How Do We Know an Agricultural System is Sustainable?

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December 20, 2013

Executive Summary

Several conservation organizations and scientists are promoting "sustainable intensification" as a strategy for simultaneously feeding the expanding human population and minimizing the need to clear more land for agriculture. While it is clear that higher crop yields per hectare can reduce pressure on land, there is no guarantee these yield gains will come without tradeoffs that degrade the environment in other dimensions. For this reason, there is a need for sustainability metrics that reflect the outcomes of agriculture as opposed to the practices of agriculture. Ideally sustainability metrics for agriculture should rely on data that exist at multiple scales, and that can be remotely sensed as much as possible. By focusing on outcomes, one can escape the often ideological arguments about GMO versus non-GMO, organic versus non-organic, and so forth. Most importantly, by focusing on outcomes, conservation interventions in the name of "sustainable intensification" can be evaluated.

In this paper a measures framework is developed that can be used from the global scale all the way down to the plot level. At the global or national scale, data can identify areas of concern or opportunity (such as arable land that is not converted yet, or areas where agriculture is driving water scarcity), even though those data are coarse and infrequently updated. At the landscape (or watershed) scale it becomes possible to directly measure the impact of agriculture on the environment, and to compare outcomes with data on practices aggregated from the plot scale. Finally, at the plot level, one can engage with farmers, and validate measurements taken at the landscape scale.

The need to align with existing metrics being promoted by other organizations, as well as the need for practical assessments of conservation interventions yields four dimensions of agricultural sustainability:

1. **Soil health** measured in terms of soil erosion and soil organic carbon
2. **Fresh water** consumption and quality.
3. **Landscape degradation** measured by habitat conversion, fragmentation, and composition.
4. **Biodiversity** measured in terms of species richness and abundance for both birds and amphibians.

The reasoning behind each metric, as well as existing sources of data and potential future sources of data, are discussed in the report that follows. This leads to a concluding section that outlines some next steps for obtaining the needed data, as well as future studies to fill in critical gaps in our understanding of agricultural sustainability

Overview

The combination of population growth and economic development will require that the world produce 70% more food by 2050 (FAO 2011). Several conservation organizations and scientists are promoting sustainable intensification as a strategy for simultaneously feeding the expanding human population and minimizing the need to clear more land for agriculture. While it is clear that higher crop yields per hectare can reduce pressure on land, it is less obvious that those yields can be achieved without secondary effects that run counter to the principles of sustainability. For example, improving irrigation efficiency often *increases* the amount of water *consumed* by agriculture (see the water consumption section below for details, Contor and Taylor 2013, Samani *et al.* 2012, Ward and Pulido-Velazquez, 2008, Ward 2014).

Even more problematic are debates about whether or not GMO crops should be called sustainable (Carpenter 2010, Conner *et al.* 2003, Gilbert 2013, Wolfenbarger and Phifer 2000), or whether highly concentrated cattle feedlots are appropriate. What is needed are some practical metrics of sustainability that allow detection of improvements or reductions in sustainability outcomes as a consequence of different management practices or conservation interventions.

Here we develop a framework and some initial ideas for multi-scale sustainability metrics that could in theory be applied to agricultural systems around the world. Our framework is distinguished by three features. First we argue the framework must operate at more than one scale—ideally from farm, to landscape, to regional or national (or even global). Second we seek a framework that requires a minimum amount of data, and that uses remotely sensed data (e.g. satellite imagery) when possible. In other words, we trade off complexity and accuracy for practicality and scalability. Third we examine sustainability through both ecosystem services and ecosystem impacts, with particular attention to landscape features as well as water and soil. By focusing on ecosystem services we can ask whether these services are stable, increasing, or declining—when services are declining it implies a lack of sustainability. By also including ecosystem impacts, we recognize that loss of ecosystem function (e.g. declining biodiversity) is a problem even if agricultural productivity is not immediately diminished because of the losses. This builds on the framework outlined by Dale and Pulasky (2007), who outlined the need for practical measures at multiple scales of ecosystem services provided to agriculture, generated by agriculture, and impacted by agriculture.

Measurement Framework and Objectives

Definition and Scope

There are numerous efforts to define sustainability in agriculture, as well as to establish monitoring programs to track it. The 1990 Farm Bill in the United States laid out a broad definition that includes: meeting human food and fiber needs, sustaining the economic viability of farms, enhancing the quality of life for farmers and society, enhancing environmental quality, and efficiently using nonrenewable resources (U.S. Congress 1990). A more practical definition for our purposes is provided by Pretty (2008), who calls for the development and use of agricultural technology and practices that “(i) do not have adverse effects on the environment, (ii) are accessible to and effective for farmers, and (iii) lead to both improvements in food productivity and have positive side effects on environmental goods and services.” Pretty also argues that sustainable agricultural systems should be both resilient to shocks and stresses, and capable of persisting over long time periods.

It is well-known that agriculture has been highly successful in increasing production, but that these production gains often comes at the cost of increased water quality problems, fossil fuel use, and conversion of natural habitat (Pretty 2008). This does not mean, however, that increased yields automatically coincide with environmental costs. In particular, several projects in the developing world have demonstrated that it is possible to simultaneously increase yields while reducing environmental impacts. For example, a review of 198 sustainable agriculture projects in the developing world reported a mean relative yield increase of 79%, with 25% of projects reporting a yield increase of 100% or greater (Pretty *et al*. 2006). Similarly, a review of 62 Integrated Pest Management (IPM, a holistic approach focusing on naturally occurring pest control agents, Bottrell 1979) projects found that over 60% of them were able to reduce pesticide use while increasing yields (Pretty 2008). Most of the remaining 40% involved either small yield drops along with large decreases in pesticide use, or small increases in herbicide use to make reduced tillage practical (which offers substantial environmental benefits to compensate for the increase in pesticide). As new techniques develop (e.g. “push-pull” strategies in integrated pest management) it should be possible to further improve upon these results (Cook *et al.* 2007, Khan *et al.* 2011).

We do not explicitly consider sustaining yields, social equity issues, or economic concerns here. Monitoring changes in yields is straightforward, and data on the production and spatial distribution of crops are already available (FAO 2013a, MapSpaM 2010).

Sustainable Agriculture Standards / Metrics

Several organizations have developed approaches for measuring sustainable agriculture (see Table 1 below). We have borrowed from many of these existing efforts to develop our framework, which is distinguished by its emphasis on environmental features critical to conservation, and attention to the need for an approach that applies to different scales (field or farm, landscape, and regional or national). Our emphasis on scale and conservation is driven by the fact that sustainable intensification is proposed as a “conservation strategy.” Given this, and potential concerns about sustainable intensification (Angelsen and Kaimowitz 2001, Vandermeer and Perfecto 2007), it is imperative that we have metrics to verify that it can indeed succeed as a conservation strategy. We also seek metrics that might be remotely sensed; in some cases this is impractical now but will be possible in the near future with improved technology and research. We also identify existing data sets wherever possible to aid in the immediate implementation of measures, although there is a bias towards North American data as the authors have more experience with those data sets.

We focus on four dimensions of sustainability: **freshwater**, **soil health**, **biodiversity**, and **landscape ecology**. These are similar to the meta-metrics called for by the “Pathways report” (Solutions from the Land, 2012).

It is critical to understand that these sustainability variables or dimensions are not independent. Plowing land not only has major implications on soil health, but the resulting soil erosion will also impact water quality and aquatic biodiversity. Landscape fragmentation and habitat conversion can have major impacts on water supply and quality as well as biodiversity (and ultimately on soil health as well).

Table 1: A comparison of sustainability metrics used by different organizations.

	Soil	Water	Landscape Ecology	Biodiversity	Crop Production / Pest Management	Energy / Emissions / Air quality	Waste
This white paper	erosion, carbon	water consumption, water quality	conversion, composition, connectivity	richness & abundance for birds & amphibians			
Field to Market	erosion, carbon	irrigation water use	land use			energy use, greenhouse gas emissions	
Food Alliance	soil & water conservation *(erosion, organic matter, compaction)*	soil & water conservation *(irrigation water use, water quality)*	wildlife habitat & biodiversity conservation *(connectivity)*	wildlife habitat & biodiversity conservation *(biodiversity & wildlife protection)*	integrated pest/weed/disease management, pesticide risk reduction		
Leonardo Academy	soil resources *(erosion, structure, organic matter, biological activity, degradation, nutrients)*	water resources *(quality and quantity)*	biotic resources *(conversion, high ecological value areas)*	biotic resources *(biodiversity, invasives)*	production system *(crop types, pest control)*	energy resources, air quality *(including greenhouse gases)*	waste management
Stewardship Index of Specialty Crops (SISC)	soil organic matter	water use			fertilization	energy use	
Sustainable Agriculture Network (SAN)	soil management & conservation *(erosion, organic matter, nutrients)*	water conservation *(water use, water quality)*	ecosystem conservation *(conversion, connectivity)*, wildlife protection *(conversion)*		integrated crop management *(pest management, chemical use, GMOs)*		integrated waste management
World Wildlife Fund (WWF)	soil erosion / degradation *(erosion, others vary by crop)*	water use, nutrient loading & eutrophication	biodiversity loss & conversion	biodiversity loss & conversion	pesticides & toxicity *(chemical use, GMOs, water discharge)*	climate change & air quality	

Applications of Metrics and the Importance of Tailoring Measurement at Different Scales to their Expected Uses

There are two distinct applications for the metrics we propose: identifying the overall impact of agriculture on the environment (and observing spatial and temporal trends), and identifying the impact of an individual farmer / field (and observing changes as interventions are applied). The former is useful to define the scope of the issue, and identify areas that require further attention. The latter is where we can actually promote changes in farming practices, providing that farmers understand how their actions relate to the impact we hope to achieve.

At the **global** level, coarse metrics identify areas of potential concern that require further study (e.g. agricultural lands at risk of desertification)and changing global trends which may have policy implications (e.g. increasing water scarcity). However, these global data sets are generally too coarse and dependent upon assumptions to be suitable for decision making. For example, GLASOD (Global Assessment of Human-induced Soil Degradation) is useful to identify areas of general concern for soil degradation but inconsistent and non-reproducible with regard to actually quantifying degradation (Sonneveld and Dent 2009).

