DEVELOPING "REALITY CHECKS" ON ECOSYSTEM SERVICE VALUES:

CHARACTERIZATION AND BOUNDING RESULTS FOR A BROAD CLASS OF MODELS

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ABSTRACT

Despite the volume of work that is being done on ecosystem services, it is often not clear how to identify credible estimates of their value. In this paper I suggest some standards by which we might gauge the plausibility of estimates of ecosystem services values. I highlight a previously unnoted commonality among many types of ecosystem services: they may be regarded as "treatments" performed on a finite set of objects. I show how a number of types of ecosystem services may be interpreted in this framework, work examples for pollution treatment, flood control, and pollination, and note a number of other potential applications. The interpretation of ecosystem services as treatments leads to three implications:

- I call the first the "If-a-little-goes-a-long-way, you-don't-need-a-lot" effect: the more effective an ecological asset is in providing initial treatment – and hence, the more valuable it is initially – the less treatment remains to be performed, and hence, the less valuable the asset will be on the margin when it is more abundant.
- We can derive an upper bound on the marginal value of the value of the n^{th} unit of the ecological asset providing an ecosystem service *regardless* of how effectively it provides treatment.
- There is a sort of *Catch-22*: The conditions under which abundantly available ecological assets would provide the most valuable services are also those under which substitute sources of such services might be most attractive.

These implications do not imply that ecological assets providing some services in some places are not valuable. To the contrary, they could be very valuable. My results show, however, that the conservation incentives generated by ecosystem services may be limited. The very conditions under which incentives are strongest to conserve *some* ecological assets are also those under which incentives for conserving large quantities of them may be weakest.

These results may have many practical applications. These include conducting "reality checks" on valuation studies, conducting preliminary scoping studies to determine when more detailed analyses might be justified, identifying "sufficient statistics" to conduct valuations, informing empirical approaches, guiding benefit transfer exercises, and suggesting ancillary approaches to inform ecosystem service valuation.

Keywords: Ecosystem services; ecological production function; marginal value; extreme value distribution; exponential function; bounds.

JEL classification codes: Q51; Q57

I. Introduction

The rise in interest in ecosystem services over the past few decades has been well documented (see, e. g., Fisher, *et al*., 2009; Munns, *et al*., 2015). In the words of two prominent contributors to the literature on the topic, a focus on ecosystem services[1] has sparked a "growing feeling of Renaissance in the conservation community. This flows from the promise in reaching . . . for new approaches that align economic forces with conservation" (Daily and Matson 2008). Many researchers are now asking under what circumstances the economic value of the goods and services provided by natural[2] habitats and their constituent elements might exceed the opportunity cost of exploiting them more intensively for other purposes. A number of academic, government, and international programs has arisen for the study of ecosystem services (see, e. g. Olander 2014, Landers and Nahlik 2013, Sukhdev, *et al*. 2010, WAVES 2015). Curiously, however, given that much of the interest in ecosystem services arises from a desire to make an economic case for the preservation or restoration of natural habitats, there appears to be what one set of authors who recently surveyed the literature characterizes as a "blindspot" in the actual application of ecosystem service values in decision making (Laurans, *et al*., 2013).

If ecosystem service values are not widely applied in decision making it is likely because the economic valuation of ecosystem services remains a fledgling undertaking. While wide-ranging efforts have been made in recent years, the situation remains largely as it was when Peter Kareiva and Susan Ruffo introduced a special issue of *Frontiers in Ecology* on ecosystem services by noting that

> [G]etting beyond the platitude of nature's value has proven to be a challenge for both science and policy. Why? Because we have not yet found a convincing way to talk about this issue . . . Because we do not have enough science to back up our hypotheses of how and when services are delivered . . . . In short, because we have not proven, on the ground, that these ideas work. [2009; p. 1]

In this paper I aim to provide some standards with which we might determine whether and when such ideas "work on the ground," which I interpret as "demonstrate values large enough to offset the opportunity costs of conservation". Key aspects of this agenda are to identify the minimal set of information we would require in order to determine ecosystem service values (the "sufficient statistics" for policy making, in the sense of Chetty 2009), and bounding results with which we can determine

[1] The term "ecosystem services" has come to be used to refer to the capacity of ecosystems to produce both goods and services.

[2] The question of what comprises a "natural" habitat in a human-dominated era is a thorny one from both scientific and philosophical perspectives. Despite the difficulty of definition, there is often a sense in the literature on ecosystem services that what we are most concerned with are the services provided by "natural" ecosystems. Gretchen Daily's definition of ecosystem services as *"the conditions and processes through which natural ecosystems, and the species that make them up*, sustain and fulfill human life" (1997; emphasis added) better represents my sense of how the term has come to be used than does a broader definition, such as that proposed by the Millennium Ecosystem Assessment of ecosystem services as simply "The benefits people obtain from ecosystems" (2005).

when estimates of value are plausible. I show how to obtain such results for a broad class of ecosystem services by exploiting a previously unnoted commonality among ostensibly different types of services.

