result of data gaps and the difficulties in handling multi-attribute optimization. However, the reality is that different groups of biota and different assessment endpoints for the same groups of biota are sensitive to the toxic effects of different pesticides. The fact that no one group of biota can stand in as an effective proxy for all in hazard rankings of pesticides is well illustrated in the following list of endpoints susceptible to the impacts of viticultural pesticides used in the United States (CATS 1997) (Table 2). It is further demonstrated in Table 3, which compares the pesticide hazard rankings produced by three risk indicators, each of which is based on slightly different assessment endpoints and method for calculating hazard (*i.e.* the indicators have different structures and differently weight the variables).

Table 2. Disparate hazards of viticultural pesticides used in the United States (CATS 1997)

Hazard Endpoints	Chemical
Human cancers	metam sodium, mancozeb, dimethoate, benomyl, propargite,

[11] Ying and Williams (2000) found that the herbicides norflurazon (a pyridazinone), oxadiazon (an oxadiazolone), trifluralin (a dinitroanaline) and oxyfluorfen (a diphenyl ether) showed rapid initial loss in sandy soils followed by slow degradation (especially oxyfluoren, with a 119-day first half-life). Herbicides applied in one year were detected in the soil the following spring. Degradation rates differed greatly with soil and climate conditions, to the extent that the order of ranking the pesticides in terms of dissipation rate could change under different sets of environmental conditions. Relative ranking of the four pesticides were also different on soil than on grapes, where trifluralin and oxyfluorfen were not detected after 4 days and norflurazon and oxadiazon residues were found a month later. Wind and sunlight play a role in dissipation rates even in absence of rain, acting on variables like texture of the grape surface.

	oxyfluorien, oryzalin and diclobenil
Human birth defects:	metam sodium, mancozeb, benomyl and oxyfluorfen
Reproductive problems:	dimethoate and myclobutanil
Human nerve disorders:	methyl bromide, inancozeb, dimethoate and chloropicrin
Air contamination:	methyl bromide, metam sodium, mancozeb, dimethoate, benomyl, glyphosate, propargite, oxyfluorfen, oryzalin, chloropicrin, carbaryl and diclobenil
Water pollution	simazine, oryzalin, oxyfluorfen, dimethoate, benomyl and glyphosate
Fish	simazine, mancozeb, dimethoate, benomyl, glyphosate, propargite, oxyfluorfen and oryzalin
Aquatic invertebrates or plants:	dimethoate, oryzalin, oxyfluorfen, propargite and glyphosate
Birds:	oryzalin, dimethoate, benomyl
Bees:	dimethoate
Mycorrhizal fungi	methyl bromide and glyphosate

Difficulties in multi-attribute optimization
Given these problems with using one species (or endpoint) as a proxy for all others in assessing environmental impacts, another option would seem to be the consideration of multiple endpoints in the indicator. The difficulties with this approach lie in how to integrate results, either into a composite picture of environmental impacts or by prioritizing the most critical impacts in a given situation. To grasp the conceptual challenge, think about how you might jointly weigh serious impacts on people with impacts on non-human biota.

Data gaps
Pesticide risk indicators in particular (as compared with the broader field of environmental risk indicators) depend upon comparable data for the full array of pesticide options. Without this, comparisons cannot be made unless one of several means is used to fill data gaps, each of which introduces a different bias into the dataset. Complete datasets are rare: of 150 commonly used pesticides evaluated by Pease et al. (1996), toxicity data for invertebrates were available for just 44 of the pesticides, and field half-life data for only 106. Moreover, some of the existing ecological effects data are inappropriate to use for assessing relative impacts because standardized testing protocols were not used and the data are not comparable (Levitan et al. 1995). Often the best sorts of data on non-target impacts to non-human biota are from limited datasets developed for specific crops and defined applications. Extrapolating from these parameters to other crops or organisms is difficult because of lack of overlap in pesticides used on different crops. Drawing from a recent Australian example, Williams and Nicholas (2001) tested impacts of pesticides used in Australian apple orchards on the European earwig, a beneficial predator in the orchards that can consume scores of woolly aphids each night. The pesticides were ranked as having high, medium and low toxicity to the earwig. All of the fungicides tested were shown to have low toxicity to the earwig adults, which is important information to have in developing and auditing BMPs. However, only one of the fungicides considered important in Australian viticulture was tested in this study and impacts on immature stages were not assessed.