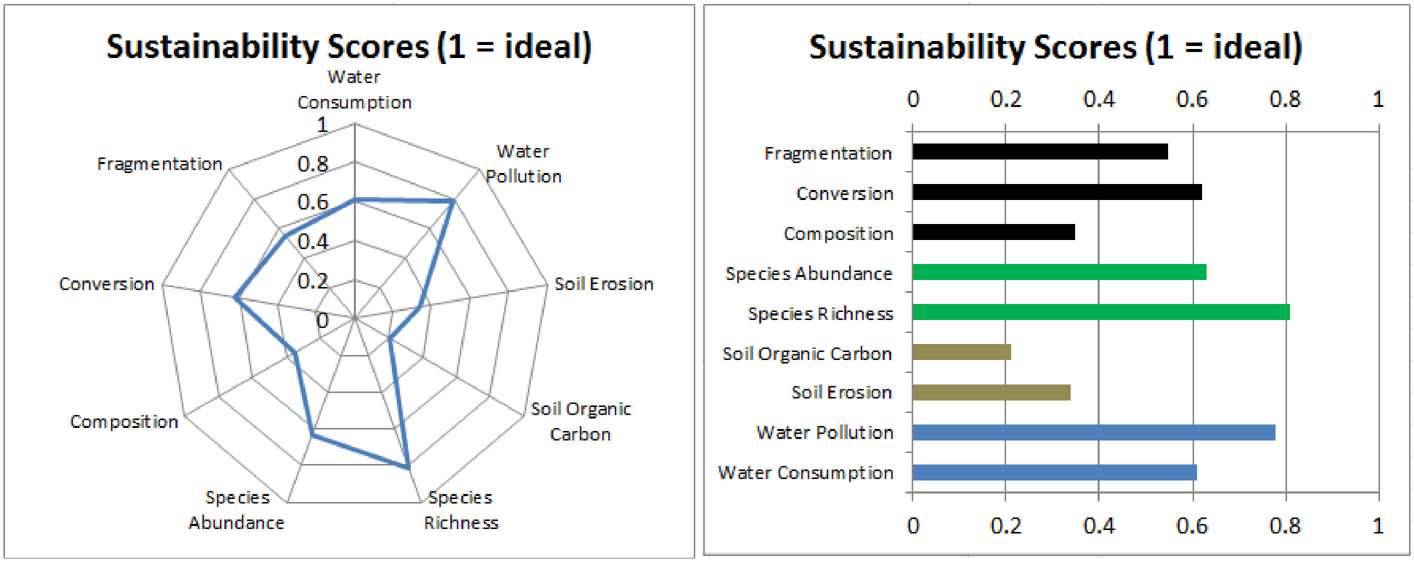
At the **national** or **regional** level, better data sets may permit us to validate or refute global findings and quantify the impact of agriculture in a given region / country (e.g. using national water footprint data). Finer scale data in many countries or regions make it possible to identify more specific trends in ecosystem services, and to relate data on agricultural *activities* (e.g. fertilization and tillage) to *impacts* (e.g. levels of nitrogen and phosphorous in streams). Nonetheless, at this scale data is generally unsuitable on its own to guide local decisions.

We consider the **landscape level** to be perhaps the most important scale. We envision this scale ranging from roughly 500,000 – 1,000,000 acres (2,000 – 4,000 km^2). For example, in the United States this is the size of a "HUC 8" watershed; where to actually draw the boundaries for a "landscape" would depend on both data availability and are of interest. While watersheds are natural ecological units for several of our indicators, it will often be more practical to use counties (or combinations of counties) as data is often aggregated by county. A county-level scale is small enough to make it practical to collect new data on both environmental impact and agricultural management practices (e.g. surveying farmers or commissioning new aerial photography). For example, remote sensing could be used to detect crop residue – materials left over after harvest such as corn stover – which can provide soil stability. At the same time, it is large enough to be able to evaluate the aggregate impact of several farms (e.g. measure water pollution levels). This creates a connection between the activities of farmers, and the impact farming has on the environment (which can then be related to conservation outcomes like changes in fish communities). Given sufficient data, one can compare different landscapes, and even potentially design interventions and test their effect. At this level, we propose not only metrics, but potential ways to normalize the metrics to a scale from 0 to 1. This normalization would allow the comparison of each metric across all four categories via a spider / radar diagram (see Figure 1 on next page) or a small table.

Finally, the **plot** or **field** level is the scale at which it becomes possible to engage with farmers While it is unlikely that changes in the management of a single farm will result in detectible changes in water quality or biodiversity one *can* measure whether or not individual farms are taking the actions that on aggregate, lead to system-level impacts.

In each section below, we break out proposed metrics by scale, and furthermore into data or metrics that already exist, and data that would be useful but do not yet exist (either new analysis using existing data, or new projects to collect data). Two potential examples for what this could look like (with the same hypothetical sample data) are shown in Figure 1 on the next page.

Figure 1. Two examples showing how these metrics could be reported on via diagrams.



While our metrics as proposed are normalized to a simple 0 to 1 scale, it should be recognized that important thresholds (critical values beyond which ecological state changes occur) likely exist within that range (Huggett 2005, Toms and Lesperance 2003). For example, at some point nutrient loading into aquatic systems can become so severe that algal blooms form and anaerobic conditions arise. It is beyond the scope of this paper to identify those thresholds, but basic research into the presence of such thresholds will ultimately be key to any implementation of sustainable intensification programs. For that reason, in each section below we have provided some thoughts about potential thresholds, recognizing that appropriate thresholds will vary with location and ecological context.

Soil Health

Overview

Soil health or soil quality can be defined as “the continued capacity of soil to function as a vital living system, within ecosystem and land-use boundaries, to sustain biological productivity, promote the quality of air and water environments, and maintain plant, animal, and human health” (Doran *et al.* 1996).

Soil Health Indicators

Practically speaking, soil health includes several soil characteristics (physical, chemical, and biological), including sufficient soil organic carbon (SOC), good soil structure, abundance and diversity of soil microorganisms, soil enzymes, presence of nutrients, soil resilience or resistance to degradation, etc. (Doran *et al.* 1996, Doran and Zeiss 2000, Seybold *et al.* 1999, Trasar-Cepeda 2008). While healthy soil is not necessarily fertile soil for agriculture (as crops have specific pH and nutrient needs), it serves as a key foundation for both a healthy ecosystem and a productive farm. There have been several efforts to develop composite soil quality indices (SQIs) that combine measurable soil properties into a metric of overall soil quality (Aziz *et al.* 2011, Jackson *et al.* 2003, Jokela *et al.* 2011, Karlen *et al.* 2006, Seybold *et al.* 1999, Trasar-Cepeda *et al.* 2008). In terms of feasibility, two soil attributes stand out as pragmatic indicators: **soil retention / soil erosion, and soil organic carbon (SOC)**. The two are closely related as soil erosion is a major contributor to declines in SOC within agricultural fields (Gregorich *et al.* 1998).

Erosion is the primary driver of agriculture-induced changes in soil health, and is straightforward to measure, model, and predict. SOC is more complicated (Doran 1996), but still may be remotely sensed at the field scale (details below). A key benefit of these two metrics is that farmers themselves view them as good indicators of soil health (Romig *et al.* 1995) An additional benefit of using SOC as an indicator is that it has been shown to be more sensitive than other indicators to some changes in agricultural management such as crop rotations (Aziz *et al.* 2011, Karlen *et al.* 2006).

Soil Erosion

Soil erosion is the primary form of soil degradation (Lal 2003), and it is a major driver of desertification of agricultural lands as well (Dregne 1986). On a global scale the annual loss of 75 billion tons of soil costs the world about US$400 billion per year (Eswaran *et al.* 2001). Erosion is also a major contributor to agricultural freshwater pollution (primarily via nutrient loading and sedimentation).

Erosion rates from agricultural fields with conventional tillage average 10-100 times greater than rates of erosion under native vegetation or rates of soil production (Montgomery 2007). By contrast, no-till methods reduce erosion between 2.5 and >1,000 times relative to conventional methods (Montgomery 2007). There may be thresholds where below a certain amount of ground cover, soil erosion sharply increases (Davenport *et al.* 1998). There are also several studies showing how to use remote sensing (SINDRI or CAI) to detect crop residue (Serbin *et al.* 2010, Agiular *et al.* 2012, Serbin *et al.* 2009) and thus tillage practices.

While any soil erosion that exceeds soil formation cannot be considered sustainable, in the short term some agricultural soils with thick A horizons may be able to sustain moderate rates of erosion for some time without a significant decrease in yield.

As erosion increases, positive feedback can occur due to channelization and decreased cover (Davenport *et al.* 1998). Where these thresholds occur will vary depending on soil type, slope, and many other factors, and the use of the Revised Universal Soil Loss Equation (RUSLE2) is highly recommended to evaluate this (USDA ARS 2010). **The core metric for soil erosion should be fraction of the soil retained**, and the ideal should be complete retention (or even accretion). It is desirable to have a scale where observed values will potentially fall along the complete range from 0 to 1, but determining what a score of “0” should represent requires further study as complete topsoil loss is unlikely to occur in a short time period.

Soil Organic Carbon

SOC (aka Total Organic Carbon or TOC) is a measure of the carbon stored in soil organic matter (SOM). Baseline levels of SOC depend upon several factors including historic vegetation (Archer *et al.* 2001), soil texture, temperature, moisture, and plant lignin content (Parton *et al.* 1987).

Low levels of soil carbon have adverse impacts on biomass productivity and biodiversity and can increase erosion (Loveland and Webb 2003), while increased SOC content can increase the ability of soil to retain water, leading to enhanced productivity (Lal 2004). Improving the SOC of an agricultural area can benefit both farmers and conservation, as increased productivity can potentially benefit both crops and natural biodiversity (Chase and Leibold 2002). While it has been suggested that SOC <2% is a critical threshold for soil quality, there is inadequate quantitative evidence for a single threshold across all sites (Loveland and Webb 2003).

In addition to affecting productivity of the soil, soil carbon also has implications for climate change; the soil carbon pool is about three times larger than the atmospheric carbon pool (Lal 2003). Furthermore, atmospheric carbon emissions due to soil erosion are significant; roughly equivalent to 16% of carbon emissions from fossil fuel combustion and cement production, or 55% of carbon emissions from land use change (Lal 2003).