Valuing ecosystem services is inherently difficult. Services provided by preserved or restored areas of natural habitats and their constituent elements include things like storm and flood protection, recharge of groundwater, treatment of pollution, carbon sequestration, providing habitat that protects animals that may provide pollination or pest control services to adjacent farm fields, and the preservation of endemic species that might prove to be of value in research, or which may simply be appreciated for their continuing existence. These services are often externalities that cannot be captured by the person who owns the habitat, and thus which do not command observable prices.

Ecosystem services are produced by ecological assets. The formula for calculating the value of an asset is deceptively simple. The marginal value of an asset – the price it would command if it could be allocated in a market – is equal to the marginal value of the good or service it produces times the marginal product of the asset in producing that good or service.[3] To calculate the marginal product of an ecological asset we must know the *ecological production function* by which a natural ecosystem generates goods and services that benefit humanity (see, e. g., Polasky and Tallis 2009). Progress in valuation has been slow largely because there are a multitude of such production functions and their attributes are not always well known.

There is, however, a common feature of many the ecological production processes that we can exploit. Many may be interpreted as a series of “treatments” applied to some exhaustible set of objects. The more pollution is treated by a riparian buffer strip, for example, the less remains to be treated. A similar interpretation may be made of probabilistic phenomena. The larger is the area of wetlands conserved to retain precipitation, the less likely is it that a heavy enough rainfall to cause flooding will occur. This interpretation of ecosystem services as treatments leads to a generic three-factor expression for the marginal value of the ecological assets providing the services:

The marginal value of the ecological asset in producing a good or service	=	The marginal value of the good or service being produced	x	The effectiveness of the marginal ecological asset in providing treatment	x	The size of the set remaining to be treated when or if the marginal ecological asset is employed

Much of what I say in the remainder of the paper follows from the observation that considerations which may make the middle factor larger tend to make the first[4] and third factors smaller. Consider the

[3] One could, or course, also note that marginal values and marginal products could vary with the state of the world and introduce stochastic considerations. I largely avoid these embellishments here, though the results I demonstrate below could be expanded to incorporate them. At the risk of getting ahead of myself, such stochastic considerations would not obviate the considerations that drive my results.

[4] For reasons I will explain below, I will general focus on countervailing effects on the second and third factors rather than those involving the first.

example of pollution control by riparian buffer areas. The natural science literature on the topic often refers to the "removal rate" of a meter of riparian buffer (see, e. g., Polyakov, *et al*., 2005): the fraction by which pollutant concentration is diminished as a polluted stream traverses a meter of buffer. The higher is the removal rate, the more pollution the marginal meter of riparian buffer could remove *if the flow traversing it were heavily polluted*. However, the higher is the removal rate, the *less pollution will remain to be removed.*

These considerations motivate three key implications which can be used to characterize and bound ecosystem service values:

- Let me characterize the first implication as the "*If a little goes a long way, you don't need a lot*" principle. If ecological assets are very effective in providing a service, then the first few units of assets will be very valuable. However, this necessarily implies rapidly diminishing returns. It is important to be clear. The point here is not the simple and unexceptionable observation that diminishing returns imply downward-sloping demands. Rather, it is that if we drew implied demands for ecological assets providing services calculated at higher and lower rates of effectiveness, they would *cross*. The fact that the value of the first unit of an ecological asset providing a service will be *higher* if it is more effective in providing that service necessarily implies that the value of the N^{th} unit of that asset will be lower than it would be if it were less effective, for a large enough value of *N*.

- An immediate corollary to the above implication is that *there is an upper bound* on the value of the marginal unit of an ecological asset in a certain application as a function of the number of units of the asset providing the service. This observation might be useful when natural science research is not available to tell the economist how effective ecological assets are in providing a particular service at a particular place. While upper bounds may be relatively loose in many circumstances, they can at least tell us whether particular claims of value are within the realm of possibility, and if the maximum possible value of an ecological asset would be enough to cover the opportunity cost of its conservation.

- I will call the third implication the *Catch-22* effect.[5] As just explained, when an ecological asset is relatively abundant, its marginal value will be *higher* if it is *not* very effective in providing the service for which it is valued. If ecological assets are *not* very effective in providing a service, however, it may be more cost-effective to address the problem by different means. Ecological assets typically provide services that could be procured from substitute sources. The *Catch-22* effect is that the conditions in which marginal values would *not* decline into insignificance are also those under which it may be preferable not to rely on ecosystems to provide such services at all.

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[5] If readers – perhaps particularly younger readers? – do not recognize the allusion, *Catch-22* was the title of Joseph Heller's 1961 novel of the Second World War, and has entered the vernacular as a description of a self-contradictory proposition. In the novel, Catch-22 prevented a soldier from seeking a discharge from military service on grounds of insanity, as seeking a discharge in the face of such extreme danger would demonstrate that the soldier was, in fact, behaving rationally.

My goal in this paper is both modest in some respects and ambitious in others. It is modest, in that I do not posit definitive estimates for the value of any particular ecosystem service at any particular place in this paper. My purpose here is illustration rather than estimation. My goal is also ambitious, though, in that I propose a generic framework for the valuation of many, if admittedly not all, types of ecosystem services. Using procedures such as those I sketch below, one *could* derive characterization and bounding results for a multitude of particular services and places. Doing such exercises, elaborating them, and reducing them to practice should be a priority for ongoing research on ecosystem service valuation. While this research agenda might be both an extension of, and a complement to, current practice, it would also make a major contribution by providing much-needed "reality checks" on valuation results.