Interactive effects
In addition to possible synergies or antagonisms in the toxicity of mixtures of pesticides, interactive ecological effects add a complexity that is difficult to accommodate in simple indicators of pesticide risks, especially if they are based on a one-time application rather than on impacts over an entire production season, or longer. For example, recent work by Reding et al. (2001) has shown an association between powdery mildew and spider mite populations in apple and cherry orchards, with higher densities and enhanced reproduction of spider mites on leaves with mildew. The apparent conclusion from this is that effective management with fungicides could reduce the need for acaricides. Also, that limiting damage from insects and mites may have a longer term impact in reducing severity and incidence of fungal damage because there would be fewer wound-sites for pathogen entry.

Finding a common currency to measure and describe different types of impacts
Assessment indexes are well-suited for comparing relative impacts of a similar set of options (such as comparing toxic impacts of pesticide options, all of which can be expressed in terms of a pesticide dose-toxicity response relationship), but may be less successful in comparing impacts that do not share a common currency of accounting units. For example, we do not have common units to compare toxic effects of pesticides produced regionally with energy costs of transporting organic produce from a distant agricultural region. We are accustomed to using money as a common currency for trade, but money is inadequate for describing non-market costs such as the loss of an individual life, loss of biodiversity, future costs of current soil erosion, or loss of non-replaceable resources. Although methods such as contingent valuation, apportioning remediation costs, and using travel or avoidance costs have been developed for assigning a monetary value to non-market goods, they generally fail to capture the full dimensions of the richness and value of non-market attributes.

Conclusion

Decisions about pesticide use and other production inputs have traditionally been based on cost and efficacy. However, a disparate array of individuals and institutions–including farmers and other land managers, consumers and consumer groups, food retailers and agribusinesses, regulatory agencies and regulatory 'watchdogs'–also have a stake in knowing about *non-target impacts* of agricultural management practices. Thus–and especially since the mid-1990s–a number of 'environmental risk indicators' have been developed to measure, evaluate and rank practices and inputs in terms of their non-target impacts and risks. Some of these indicators consider only pesticide risks; others put pesticides into a comparative context with other environmental stressors (such as habitat destruction and the consequent impacts of erosion, sedimentation, and loss of biodiversity); some include economic as well as environmental considerations. Some consider only one type of risk (*e.g.* acute effects on people), while others evaluate various possible impacts on multiple species (*e.g.* both short and long-term human health impacts as well as ecosystem effects).

The challenge is to develop and apply scientifically robust risk indicators that consider both hazard and exposure factors and provide useful guidance and/or feedback to the intended target audience. Risk indicators must be designed—in terms of the choice of endpoints, method of calculation, and format—appropriately for their intended purpose. In order for indicators to be useful and accurate in informing and/or auditing farm environmental management systems, they must be sufficiently flexible to incorporate situation-specific inputs and prioritization of criteria. The imperative it that although agricultural systems have always produced non-target effects, the natural resource constraints now faced worldwide are more pressing because of the magnitude of demand and the damages already wrought.

References

AWRI (May 26, 2000 Update) *Agrochemicals registered for use in Australian viticulture 2000/2001*. The Australian Wine Research Institute, Adelaide, South Australia.

Australian Dryland Salinity Assessment 2000. (2001). National Land and Water Resources Audit. The Australian National Resources Atlas. National Heritage Trust. Canberra. (http://audit.ea.gov.au/ANRA/land/docs/national/Salinity_Contents.html)

Australia Water Resources Assessment 2000. (2001) National Land and Water Resources Audit. The Australian National Resources Atlas. (http://www.nlwra.gov.au/) National Heritage Trust. Canberra.