Using remote sensing to estimate SOC has shown promise at a field scale, with up to 77% accuracy on bare soil (Chen *et al.* 2000, Chen *et al.* 2005, Gomez *et al.* 2008). Typically reflectance within visible and near infrared (VNIR) bands (e.g. a color aerial photograph) is used to estimate SOC. This is only effective on bare soil (Chen *et al.* 2005), and only measures surface SOC; as conservation tillage practices may increase the SOC in the upper layers of the soil without affecting overall soil SOC (Baker *et al.* 2007, Blanco-Canqui and Lal 2008, Christopher *et al.* 2009, Yang and Wander 1999) and increase ground cover it will not always be possible to accurately remotely sense SOC.

Overall, effects of agriculture on SOC (and soil N & P) depend on the nature and longevity of the agriculture management practices, with fertilization being another key factor (McLauchlan, 2006). In general, climate and parent material define the parameters (bounding levels of C, N, P), while agricultural management (tillage, fertilization, harvest practices, etc.) determines levels within those bounds (Clapp *et al.* 2000, McLauchlan, 2006, Wang *et al.* 2008).

We propose measuring SOC (from remote sensing or aggregated field-scale soil samples) and normalizing the results on a 0 to 1 scale by **dividing the measured SOC by a baseline SOC for that soil type / region** (from SSURGO or similar data sets in other countries).

Measuring Soil Erosion at Different Scales

Global

Existing Data:

The NRCS produced several global maps based on soil climate in 1998, including water erosion (USDA-NRCS 1998a), wind erosion (USDA-NRCS 1998b), and desertification (USDA-NRCS 1998c). These maps could be either used as is or updated (using soil data, slope, precipitation, and land cover). While the maps were based on mean soil climate data from 1960-1990, the author of the maps believes that even with the effects of climate change that they should still be valid (personal communication, April 2, 2013).

GLASOD (Global Assessment of Human-induced Soil Degradation) is an expert-based assessment of global degradation which may also be helpful in identifying areas deserving local focus (ISRIC 2013). It has been identified as inconsistent and non-reproducible (Sonneveld and Dent 2009) with regard to *quantifying* degradation, but it may be appropriate for identifying study areas.

Potential Analysis / Data Collection:

The Global Soil Map project is a global consortium organized by the Digital Soil Mapping Working Group of the International Union of Soil Science (IUSS) working to create a global digital soil map at roughly 100m resolution (GlobalSoilMap, 2013). The Global Soil Map project is using a regional approach, and while the global data are not available yet, data for Australia has been published, and an $18 million effort to map and share soil data for Sub-Saharan Africa is underway. It is not yet clear when the global data will be available, nor whether or not potential for erosion will be explicitly included or if that would have to be derived from other fields present in the data.

Regional / National

Existing Data:

The resolution of the global data on soil erosion suffices for use at the national or regional scale.

The FAO's AQUASTAT database has data on "conservation agriculture area" (how much area is under conservation tillage or no till) at a country level (FAO 2013c).

In the United States, the NRCS Natural Resources Inventory (NRI) includes a statistical sample of cropland soil erosion at the plot scale across the country (USDA-NRCS 2010b). The summary report includes data on erosion at the national scale, and is also broken down into 10 farm production regions. Maps at the national scale also exist which show finer scale erosion data, making it possible to identify areas within the US where cropland soil erosion is a problem.

The Global Soil Map project may be able to provide national or regional data (GlobalSoilMap, 2013).

Potential Analysis / Data Collection:

The Global Soil Map project should continue to publish some national data incrementally.

Landscape

Existing Data:

Within the United States the Natural Resources Inventory is a sample of cropland soil erosion across the United States, and some of the regional breakdowns may be suitable for use at this scale. While the most current fine-scale data (2007) are not available for download from their site, data from 1997 are available for download, and the NRCS may be willing to provide the 2007 data upon request.

Potential Analysis / Data Collection:

To predict or model erosion we need data on agricultural management practices in addition to erosion potential based on soils, slope, etc. There are data sources available in the US at the state level, (USDA ERS 2012) and precise local data may be available if a data sharing agreement can be worked out with groups surveying farmers like Field to Market and the Stewardship Index on Specialty Crops. There are also several studies showing how to use remote sensing (SINDRI or CAI) to detect crop residue (Serbin *et al.* 2010, Agiular *et al.* 2012, Serbin *et al.* 2009) and thus tillage practices (no-till vs. conventional tillage) at a local or regional level. Since tillage has a substantial influence on nutrient loss and soil erosion, we could also investigate soil erosion and nutrient levels at selected study areas. This could complement data on irrigation from SAGE (SAGE, 2001). If this approach is pursued, an attempt to get data on fertilization and harvest practices as well should be made as the data are likely held by the same entities, and this information could be useful in predicting SOC and water quality.

Plot / Field

Existing Data:

As noted under Regional / National and landscape above, fine-scale Natural Resources Inventory data may be available from the NRCS upon request.

Potential Analysis / Data Collection:

We could obtain data either by surveying farmers or via remote sensing on tillage practices. A logical normalized variable would be the **% of field area with ground cover or crop residue** (this metric would be used at both the field and landscape level). A no-till system would have 100% cover, and “conservation tillage” is generally defined as 30% or more cover (Gebhardt *et al.* 1985, FAO 2013c). This data could potentially be rolled up into a landscape level metric. There are a few different remote sensing methods available that could be evaluated, and we could also investigate existing government or private (e.g. Field to Market) sources of data on tillage as described in the national section above.

Measuring Soil Organic Carbon at Different Scales

Global

Existing Data:

The best measures of soil carbon require soil testing, but some coarse global soil carbon data and maps exist (USDA-NRCS 2006, Global Soil Data Task Group 2000). The Harmonized World Soil Database also has information on soil carbon, salinity, and toxicity (FAO/IIASA/ISRIC/ISSCAS/JRC 2012).

Potential Analysis / Data Collection:

The Global Soil Map project (described above under global potential data for soil erosion) will include data on soil carbon; while global data is mostly unavailable as of 2013, it has been released for Australia (Rossel and Chen 2011, GlobalSoilMap 2013).

Regional / National

Existing Data:

The NRCS is completing a rapid carbon assessment in the United States (using actual soil samples); the data are not publicly available yet but should be soon (USDA-NRCS 2010a, personal communication, March 26, 2013). A similar approach has been carried out in Australia (Rossel and Chen 2011) as part of the Global Soil Map project. If the underlying soil sample data behind these national data sets was made available it would allow us to test and validate remote sensing methods for estimating SOC.

Potential Analysis / Data Collection:

Global Soil Map project plans to release additional national data sets as they are completed, although the timeline is unknown.

Landscape

Existing Data:

In the United States the NRCS rapid carbon assessment could be a useful source of data on SOC at the landscape level if the underlying sample data are made available, and the SSURGO / STATSGO databases provide information on the typical or baseline level of soil carbon, which can then be used to compare measured values of SOC.

Further investigation is needed about sources of data on SOC at the landscape scale outside the United States.

Potential Analysis / Data Collection:

As described below, **data on agricultural management practices (e.g. tillage, fertilization, harvest practices, etc.) at the field level should be compiled, and then aggregated** to the landscape level. We could then compare these practices to measured SOC in an attempt to determine which practices are key predictors of SOC.

Plot / Field

Existing Data:

The NRCS data described under the Regional / National section includes roughly 35,000 plot-level sampling points.

Given that the Long Term Ecological Research Network (LTER) has “movement of organic matter” as one of their five core research areas (LTER 2013), it is likely that LTER would be a useful partner in identifying and providing data on SOC at this scale.

Potential Analysis / Data Collection:

As described in the overview for SOC, **insufficient data currently exist to propose a single metric of agricultural practices that can reliably predict SOC**. While tillage is one important factor in determining soil organic carbon rates, it is clear that fertilization practices and harvest practices (whether leaves and stalks – e.g. corn stover – are harvested or left on the field) are also important variables to consider. Data on these practices could be analyzed and compared to measured SOC data in hopes of identifying the key practices to predict SOC.

Freshwater

Overview

A recent study found that 50% of river basins studied (with 2.7 billion people living in them) experience severe water scarcity during at least one month of the year (Hoekstra *et al.* 2012), and with increases in population and food demand, water scarcity is only expected to increase. The continuing conflict between farmers and conservationists in the Western United States (Colby 1998) over freshwater use illustrates the need to find ways to improve management of our freshwater resources.

Freshwater Indicators

Conservation organizations often focus freshwater conservation efforts on goals like avoiding hydrologic alterations (e.g. dams, channelization, reduced flows, etc.) and maintaining or restoring freshwater biodiversity (or other biotic metrics). While these are certainly important characteristics of freshwater ecosystems, they are difficult to measure, and especially challenging for farmers to relate to their activities. **We suggest focusing measurement efforts here on two surrogates instead: water consumption and water quality**.

Water Consumption

Agricultural water use (surface water withdrawals in particular) is the primary driver behind the construction and maintenance of dams, as well as the cause of low flows or even dry rivers in arid regions (e.g. the southern portion of the Colorado River, many rivers in the North China Plain, the lower Indus River, and many more). Agricultural water use in arid / water scarce regions also contributes to desertification or other forms of soil degradation. Data on agricultural water use are relatively easy to obtain, and farmers understand how their management activities influence their water use.

Usually the term “water use” in agriculture means the total quantity of water that either falls on fields as rain or is withdrawn from surface water or groundwater for the purpose of irrigation. However, not all of this water “used” by agriculture is truly “consumed” (evaporated, transpired, or incorporated into a plant) in the sense of being unavailable for other uses. The non-consumptive fraction of water use percolates into groundwater or returns to streams, where it can be used again. We focus here on water consumption as it is the driver of water scarcity concerns (including inadequate flows for streams and inadequate supply for humans).

When the term “water footprint” is used, it breaks water consumption down into three components: “green” (volume of rainwater consumed), “blue” (volume of surface and ground water consumed), and “gray” (volume of water needed to assimilate / dilute pollutants to meet existing ambient water quality standards) (Hoekstra and Mekonnen 2012). Unlike the green and blue footprints, the gray water footprint is partially reusable; it couldn't be reused for the purpose of diluting the same pollutants, but it *could* be used for irrigation or drinking water (after treatment).

The global water footprint from crop production is 7404 billion m^3/yr, which is 78% green, 12% blue, and 10% gray (Mekonnen and Hoekstra 2011). This doesn't include 913 billion m^3/yr for grazing and 46 billion m^3/yr for direct livestock water footprint (Mekonnen and Hoekstra 2012). Altogether, agricultural production takes up 92% of the total global water footprint (Hoekstra and Mekonnen 2012) and accounts for 70% of water withdrawals (surface water and groundwater use rather than consumption) (FAO 2013b).

While green water dominates water consumption globally, blue water footprints are higher in arid regions where water scarcity is high (Mekonnen and Hoekstra 2011). Rain-fed regions have much more opportunity to increase yields (up to 4 times) within the existing water balance (Mekonnen and Hoekstra 2011). As such, **we propose measuring both "green" and "blue" agricultural water consumption, relative to water availability.**

Some efforts to save water have focused on improving the efficiency of irrigations systems by reducing water losses during storage, conveyance, and application to the field. However, improvements in irrigation efficiency do not necessarily lead to a reduction in water consumption or even water use (Samani *et al.* 2012, Ward and Pulido-Velazquez, 2008, Ward 2014). One case study found that improving irrigation efficiency from 60% to 80% reduced the water applied to the field by 15% but led to a 3% increase in overall water consumed (Contor and Taylor 2013). This highlights the importance of measuring outcomes rather than relying on measurements of changing practices; irrigation efficiency is unlikely to increase local water supplies but may be effective in increasing crop yields per unit of land area and water consumed (Ward 2014).

Water Quality

Water pollution (from nutrients, pesticides, or sediment) has a clear, direct impact on freshwater systems. Nutrient loading from fertilization of agricultural lands can cause eutrophication and loss of biodiversity (Smith *et al.* 1999, Carpenter *et al.* 1998). We propose addressing sedimentation via the soil health metric, and exclude pesticides from consideration due to the desire for simple metrics and the challenge of finding good data on pesticide application or concentrations. The most significant factor in evaluating the effect of agriculture upon freshwater systems is likely measuring the contribution of nitrogen and phosphorous to those systems (Carpenter *et al.* 1998).

Agricultural management practices (fertilization, tillage, harvest practices, and riparian buffers in particular) can have a significant effect on the amount of potential water pollution from a farm. Since it is easier to measure implementation of agricultural practices on a farm than to attribute changes in water quality to individual farms within the watershed, measuring these practices as a proxy for water quality is fairly common.

However, recent research on agricultural practices on the Mackinaw river has found that not only is it difficult to encourage adoption of these practices (Lemke *et al.* 2010), but adoption of improved agricultural management will not necessarily lead to an improved outcome of water quality (Lemke *et al.* 2011). While practices were improved that reduced sediment transport (wider riparian buffers and increased area of conservation tillage), tile drainage on the fields allowed dissolved nitrate (the dominant fraction of N in this landscape) to be transported to the stream without being filtered. In fact, nitrate and pesticide concentrations in drainage water may actually be increased under conservation tillage (Stoddard *et al.* 2005, Isensee and Sadeghi 1996).

The Conservation Effects Assessment Project (CEAP) of the USDA-NRCS has also had difficulty demonstrating that improved practices lead to improved water quality (USDA-NRCS 2012a). The USDA-NRCS has offered several recommendations for designing water quality monitoring to ensure that any potential effect will be observable, and similar to the Mackinaw study, highlighted the importance of the right practice improvements being chosen for a given site (USDA 2012b).

There is no single approach that will be effective in all agricultural landscapes. Along the Mackinaw, the authors believe that treating the drained water (e.g. via wetlands) is likely to be the most effective agricultural practice change. It is also possible that the most efficient way to reduce nitrogen load from tile-drained agricultural land is altering the amount and timing of fertilizer applied (Petrolia and Gowda 2006). Unfortunately, data on the location of tile drainage are not widely available, but efforts to map this information are underway using a variety of sources including NRI and USGS (personal communication, July 15, 2013). Even without tile drainage, the effectiveness of riparian buffers is highly dependent on several other factors (Gergel *et al.* 2002, Weller *et al.* 1998).

While less research has been done directly linking the relationship between agricultural practices, water quality, and freshwater fish communities, a recent CEAP project in the Great Lakes Region has begun to do just that (Sowa *et al.* 2011, Sowa *et al.* 2013). Sowa et al (2011) used the Soil & Water Assessment Tool (SWAT) to predict the water quality expected from changing agricultural practices, and developed a model to quantitatively predict changes to fish communities based on water quality and flow. This study is also looking at the predicted impact of climate change on fish communities under various conservation agriculture scenarios. The model is still being refined, and the latest iteration includes tile drainage which should significantly improve the results (personal communication, July 15, 2013).

Improving water quality at the landscape level will likely require ambitious action. For example, the Iowa plan to reduce nutrient loading to the Gulf of Mexico as part of the 2008 Gulf Hypoxia Action Plan (MR/GMWNTF 2008) had a goal of reducing load from nonpoint sources in Iowa by 41% for nitrogen and 29% for phosphorous (ISUCALS 2012). The authors of the plan modeled the effect that different management practices would have, and found the scenario with the *lowest* equal annualized cost (EAC) that met the objective required an initial investment of over $4 billion, plus an ongoing investment of $77 million per year (EAC of $4/acre), and changes on virtually all agricultural lands. The Great Lakes study concluded that even with an investment of $44 million (more than the $41 million Michigan received via the 2008 Farm Bill) to the four focus watersheds, ecological goals could not be met in all streams. As a result they recommended the development of new conservation practices and strategies that could prove more effective (Sowa *et al.* 2013). As Lemke *et al.* (2011) have found, these modeled results may overestimate effectiveness of these agricultural practice changes, so the scope and cost of change needed to affect water quality at the landscape level should not be underestimated.

Measuring Water Consumption at Different Scales

Global

Existing Data:

The Water Footprint Network has published maps of global water scarcity (monthly blue water consumption compared to blue water availability), and the underlying data permit examination just for agriculture (Hoekstra *et al.* 2012). This is likely the most useful metric at the global scale: the **% of total blue water supply being consumed by agriculture**. To fit our normalized scale where 1 is the best output, we would report 1 minus the fraction of total blue water supply that is consumed by agriculture (which can be described as the % of total blue water supply *not* consumed by agriculture).This identifies areas where agriculture is the dominant driver of water scarcity.

Several other global data sets exist with complementary information. IWMI (International Water Management Institute) has produced several global maps relating to agricultural water use / consumption and water supply in their World Water and Climate Atlas: rain-fed cropland, irrigated cropland, and several other relevant layers relating to moisture availability for agriculture (IWMI 2013).To complement the IWMI data, the FAO (Food & Agriculture Organization) has their own global map of irrigation areas (Siebert *et al.* 2007).

The World Resources Institute has also produced a series of 16 global maps called Aqueduct Global Maps 2.0 (Gassert *et al.* 2013) summarized at the watershed level which include several layers of interest: total blue water availability, baseline water stress, drought severity, and more. WRI recently released an interactive tool called the Aqueduct Water Risk Atlas that allows you to view and download this data in an interactive format, as well as to create your own water risk maps by altering the way that the data are weighted and combined (WRI 2013).

Potential Analysis / Data Collection:

The approach used to produce global water consumption maps (comparing precipitation to evapotranspiration [ET]) relies on two assumptions. The first assumption is that we have perfect irrigation efficiency (ET water beyond precipitation is equal to blue water consumption), which leads to underestimating water consumption as water may be lost /consumed during reservoir storage and due to wind drift and evaporation before the water hits the field (Smajstrla *et al.* 1991). Actual irrigation efficiency will vary considerably (e.g. 73-93% efficiency in New Mexico where surface irrigation dominates and most crops are grown under water deficit conditions (Samani *et al.* 2005), to 57-68% efficiency in Indian rice-wheat paddy systems (Jeevandas *et al.* 2008), to 24% in Latin America (FAO 2013d).The second assumption is that optimal growing conditions exist for all crops. Agricultural lands with a water deficit will have lower actual ET than the remotely sensed ET data indicate (Yang *et al.* 2006, Samani *et al.* 2012), which leads to overestimating water use. The information on the degree of water deficit in fulfilling the evaporative demand is generally not available at the country level (Yang *et al.* 2006). Therefore, to improve the quality of this global data, **we could identify areas where irrigation inefficiencies or water deficits would dominate, and calibrate the global data to more local studies**. It appears that this work may not have been done yet at a broad scale.

Regional / National

Existing Data:

The Water Footprint Network has published data on the water footprint of crops (and livestock) which includes national and sub-national breakdowns by blue, green, and gray water footprint (Mekonnen and Hoekstra 2011). The Network has also calculated water consumption relative to the quantity of crop produced; this data can be inverted to calculate water *efficiency* or units of crop produced per unit of water consumed (Keating *et al.* 2010).

The FAO's AQUASTAT database has data by country on agricultural water use / withdrawals, irrigation, tile drainage, water supply, and related variables (FAO 2013b).

These two data sets should suffice to produce estimates of **agricultural blue water consumption as a % of blue water supply** for the regional / national scale.

Landscape

Existing Data:

While many of the data sets mentioned above have a resolution that would in theory allow their use at the landscape / watershed scale, most of them explicitly caution that the data are not designed for this use. However, higher quality versions of the same data sets are available in some areas at a local scale (both remotely sensed and survey based), including evapotranspiration (Couralt *et al.* 2005, Rocha *et al.* 2012), crop types (Jensen *et al.* 2006), and other indirect measurements of water consumption. For example, in the United States county level data are often readily available, but in other countries these data may not be available through a common portal.

The fundamental metric remains the same: the **% of total blue water supply being consumed by agriculture in the landscape**.

Some interesting data on agricultural water footprint by river basin are available in an interactive map (Water Footprint Network, 2013).