Boller, E.F. (1990) The ecosystem approach to plan and implement integrated plant protection in viticulture of eastern Switzerland. In: *Plant-Protection Problems and Prospects of Integrated Control in Viticulture*. Proceedings International CEC-IOBC Symposium, Lisboa, Portugal, 1988. Cavalloro, R. (Ed.) Report EUR 11548, pp 607-617.

Boller, E.F. (1992) The role of integrated pest management in integrated production of viticulture in Europe. *Brighton Crop Protection Conference—Pests and Diseases*. 6B-1, 449-506.

Cabras, P.; Meloni, M.; Pirisi, F.M. (1987) Pesticide [Fungicide] fate from vine to wine. *Reviews of Environmental Contamination and Toxicology* 99, 84-117.

Cabras, P.; Garau, V.L.; Melis, M.; Pirisi, F.M.; Spanedda, L. (1995) Pesticide residues in Italian wines. *Ital. J. Food Sci*. 2, 133-145.

Carruthers, G.; Murray, S. (1999) Environmental management systems and agriculture: How can they be applied and what are the benefits? *1999 Production and Environmental Monitoring Workshop*. Paper No PEM02. 17 pp.

CATs (1997) *Time for a Change: Pesticides and Wine Grapes in Sonoma and Napa Counties, California*. Californians for Alternatives to Toxics (CATs). 44 pages. (accessed September 2001 at http://www.alternatives2toxics.org/contents.htm)

Hornsby, A.G.; Wauchope, R.D.; Herner, A.E. (1996) *Pesticide Properties in the Environment*. Springer. 226 pp.

*Integrated Pest Management Measurement Systems Workshop, June 12-13 1998, Chicago, Illinois. Working Papers (*1998) American Farmland Trust, Center for Agriculture in the Environment.

Jolley, A.V.; Johnstone, P.K. (1994) Degradation of trifluralin in three Victorian soils under field and laboratory conditions. *Austr. J. Exp. Agric*. 34, 57-65.

Kookana, R.S.; Baskaran, S.; Naidu, R. (1998) Pesticide fate and behavior in Australian soils in relation to contamination and management of soil and water: a review. *Aust. J. Soil Res*. 36, 715-64.

Kovach, J; Petzoldt, C.; Degni, J.; Tette, J. (1992) *A Method to Measure the Environmental Impact of Pesticides*, New York's Food and Life Sciences Bulletin No. 139, Cornell University, Ithaca, NY, USA, 8 pp.

Levitan, L; Merwin, I.; Kovach, J. (1995) Assessing the relative environmental impacts of agricultural pesticides: the quest for a holistic method. *Agric., Ecosys. Environ.* 55, 153-168.

Levitan, L. (1997) *An overview of pesticide impact assessment systems (a.k.a. 'pesticide risk indicators') based on indexing or ranking pesticides by environmental impact*. Background paper presented at the Organisation of Economic Cooperation and Development (OECD) Workshop on Pesticide Risk Indicators, 21-23 April 1997, Copenhagen, Denmark. 89pp. (access at http://www.cfe.cornell.edu/risk/pri/LCL-PestRiskInd7-97.pdf)

Levitan, L. (2000) 'How to' and 'why': assessing the enviro-social impacts of pesticides. *Crop Prot*. 19 (8-10), 629-636.

Levitan, L. (accessed 2001) *Pesticide Risk Indicators—Publications and Resources* (http://www.cfe.cornell.edu/risk/pri/index.html). Environmental Risk Analysis Program, Cornell University Center for the Environment.

Levitan, L. (in preparation) Criteria and indicator matrix of environmental impacts of pest control methods.

Meister (1999 and annual editions) Farm Chemicals Handbook '99. Meister Publishing Company: Willoughby, Ohio, USA.

Pease, W.S.; Liebman, J.; Landy, D.; Albright, D. (1996) Pesticide Use in California: Strategies for Reducing Environmental Health Impacts. An Environmental Health Policy Program Report, Center for Occupational and Environmental Health, School of Public Health, California Policy Seminar, University of California, Berkeley, CA, USA. 116 pp.