Potential Analysis / Data Collection:

Where landscape scale water consumption data do not exist, there are two potential sources of data. One is to assemble plot / field level data (typically from surveys, potentially from remote sensing) across a given landscape. The other would be to develop / interpret new remotely sensed data in a landscape where it does not exist (or where the quality is insufficiently high). There are examples of developing landscape-scale water consumption data via remote sensing in the literature (Samani *et al.* 2012).

Plot / Field

Existing Data:

There are several groups currently working on field-level metrics which include water use and water consumption (see overview section for an overview of these initiatives), mostly in the United States and Europe. These data are currently not being made available to the public, although The Nature Conservancy is investigating options to obtain them.

It is also likely that some of the landscape scale data sets on water consumption (e.g. county level data) which are derived from surveys or field measurements have field level data which are not published.

Blue water availability does not have real meaning at the field scale, so we recommend focusing efforts instead on **measuring water consumption per acre** and **per unit of crop yield**. One way to normalize these data would be to calculate the % of crop water consumption on a given field that is blue as opposed to green, or to take watershed values of blue water supply and multiply by plot area divided by total watershed area (or other landscape estimates of total blue water supply).

Potential Analysis / Data Collection:

As noted above, there are some excellent field level data on water use and water consumption being collected but which are not publicly available. It would be extremely valuable to be able to obtain these data, even if their use was restricted by a nondisclosure agreement. An effort could be made in focal areas to obtain data via counties (or other local governments), and organizations like Field to Market (who are asking farmers to report these data). These data could then be used to calibrate and validate coarser data sets and remotely sensed information.

Measuring Water Quality at Different Scales

Global

Existing Data:

A global map of nitrogen load in rivers was produced in 2011, and validated with nutrient data measurements around the world (He *et al.* 2011). In a similar vein, MacDonald *et al.* (2011) produced a map which identified areas of “phosphorous surplus” where more phosphorous was applied than could be used by crops. Additional work has been done to quantify phosphorous flows (US EPA 2011) although maps do not appear to have not been created and published from this work.

GEMStat is a global index of local water quality monitoring data, and could also be used to identify areas of concern (Barker *et al.* 2007). It includes data on nutrient pollution, as well as suspended solids, dissolved oxygen, and other relevant indicators.

At the global and national level we recommend focusing on **identifying areas with relatively high nitrogen and phosphorous loading in streams**.

Potential Analysis / Data Collection:

To our knowledge, a global map of phosphorous load in rivers has not been produced. However, the combination of the phosphorous surplus data and national data should meet the need for global data to identify areas of concern for phosphorous pollution.

Regional / National

Existing Data:

National data exist on agricultural gray water footprint from Water Footprint Network (a measure of how much water is needed to dilute pollution).

Some countries have excellent national data on nutrient loading. For example, in the United States the National Water-Quality Assessment (NAWQA) Program has detailed information on water quality trends and conditions in 51 basins across the nation. The SPARROW model also provides N and P data by HUC 8 watershed in most areas. The Water Quality Portal from the National Water Quality Monitoring Council (NWQMC) includes extensive data for the United States, Canada, and Mexico; and limited data for Cape Verde, Guatemala, Iraq, and Peru (NWQMC 2013).

While tile drainage is not a direct measure of water quality, it is important to consider if water quality is to be modeled. Data on tile drainage are available for some countries, although it is more useful at the landscape or plot scale. For example in the United States Pavelis (1987) is the best available national data despite its limitations, including problems with consistency, accuracy, spatial resolution, and its age (WRI 2007).

The Soil Health section of this paper has ideas for getting at erosion and soil loss (the primary drivers of sedimentation).

We believe these data sets should suffice to produce estimates of water quality for the regional / national scale.

Landscape

Existing Data:

Since water quality within a stream is influenced by both the water upstream and the land within the watershed, it is arguable that measured water quality is fundamentally a landscape-scale metric. As such, data from the Water Quality Portal (NWQMC 2013) should be suitable for use at the landscape scale.

In 2007 the World Resources Institute reanalyzed the data on tile drainage from Pavelis (1987) to produce estimates of tile drainage at the county level (WRI 2007) for 18 Midwestern states within the United States. There is additional work currently being done to map tile drainage in the Great Lakes states by USDA ARS SWAT modelers based on various sources, including NRI and USGS sources (personal communication, July 15, 2013).

Given that the Long Term Ecological Research Network (LTER) has “movement of inorganic matter” (with a focus on mineral nutrients including nitrogen and phosphorous) as one of their five core research areas (LTER 2013), it is likely that they would be a useful partner in identifying and providing data on water quality at this scale.

Potential Analysis / Data Collection:

To measure water quality at the landscape level, we recommend **comparing measured nitrogen and phosphorous loads relative to the Total Maximum Daily Load (TMDL) for the watershed** where these TMDLs are available. Since these limits vary substantially by watershed (as nutrient thresholds vary considerably), and have not been quantified in many places there is no one single meaningful standard. To producing a normalized metric on a 0 to 1 scale, we recommend dividing the nutrient limit by the measured nutrient load, and rounding any value over 1 down to 1 (if the load is below the limit, this will be the case).

F**urther research is needed in order to better understand how agricultural management practices impact water quality**. Measuring agricultural practices alone will not enable accurate predictions of water quality impacts, although it is important to continue enhancing our understanding of how these practices do relate to water quality. The work being done by The Nature Conservancy and NRCS represent an excellent opportunity to collaborate to answer this crucial question.

Work being done by The Nature Conservancy in the Great Lakes region is currently underway to improve the ability of models to predict the effect of changes in agricultural management on fish communities, including the incorporation of tile drainage into their model (personal communication, July 15, 2013).

Another potential research topic to explore would be to compare self-reported data on conservation tillage, riparian buffers, and other practices to remotely sensed data. While some data sets rely on actual sampling, in many cases a survey is the sole source of information. To our knowledge the accuracy of these reports has not been investigated.

Additionally, at this scale the *impact* of water quality can perhaps best be measured through measurements of the fish and aquatic macroinvertebrate communities (where local data exist or can be collected). Several indices of biotic integrity (IBIs) exist which can provide valuable information on water quality by comparing species composition to known information about the tolerance of those species to various conditions (Lammert and Allan 1999).

Plot / Field

Existing Data:

As noted in the water use section, there is substantial data in some areas at the county level on agricultural management practices (including tillage). While work is being done to strengthen our ability to model changes in water quality and even freshwater fish communities due to changing agricultural practices (Lemke *et al.* 2011, Sowa *et al.* 2011, Sowa *et al.* 2013), we are not yet confident that we can make quantitative predictions about how conservation agriculture practices will impact water quality. Actually using measured agricultural practices as a proxy for water quality is not recommended.

Potential Analysis / Data Collection:

The work proposed under the landscape scale section above would apply at this scale as well

Landscape Ecology

Overview

Landscape ecology describes the relationships between ecological processes at a variety of scales, and in a variety of human-modified ecosystems, including agricultural landscapes. It has been estimated that up to 83% of the terrestrial biosphere is under some type of human influence (Sanderson *et al.* 2002), and that 50% of the world's surface has been converted to grazing land or cultivated crops (Kareiva *et al.* 2007). Another way to measure our impact is the appropriation of net primary productivity (NPP), which globally, is estimated at 30% (Imhoff *et al.* 2004). Anthropogenic habitat conversion (leading to differences in landscape composition and fragmentation) is also a key driver of global biodiversity loss, and much of this is a direct result of the increasing footprint of agriculture (Andren 1997, Fahrig 2003, Gaston *et al.* 2003, Vandermeer and Perfecto 2007).The Atlas of Global Conservation (Hoesktra *et al.* 2010) that has a map showing arable land not yet converted which could be used to focus on important landscapes, those that still have some natural lands available for conservation.

Landscape Ecology Indicators

While landscape ecology is a complex topic, by utilizing remotely sensed land cover data we can calculate three indices which collectively provide information on key ecological processes at the landscape level. We propose examining **conversion** (the proportion of an area that has been converted from a natural state to agriculture), **landscape composition** (the number of habitat types and their relative abundance) and **fragmentation/connectivity** (the degree to which large patches of natural habitat are broken up into smaller isolated patches). These three metrics should be measured at a single point in time to establish a baseline, and tracked over time to obtain trends.

Most available land cover data sets are not developed with conservation in mind, and thus are not ideal for measuring these characteristics. The development of a conservation-specific land cover database is a potential project that would be not only useful to evaluating sustainable agriculture, but for many other applications as well. Several complementary remote sensing technologies exist to measure land cover, from visible and infrared satellite imagery, to radar (in very cloudy regions and moist/wet forest/wetlands), to a complex LIDAR classification of habitat structure (studies use LIDAR to exploit its ability to record detailed and accurate measurements of height through different levels of forest). While vegetation indices (such as NDVI) and spectral remote sensing are commonly used in ecology, radar and LIDAR are becoming more widespread as more data are being collected, and becoming cheaper to collect. Remote sensing information needs to be calibrated and ground-truthed, typically via field visits and botanical surveys.

Species differ in their tolerance to changes in conversion, composition, and fragmentation; as natural habitat is converted thresholds of interest are likely to exist as individual species of interest and ecosystem services such as pollination are adversely affected or extirpated (Huggett 2005, Riitters *et al.* 1997, Keitt 2009). Since appropriate thresholds vary considerably depending on the species or ecosystem service under consideration, we have not proposed any within the metrics below.

Conversion

Conversion refers to the percentage of natural land that has been converted to some form of human use. While conversion might be seen as a form of composition, it should be studied independently as it has a direct bearing/impact on diversity. Not all conversion is equal in impact; conversion to pasture is likely to have less impact than conversion to row crops or urban land uses. Generally low intensity conversion will be harder to detect, especially at coarse scales (global / regional) where high-resolution data is likely to be unavailable.

Landscape Composition

Composition is an indicator of the number and amounts of different habitats within a given area (Uuemaa *et al.* 2009). This indicator has a strong correlation with biodiversity (including in agricultural landscapes) – the more heterogeneous an area, the more species are likely to occur (Benton *et al.* 2003, Goetz *et al.* 2007).

Fragmentation/Connectivity

Fragmentation refers to the degree to which large patches of natural habitat are broken up into smaller isolated patches. Highly fragmented landscapes have low connectivity. While typically the term “patch” is used to refer to a given type of habitat (e.g. forest or grassland), we use the term here to refer to a contiguous area composed of any mix of natural habitat types. In other words, a forest adjacent to a natural shrubland would count as one patch in our measurement scheme. Since fragmentation is a landscape level process rather than a patch-level one, measures for fragmentation are generally more appropriate at the global, regional, and landscape level than the plot or field level.

Relatively unfragmented habitats are thought to have more inherent conservation value than do fragmented ones, with species richness declining with decreasing fragment size (Quinn and Harrison 1988). Fragmentation is therefore an important representation of landscape ecology, as its impacts are a far-reaching, significant, and measurable aspect of changing habitat dynamics. It should be noted that in the study of fragmentation, typically the higher the resolution of your data (the more detailed the imagery), the more disturbance is detectable.

The primary metrics to measure fragmentation/connectivity are **largest fragment size** and **connectivity.** Largest fragment size is measured by the largest patch within a given study area (i.e. a watershed or ecoregion), and is often used as the primary indicator of fragmentation (Hoekstra *et al.* 2010). There are a few different types of connectivity scores and connectance indices. Most are based on weighted distance between patches, with variations of least-cost distances, habitat quality and user specified distance thresholds. The connectance index in FRAGSTATS gives a range from 0 to 100, which can be converted to 0 – 1. The connectivity score is similar, but is potentially a more robust metric in that it encapsulates functional connectivity through the use of least-cost distances between patches, and allows for the incorporation of additional measurements of patch habitat quality (Opdam *et al.* 2003). We could also potentially create a new statistical representation of connectivity that allows the combination of the spatial habitat needs of many species into a single metric. We also propose using the **interior-to-edge ratio** (the higher the better, meaning more core habitat and less edge) at the landscape at plot scales.

Other indicators of fragmentation that could potentially be incorporated at finer scales if desired include nearest-neighbor distance (average distance to the nearest neighboring patch of the same habitat type) and proximity index (similar to the connectivity score - the sum of the ratio between patch area and inter-patch distance for all patches within a specified buffer distance around a focal patch) (Gustafson and Parker 1994).

Measuring Landscape Ecology at Different Scales

Global

Existing Data:

Global land cover data are currently only available at moderate resolution (300m pixel size), and moderate accuracy. The best available data source is the GlobCover 2009 Global Land Cover Map from the European Space Agency (ESA 2010). Rather than using these data as is, we propose reclassifying it into two categories for “natural” and “converted” as described below. For global data sets fragmentation should be measured by watersheds, with HYDROSHEDS of the World, by Lehner *et al.* (2008) working the best for our purposes.

Potential Analysis / Data Collection:

We propose reclassifying the GlobCover 2009 dataset into two categories: natural (forest, grassland, shrubland, etc.) and converted (urban, cropland, pasture, etc.). Mosaic cropland would be scored as converted, and mosaic vegetation would be scored as natural. This allows us to consider “patch size” from the perspective of natural habitat as opposed to converted habitat, as opposed to having to consider each habitat type individually.

From this reclassified dataset, we would use FRAGSTATS software to calculate the **largest patch size by watershed** (normalized by total watershed area) as our global metric of **fragmentation**.

To measure **conversion**, we would simply calculate the **percent natural area** (as opposed to converted area) within each watershed.

To measure **composition**, we *could* use the original GlobCover 2009 data, and calculate the number of land-cover classes that occur within each watershed. However, there are not enough natural classes (14) to create a meaningful and comparable result, so this metric will not be used. Instead, we will focus on conversion and fragmentation at the global scale.

Regional / National

Existing Data:

Relatively high resolution data (30-100m – Landsat type data), that have been classified at a regional/national scale (not global) is needed for this scale. This is available for many countries including the following: USA, Australia, 10 West African countries (FAO Africover) and seven Southeast Asian countries (Asiacover).

Potential Analysis / Data Collection:

Higher resolution data (on three levels: spatial, temporal and classification resolution) are needed, and will produce different indicators from the global level. Both spectral and radar data should also be considered for land cover classification. All indicators are likely to show more impact (worse environmental quality) as resolution is increased.

There is a strong need in conservation for high resolution, high accuracy land cover maps. Development of such a layer is a potential project that would be extremely useful for measuring sustainable agriculture, as well as for many other applications.

The approach to measure **conversion** (measuring **percent natural area by watershed**) and **fragmentation** (measuring **largest patch size by watershed**) would be the same as outlined in the Global section above, but using higher resolution land cover data. While these two metrics are the most critical to complement the indicators for other categories (freshwater, soil health, and biodiversity), there are several other potential metrics that would be useful indicators of landscape ecology. At this scale it becomes possible to measure **composition** as described in the Global section above (measuring the **number of land-cover classes that occur within each watershed**).

Landscape

Potential Analysis / Data Collection:

Very high resolution data (>10m resolution) is needed at this scale. LiDAR data are likely to be incorporated to increase resolution (both spatial resolution and classification classes) and increase the accuracy of the land-cover classification. Owing to the addition of much improved data, additional indicators will be used. The improved data will require extensive on-the-ground field work to calibrate and ground-truth remotely sensed data.

The approach to measure **conversion** (measuring **percent natural area by watershed**) would be the same as outlined in the Global and Regional / National sections above, but using very high resolution land cover data. Given the important of riparian buffers (for connectivity, soil health, water quality, and biodiversity), we could also measure the **mean width of riparian buffers** (non-converted habitat) along all streams within a given landscape as another facet of conversion.

To measure **fragmentation** the key metric remains **largest patch size by watershed**, but we also recommend measuring **connectivity** and an **interior-to-edge ratio** (the higher the better, meaning more core habitat and less edge). There are a few different types of connectivity or connectance indices, but most are based on weighted distance between patches, with variations of least-cost distances, habitat quality and user specified distance thresholds. The connectance index in FRAGSTATS gives a range from 0 to 100, which can be converted to 0 to 1.

To measure **composition** we recommend measuring the **number of land-cover classes that occur within the landscape** (as described above using imagery), but also supplementing it with aggregate data from botanical survey work (at the plot level) or other ground-level data. This should include a reference site if possible (an ecologically similar site, in very good or pristine condition – usually a protected/wilderness area – which the agricultural plots can be measured against). Multiple sites/plots will be needed to understand the complete natural composition of landscape, and to train the remotely sensed data.

With core research and numerous studies on primary production and disturbance patterns, Long Term Ecological Research Network (LTER 2013) sites could provide invaluable information, and should be investigated for inclusion into this analysis (i.e. the effects of land cover change on forest communities, Turner *et al.* 2003). Many of the LTER sites could be used as reference sites to compare to agriculture sites to (where appropriate).

Note that the use of remote sensing technologies (radar, LiDAR and spectral) to map different habitat types and model species richness also ties in with the biodiversity section (especially birds) – on a landscape to plot/field scale (Goetz *et al.* 2007, Goetz *et al.* 2010).

Plot / Field

Potential Analysis / Data Collection:

Extremely high resolution data (1m) are needed at this scale. LiDAR and radar should be used in conjunction with spectral data, and hyperspectral data should be considered. Plot/field analysis will require intensive field work for remotely sensed data calibration and ground-truthing of results. Even at this finest scale level, plot level indicators are all relative to other plots/sites in the landscape, including a reference site (if possible).

The approach to measure **conversion** (measuring **percent natural area by watershed**) would be the same as outlined in the other sections above, but using extremely high resolution land cover data. As with the landscape level, we could also measure the **mean width of riparian buffers** along all streams within the plot (if any are present).

The approach to measure **fragmentation** (**largest patch size by watershed**, **connectivity score**, and **interior-edge ratio**) would be the same as outlined in the Landscape section above. To measure **composition** we recommend both measuring the **number of land-cover classes that occur within each plot** (as described above), but also supplementing it with data from botanical survey work (as described under the Landscape section). The botanical survey data should be compared to multiple sites and a reference site as a BACI type analysis.

Biodiversity

Overview

Humanity has a significant impact on the biodiversity of the planet, driven largely by habitat conversion (Soulé 1991, Forester and Machlist 1996). As population increases (along with economic growth and resulting consumption), the conversion of unconverted arable land is likely to increase (Hoekstra *et al*. 2010), which in turn will cause further declines in biodiversity in those areas.

In an attempt to mitigate the complexity and potential considerable expense of comprehensively measuring biodiversity in agricultural landscapes, we propose looking at **richness** and **abundance** indicators of two surrogate taxa, **birds** and **amphibians**. There are data available on both at all scales of interest, and birds and amphibians show complementarity in use of the landscape. Birds have especially have been shown to work well as a surrogate for biodiversity (Larsen *et al.* 2012). Amphibians are added due to their global threatened status, and have been shown to be susceptible to changes in freshwater quality and quantity, soil condition (Sparling *et al.* 1995) and use of pesticides (Weltje *et al.* 2013) – they can be seen as nature's "canary in the mine".

Note that at the plot or landscape scale, measuring other species such as fish or benthic macroinvertebrates may be both more practical and more directly relevant to understanding the impact of water quality problems (e.g. using the Index of Biotic Integrity [IBI] for fish or Benthic Index of Biotic Integrity [B-IBI] for macroinvertebrates, Lammert and Allan 1999). However, since data on fish and macroinvertebrates are generally not available at coarse scales, we chose here to focus on species with greater data availability. This should not preclude the use of fish or macroinvertebrates to provide useful information about the effect of agriculture on biodiversity.

Other efforts regarding measuring biodiversity and sustainable agriculture are underway. Field to Market plans to release a framework to measure the impact of agriculture on biodiversity soon, the Stewardship Index for Specialty Crops (SISC) is looking into the topic as well, and The Sustainability Consortium is also beginning to investigate the topic. We anticipate that these efforts will focus on agricultural management practices and coarse landscape proxies like habitat conversion, and as such could be highly complementary with an effort to actually measure biodiversity itself within agricultural systems.

Pollination

It should be noted that while our main focus here is in considering the impact of agriculture *on* biodiversity, biodiversity can also provide ecosystem services *to* agriculture. Since one third of the global food supply is facilitated by animal pollination (Klein et al 2007), this is an important service to consider. Wild-pollinated agricultural products have a value that may be as high as $127 billion per year worldwide (Costanza *et al.* 1997.) Especially in light of current bee declines, it is particularly important to understand the dynamic interactions between pollinators, biodiversity, and agro-ecosystems (Kevin *et al.* 1990).