Reding, M.E.; Alston, D.G.; Thomson, S.V.; Stark, A.V. (2001) Association of powdery mildew and spider mite populations in apply and cherry orchards. *Agric., Ecosys. Environ.* 84, 177-186.

Repairing the Country—Leveraging Private Investment. (2001) Prepared by the Allen Consulting Group for Business Leaders Roundtable, Australia.

Reus, J.; Leendertse, P.; Bockstaller, C.; Fomsgaard, I.; Gutsche, V.; Lewis, K.; Nilsson, C.; Pussemier, L.; Trevisan, M.; van der Werf, H.; Alfarroba, F.; Blumel, S.; Isart, J.; McGrath, D.; Seppala, T. (1999) *Comparing environmental risk indicators for pesticides: Results of the European CAPER project.* CLM 426-1999. Center for Agriculture and Environment. Utrecht, NL. 184pp. http://www.clm.nl.

Riha, S.; Levitan, L.; Hutson, J. (1997) Environmental impact assessment: the quest for a holistic picture. In: *Proceedings of the Third National IPM Symposium/Workshop: Broadening Support for 21st Century IPM.* Lynch, S.; Greene, C.; Kramer-LeBlanc, C. (Eds.) USDA Economic Research Service Report No. 1542. pp. 40-58.

Swanson, M.B.; Davis, G.A.; Kincaid, L.E.; Schultz, T.W.; Bartmess, J.E.; Jones, S.L.; George, E.L. (1997). A Screening Method for Ranking and Scoring Chemicals by Potential Human Health and Environmental Impacts. *Environ. Tox. and Chem.* **16**, 372-383.

Vasconcelos, M.C. (undated) Development and present status of integrated viticulture production in Europe. Oregon State University Department of Horticulture. 4 pages.

Victorian Produce Monitoring Results of Residue Testing 1999/2000. Jan 2001. ISSN 1039 3846. Compiled by M. Ruth McGowan, Victoria DNRE.

Ying, G.G. and Williams, B.; (2000) Dissipation of herbicides in soil and grapes in a South Australian vineyard. *Agric., Ecosys. Environ.* 78, 283-289.

Wauchope, R.D.; Buttler, T.M.;. Hornsby, A.G.;. Augustijn-Beckers, P.W.M.; Burt, J.P. (1992) The SCS/ARS/CES pesticide properties database for environmental decision-making, *Reviews of Environmental Contamination and Toxicology* 123, 1-164.

Williams, D. and Nicholas, A. (2001) Residues, toxicity and earwigs. *Pome Fruit Australia.* (Feb) 17.

Table 3. Comparison of 'most hazardous' pesticides as ranked by three assessment systems. Only 2,4-D, trifluralin and dimethoate are on more than one list. These rankings are not weighted by pesticide release, usage, environmental concentration, exposure or typical dosage and thus should not be interpreted as presenting the greatest danger or risk (from Levitan 1997).

Pesticides from List of Top 30 Chemicals, ranked by Swanson et al. Screening System, Not Weighted by Usage (1997)		Highest-Hazard Pesticides, ranked by the UC Environmental Health Policy Program (Pease et al. 1996)		Highest Environmental Impact Quotient (EIQ) values for pesticides (Kovach et al. 1992)	
Rank	**Pesticide**	**Rank**	**Pesticide**	**Rank**	**Pesticide**
1	terbufos	1	methomyl	1	disulfoton
2	**trifluralin**	2	aldicarb	2	parathion
3	hexachlorobenzene	3	carbofuran	3	propoxur
4	anthracene	4	**2, 4-D (+ salts)**	4	oxydemeton-methyl
5	chlorothalonil	5	mevinphos	5	fenamiphos
6	**2, 4-D**	6	**dimethoate**	6	**dimethoate**
7	1,3-dichloropropene	7	**trifluralin**	7	paraquat