While it appears that pollination services are more closely tied to landscape ecology factors (amount of natural habitat, composition of that habitat, and spatial arrangement of the habitat) than habitat with high biodiversity in general, it is true that overreliance on a single pollinator makes crops vulnerable when that pollinator declines (Allen-Wardell *et al.* 1998). As such, pollination is often discussed in terms of biodiversity, under the assumption that more biodiverse habitat will support a broader suite of pollinators, which in turn means resilience to changes that impact specific pollinator species.

According to the models of Keitt (2009), the creation of large, uninterrupted crop fields significantly diminishes pollination services though the density of pollinators increases. Keitt (2009) also concluded that the extinction threshold for plant-pollinator systems generally occurs at 50-60% habitat loss, unless the converted crops provide pollen, in which case as much as 90% habitat loss may be tolerated.

Habitat patches and disturbance are also important aspects to consider when investigation pollinator diversity. Patches in most agricultural landscapes vary in both their size and the resources they offer pollinator populations (Kremen et al 2004). Disturbed areas and agricultural fields themselves can offer nesting and floral resources (Brosi et al 2007, Williams and Kremen 2007), but the quality of these resources is highly dependent on farming practices (e.g. pesticide applications, tillage (Holzschuh et al 2007). As noted above, this is more properly treated as a landscape ecology issue than a biodiversity issue,

Maintaining pollination services requires the conservation of sufficient resources for wild pollinators within agricultural landscapes, including both suitable habitats and sufficient floral resources for pollen and nectar (Kremen *et al.* 2007). Bees are also central-place foragers (i.e. they return to fixed nest sites after foraging), so proximity of nesting habitats relative to agricultural fields is critical for bee-pollinated crops (Ricketts *et al.* 2008). Farms within foraging distance of suitable habitat may thus receive better pollination services, while those further away may not.

Several recent studies have investigated whether crop pollination services decline with increasing isolation from natural habitats. Ricketts *et al.* (2004) found that bee diversity, visitation rate, pollen deposition rate, and fruit set are all significantly greater in coffee fields near tropical forest than in fields further away. Pollinator richness and visitation rates significantly and exponentially decline with increasing distance from natural habitat (Ricketts *et al.* 2008). Even in those areas rich in biodiversity, pollination services decline as distance from natural habitat increases (Carvalheiro *et al.* 2010). However, other studies have found little effect of landscape pattern on pollinator visitation. For example, Winfree *et al.* (2007) found that the landscape pattern had little impact on pollination services to vegetable crops in the northeastern United States.

The role of soil biodiversity in protecting against disease and insect pests, and in increasing the efficiency of nutrient and water uptake, has already been discussed in the soil section

Biodiversity Indicators

The scope and expense of comprehensively assessing biodiversity is considerable an all-taxa inventory of one hectare of tropical forest might take 50-500 scientist years to accomplish (Lawton *et al.* 1998). Larsen *et al.* (2012) found that birds work well as a biodiversity surrogate in areas with good bird diversity, but that adding species data for other taxa with range-restricted species improves the effectiveness of the surrogate (with no particular individual taxa being especially complementary).

We propose using **birds** and **amphibians** as surrogates to measure overall biodiversity. While there is no perfect indicator species or taxa, we hypothesize that these two surrogates should be complementary, as they depend upon different kinds of habitat and are affected differently by environmental change. From a practical perspective, there are an enormous amount of data available on birds, which are often referred to as flagship indicator taxa in biological inventories (Lawton *et al.* 1998), and amphibians have in their own right been extensively studied and have high interest due to their globally threatened status (Stuart *et al.* 2004). Both can also be potentially be monitored remotely using remote and automated digital audio recording (Acevedo and Villanueva-Rivera, 2006). Both taxa will require repeated measures, especially in breeding seasons and early mornings, for a complete picture of diversity.

The first indicator for the two taxa is species **richness** - the number of different species represented in an ecological community, or for our purposes, the number of species within a given unit area. At coarse scales (global and regional), we propose using a simple species count and list, while at finer scales (landscape and plot / field), we could normalize this measure to a 0 - 1 scale, calculated as number of species per unit area/number of species from a reference area (of similar size that is relatively intact / natural).

The second indicator for each taxon will be **abundance,** which refers to the number of individuals of a species in a given area. Not all species are equally valuable from a conservation perspective, so we propose summarizing the individuals by their IUCN red list status (IUCN 2012): least concern, near threatened, vulnerable, endangered, or critically endangered. While the specific equation to use would require further study, the basic idea would be that in adding up the count of individuals within each category, rare species would be weighted more highly than common species. At coarse (global and regional) scales abundance is less meaningful (entire populations might be accounted for), but at finer scales (landscape and plot / field) the weighted abundance metric could potentially be normalized to a 0 - 1 scale by calculating the weighted abundance of a reference site. Since trend data on abundance and richness may not be available, IUCN red list status at least provides some indication of current or recent declines.

Typically discussions of thresholds in biodiversity refer to situations where beyond a certain set of habitat conditions (e.g. oxygen levels in water, natural habitat conversion on land, etc.) biodiversity sharply declines (Vaquer-Sunyer and Duarte 2008, With 2004). It is also true that as biodiversity declines, there may be a threshold beyond which biodiversity declines more sharply, whether due to interdependencies on species being extirpated or simply the resilience of most species being exceeded (Perrings and Pearce 1994). However, identification of any such thresholds is beyond the scope of this paper.

Note that measuring biodiversity in and around agricultural systems is only a first step. Eventually we would like to be able to quantify the effect that agriculture has on biodiversity, but given the many factors which can influence biodiversity this is a complex research challenge.

Birds

Birds are perhaps the most widely and extensively studied vertebrate taxa in the world, there is an enormous amount of data available on them, and they are very popular with citizen scientists – being relatively easy to identify and study

Birds score highly on many of the criterion recommended by Gregory *et al.* (2003) for selecting indicator taxa, including the following important points:

- Birds occupy a wide array of habitats and many have large geographical ranges; they are less specialized within microhabitats than plants or insects, and reflect patterns in unrelated taxonomic groups (BirdLife International, 2004).
- Changes in bird populations have also been shown to be a useful indicator of environmental change (Bennun and Fanshawe 1997, Donald *et al.* 2001, Gregory *et al.* 2003). For instance, the UK government has adopted an index based on wild bird populations as one of its 15 headline Quality of Life indicators.
- In terms of scale, birds are very useful indicators of species richness and endemism patterns at global and regional levels (Bibby *et al.* 1992).
- At finer scales, while patterns of bird distribution may not always well match the distribution patterns of other taxa (Prendergast 1993, Pearson 1995, Lawton *et al.* 1998), it has been shown that a network of sites selected as important for birds will capture most other biodiversity (Howard *et al.* 1998, Brooks *et al.* 2002). The network of sites identified as important bird areas (IBAs) covers 80% of the area identified for other wildlife groups (Birdlife International 2004).

Amphibians

Amphibians may also be useful indicator taxa. Amphibians are globally distributed (Global Amphibian Assessment 2004), have been recognized as threatened and globally declining, and have been extensively studied (Collins 2003). Amphibians are good surrogates for biodiversity, and despite their frequently high degree of ecological specialization, their representation of other taxa is often high (Moore *et al.* 2003).

Amphibian on-the-ground monitoring could be accomplished at the same time as birds, using automated digital recording systems (Acevedo and Villanueva-Rivera 2006, Dorcas *et al.* 2009). Amphibian health also has potential human health implications; amphibians are extremely sensitive to environmental change due to their sensitive, porous skin (Márque and Alberch 1995). Endocrine disruptors, such as BPA, in freshwater were first seen to feminize male frogs before their impacts on humans were suspected (Kloas *et al.* 1999). Similar trends have been observed with amphibians and other environmental toxins (Carey and Bryant 1995). Pesticide levels, freshwater quality and quantity, soil condition, and countless other factors influence amphibian population distributions (Welsh and Ollivier 1998, Culman *et al.* 2010).

Measuring Biodiversity at Different Scales

Global

The main intent of the global analysis is to establish the richness of each taxon, summarized by watershed. These watershed richness summaries will also be used as potential baselines/reference to normalize local/finer-scale richness measures against.

Existing Data:

There are several global data sets available we can use:

- Amphibian species distribution data and abundance are available from the Global Amphibian Assessment (GAA) – completed in 2004.
- Species distribution data for birds (used to calculate richness) are available from Birdlife. While there are no comprehensive, reliable global sources of data on bird abundance, a potential proxy is BirdLife International's Important Bird Areas (IBAs) data set (Birdlife International 2013a). This data set includes 11,000 sites which have been identified as holding significant numbers of threatened or range-restricted species, and/or have exceptionally large numbers of migratory species.
- The Hydrosheds watershed boundaries (as described in the landscape ecology section) provide a useful data set to summarize each of these data sets to provide a common unit of analysis.
- As described above, we would use the IUCN Red List data (IUCN 2012) for birds and amphibians to determine how to weight the abundance of each species into an overall weighted abundance metric (one number for birds, another for amphibians).

Potential Analysis / Data Collection:

We propose intersecting the watershed data with each of the species distribution data sets to produce a count of the **number of bird species in each watershed** (**richness**). In addition to providing a coarse global metric of species richness, we could also use these data as the baseline for bird richness when we examine finer scales.

In the absence of better data on **bird abundance**, we propose counting the **number of Important Bird Areas intersected by each watershed**, and could include threatened species data which would indicate a downward trend.

Regional / National

Existing Data:

Wild bird indicators show the average trends in **abundance** of a selected set of common species. Data already exist for the United Kingdom and Europe, data are being created for North America and Africa (based on Breeding Bird Surveys (USGS 2001)), and early research has begun to produce a global Wild Bird Index (BirdLife International 2013b).

There are considerable regional and national sources of data for **birds** (both **richness** and **abundance**) that should be investigated. This list includes (but is not limited to): Breeding Bird Surveys (BBS, yearly data for the United States), U.S. State Bird Atlases e.g. Delaware Breeding Bird Atlas (Delaware Division of Fish and Wildlife 2012), Southern Africa Bird Atlas Project which is repeated every 10 years (SABAP2 2013), Tanzania Bird Atlas (monthly data, TBA 2013), Atlas of Australian Birds and Birdata (Birdata 2004), Bird Atlas Britain (BTO 2011), and Bird Atlas Europe (EBC 1997). Further research is needed to evaluate the various data sources and determine which are the best suited for our purposes.

There are volunteer-collected data and maps for **amphibian richness** in the United States (North American Amphibian Monitoring Program, USGS 2012). Amphibian **abundance** and **trend** information is available for the United States through the Amphibian Research and Monitoring Initiative (USGS 2013a). Some other resources include InfoNatura (species richness maps for groups of birds and amphibians in Latin American and the Caribbean, Natureserve 2007); and Reptile, Amphibian, and Fish Conservation Netherlands (RAVON 2013).

Potential Analysis / Data Collection:

Existing data on bird or amphibian abundance could be reanalyzed to produce a weighted abundance (by IUCN red list category).

It is apparent that there is a lot of potential for amassing data via citizen science projects. Projects in the United States (Breeding Bird Survey and North American Amphibian Monitoring Program) and internationally (Wild Bird Indices) could be evaluated to determine the feasibility of obtaining early data and assisting in replicating their approaches elsewhere.

Landscape

Measures at the landscape scale will be similar to Regional/National measures, just with finer detail. However, at this scale it becomes possible to normalize data relative to reference sites to produce a 0 to 1 metric. In all cases we could **divide the richness or abundance in a given agricultural landscape to the richness or abundance in a reference natural landscape** (to show how much of an effect agriculture has had on richness and biodiversity). **Alternatively**, we could use the **richness calculated by watershed at the global scale (see above) as the baseline** for these calculations (which would be much less effort than identifying and measuring reference sites).

At this scale it is important to measure whether a given area is a "source" or "sink" for birds or amphibians. In other words, are the richness and abundance measured due to good habitat quality within the landscape (a source), or due to the proximity of neighboring high quality habitat (meaning that the agricultural landscape is a "sink" for those neighboring areas)?

Existing Data:

It is likely that in some landscapes local biodiversity data will be available (e.g. data from the Natural Heritage Programs in the United States, NatureServe 2012). With numerous population studies, Long Term Ecological Research Network (LTER 2013) sites could provide invaluable information on both bird and amphibian communities, and should be investigated for inclusion into this analysis (i.e. 30-year bird population trends in a temperate deciduous forest, Holmes 1990). Many of the LTER sites could be used as reference sites for which plot sites get compared against (where appropriate).

Potential Analysis / Data Collection:

We recommend that **data at this scale should be aggregated from the plot/field level** (see that section below for details).

Plot / Field

While biodiversity is perhaps best analyzed at a landscape level to capture the overall effect of a matrix of varied habitat, it is best *measured* at a plot or field scale. As with the landscape scale, it would be possible to normalize these data by comparing the richness and abundance for amphibians and birds in a given plot to that of a reference plot, but we anticipate that these comparisons are more useful at the landscape scale.

It is very likely that certain farming practices will positively or negatively impact biodiversity. However, while agricultural management practices are also measured at the plot level, **the effect of specific agricultural practices on biodiversity has not yet been sufficiently studied**.

Note that the contribution to biodiversity at the plot level is very difficult to measure, although we can measure some landscape ecology factors at a plot level (e.g. % area in natural land cover vs. converted, riparian buffers, etc.) that should correlate with biodiversity. The NRCS Conservation Measurement Tool (CMT) scores the conservation value of a farm (including biodiversity), and unlike tools like Field to Market also incorporates surrounding land use (USDA-NRCS 2013).

Existing Data:

As noted in the introduction of this paper, there are several ongoing efforts to collect data on agricultural management practices at the plot scale. We anticipate that the efforts will focus on using agricultural practices as a proxy for biodiversity, making them an ideal complement to direct measurements of amphibian and bird richness and abundance (to test the validity of the proxies).

While some organizations and networks (such as the LTERs) are involved with efforts to measure biodiversity at the plot scale in agricultural landscapes, given the breadth of the sustainable agriculture field, it would be worth continuing to search for additional partner organizations for this work.

Potential Analysis / Data Collection:

To measure bird and amphibian abundance and richness, we propose a monitoring project. Microphones could be placed around the area of interest, and bird/amphibian songs would then be recorded and uploaded via cell-network. Then the data could be analyzed (possibly at a partner institution such as Cornell) to produce estimates of the number of individuals of each species, which would be summarized into richness and weighted abundance. These monitoring stations could be placed throughout a given landscape so that the data could readily be aggregated to that scale. This represents a low-effort way to obtain data (potentially over the long term) relative to conventional surveys. As described above, these techniques have been successfully used before, although we do not believe that they have been applied to agricultural landscapes yet.

To complement and calibrate / validate this audio monitoring, actual ground surveys should be used to count amphibians and birds on a stratified random sample of the plots which will be monitored using audio recordings. The methodologies for transects and point counts are well understood in the literature (and have been partially used in TNC's Patagonia Sustainable Grazing Project). We recommend multiple surveys done per plot – to account for the seasonality of the taxa – especially amphibians and birds (spring or summer). In order to observe changes in biodiversity as agricultural management practices change, several years of data would be required: a few years of baseline data before the change, and a few years after, not counting the time it takes for a transition in practices to occur.

The combination of this new plot level biodiversity data and existing data sets of agricultural management practices should make it possible to begin to study the effect that varying practices has on biodiversity. This can then validate or invalidate the use of measuring specific agricultural practices as a proxy for directly measuring biodiversity, and allow us to focus efforts to improve agricultural practices on the ones that will do the most good. Depending on the quality and availability of the data, a good beginning would be to sample fields for which good data exist on practices and split the sample into fields that meet or do not meet the recommended practices for biodiversity.

A more robust study design would be to use a BACI (before-after control-impact) study model to investigate the following hypotheses:

- H1: Farms that implement the recommendations of __ organization (e.g. Field to Market) for biodiversity will show improved biodiversity. This could be tested for each organization that provides data for this study (with each set of metrics / recommendations constituting a separate hypothesis).
- H2: Conservation tillage (measured as % of area with crop residue or other ground cover) is positively correlated with biodiversity (especially for amphibians).
- H3: Smaller habitat patches (as measured in the landscape ecology section) are correlated with reduced biodiversity.
- H4: Increased land clearing / conversion (as measured in the landscape ecology section) is correlated with decreased biodiversity (the higher the % of farm area converted to crops, the lower the biodiversity).

Discussion

To fully implement the recommended measurement framework of this paper would be a substantial undertaking. While considerable data and research exist on the topic of sustainable agriculture, much work remains to analyze that data within a cohesive framework. Once the objectives for a measurement program have been set and the general metrics have been decided upon, we propose breaking the work up into categories based on level of effort.

Collect Existing Public Data

As detailed above, there are several existing global data sets already available to the public that may already be suitable to guide decisions about where to focus research; including maps of water scarcity, maps of erosion risk, maps of bird and amphibian richness, etc.

In addition to data that can be used as is, there is a considerable body of data that we would need to support the analyses we propose. This includes species distribution maps, land cover data, global evapotranspiration data, etc. These data should only be collected once it has been decided which types of analysis will be conducted, but once that is clear this represents another valuable initial step.

Obtain Existing Private Data

Several of the metrics we propose would be greatly enhanced by the use of data that are not currently publicly available. This includes underlying data behind some of the published global map products (e.g. raw data on water footprints), data being collected by other sustainable agriculture organizations (e.g. Field to Market, Stewardship Index for Specialty Crops, The Sustainability Consortium, etc.), and non-public data held by government agencies such as the USDA-NRCS. For scientific data that underlies published work, a simple request to the authors is likely to suffice, although extra time must be allowed to obtain these kinds of data.

For the data held by other organizations, we are in the early stages of investigating the possibility of obtaining access. These groups have strict non-disclosure policies intended to protect the privacy of farmers, and in some cases provide questionnaire templates but do not collect data centrally. However, in some cases government agencies have been willing to provide sensitive agricultural data under a non-disclosure agreement (e.g. the NRCS has made substantial amounts of data available to participants in their CEAP).

It is our hope that if we can obtain access to these data, we can better assess to what degree these proxy metrics are helpful in addressing our mutual goals of promoting truly sustainable agriculture.

Analyze / Process Existing Data

Once we have obtained the existing data we need (whether it is public or not), there are several new data sets that can be produced to inform our work. This includes projects like producing global summaries of bird richness, conversion, and fragmentation by watershed. While in many cases these analyses represent a substantial amount of effort, they do not rely upon the collection of new raw data. This means that the analyses can begin as soon as the data has been collected and staff time to work on them has been found.

Measuring the Impact of Sustainable Intensification

Much of the work in sustainable agriculture is predicated on assumptions which have currently not been sufficiently tested. One assumption is that “sustainable intensification” (whereby yields are increased while impact on the environment is maintained or even improved) is possible in many agricultural landscapes. Another assumption is that as yields increase in some areas, marginal croplands will be allowed to return to their natural state (or at least agricultural expansion will slow down). These assumptions have been criticized as potentially flawed (Vandermeer and Perfecto 2007). While large scale agricultural experiments may not be feasible, quasi-experiments in which project sites that implement sustainable intensification could be compared to matched areas that lack sustainable intensification efforts. This will be especially important given potentially controversial partnerships with corporations that advocate GMOs and other practices that may alienate some of the public.

Acknowledgments

Producing this paper would not have been possible without the support and participation of a number of people. Special thanks go out to Paul Armsworth, Patrick Beary, Jennifer Biringer, Bryce Contor, Dave DeGeus, Sasha Gennet, Evan Girvetz, Kris Johnson, Eloise Kendy, Marshall McDaniel, Rob McDonald, Sean McMahon, Rebecca Nelson, Sunny Power, Carmen Revenga, Scott Sowa, Luis Solórzano, Martha Stevenson, Michelle Wander, and Frank Ward for providing critical input while we were writing it. Thanks also to the many staff and members of The Sustainability Consortium who helped us understand one approach being taken to sustainable agriculture metrics, and the many challenges involved in developing and implementing them.

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