This paper consists of six sections, including this introduction. In the next three sections I illustrate the general ideas I have introduced in this introduction with examples of particular ecological production functions, note their underlying similarities, and illustrate how the key implications I highlight arise in each example. In the fifth section I briefly consider how the framework might be applied to other examples. I conclude in a final section.

Let me give a preview of those conclusions now. The results I develop might be put to a number of uses. These include evaluating the plausibility of valuation studies, doing preliminary scoping studies to determine when more detailed analyses might be justified, identifying "sufficient statistics" to inform policy analyses, cross-checking other empirical exercises, providing guidance to and checks on benefit transfer exercises, and suggesting ancillary considerations than can be used to cross-check ecosystem service value estimates.

II. Pollution treatment

In this and the following two sections I will present some worked examples of how the framework I have described above may be employed. As all three examples share some formal similarities, I will also take some space at the conclusion of this section for remarks that overarch these three examples (as well as others that will be discussed more briefly in Section 5).

One of the ecosystem services often mentioned that retained areas of natural habitat such as wetlands or vegetated riparian buffers provide is filtration and treatment of pollution runoff, particularly nutrients from farms (see, e. g., Daily, et al., 1997, MA 2005). A huge number of studies have been conducted of the efficacy of natural areas in providing such services (see, for surveys, e. g., Mayer et al. 2007; Rupprecht, et al., 2009; Mander 2008). A number of authors have proposed that the pollution control services provided by wetlands or riparian buffers can be modeled as declining exponential functions of the width of such areas (see, e. g., Mander 2008, Weissteiner, et al., 2013, Sweeney and Newbold 2014,): each additional meter of buffer removes the same fraction of pollution from the flow traversing it.[6] Thus, if L_0 is the edge-of-field pollution loading, then after traversing a buffer of width w, an amount

[6] The size of this fraction depends on factors that differ between locations, depending on factors such as the types and diversity of vegetation in the buffer zone (Mander 2008), the composition of soils, the topography of the drainage area, and connectivity to groundwater (Vidon and Hill 2006; Sweeney and Newbold 2014). The literature is also replete with contributions addressing whether, and under what circumstances, the exponential function

$$L_w \;=\; L_0 e^{-\gamma w} \qquad (2\text{-}1)$$

of the pollution load will remain to enter into vulnerable environments, such as estuaries, reservoirs, or lakes. The parameter γ is a measure of the efficacy of the ecosystem in providing the service. Referred to in the natural science literature on riparian buffers as the "removal rate" (see, e. g., Polyakov, *et al*., 2005), γ is the fraction of the load that is removed by the next meter of the riparian buffer or wetland. Estimates of this parameter have been reported in, or may be inferred from, a number of studies of the effectiveness of natural systems in providing pollution removal (for example, rates of between about one and three percent per meter may be inferred for areas in the Chesapeake Bay watershed from the work of Weller, *et al*., 2011)

If we suppose that damage is a function of the pollution load, $D(L_w)$, then

$$D\left(L_w\right) \;=\; D\left(L_0 e^{-\gamma w}\right) \qquad (2\text{-}2)$$

The value of the marginal meter of buffer is, then,

$$-\frac{\partial D\left(L_0,\, w\right)}{\partial w} \;= v\left(w\right) = \; D'\left(L_w\right)\gamma L_0 e^{-\gamma w} \;. \qquad (2\text{-}3)$$

I have inserted a leading minus sign to follow the convention that benefits are positive: a *reduction* in damages is an *increase* in benefits.

Let me also note here that I will, in this and the examples that follow, use the notation $v(\cdot)$ to represent marginal value as a function of the measure of the ecological asset (riparian buffer width w, in this case).

The three-part expression for marginal value I described above is readily apparent in (2-3): the value of the marginal meter of buffer is the marginal damage avoided, $D'(L_w)$, times the effectiveness of a meter of buffer in providing the service, γ, all times the pollutionthat remains to be treated after polluted water has traversed the first w meters of buffer, $L_0 e^{-\gamma w}$.

Let me now consider in turn the three implications I noted above. After reviewing them, I will make some observations regarding the value of pollution treatment services, and then conclude this section with some observations concerning the generality of the model.

"If a little goes a long way, you don't need a lot"

The marginal value of the *first* meter[7] of riparian buffer is the product of marginal damage, the gross load, and the removal rate:

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provides an accurate approximation to the true rate of pollutant removal. I will address the robustness of my findings to such details at the end of this section.

[7] I should, perhaps, not ask the reader to take "first meter" literally. There may well be threshold effects which compromise the effectiveness of extremely narrow buffers (see, e. g., Ferraro 2003). The literature suggests that such effects, if they arise, are overcome quickly, however. One survey article states

$$v(L_0, 0) \;=\; D'(L_w)L_0\gamma \qquad (2\text{-}4)$$

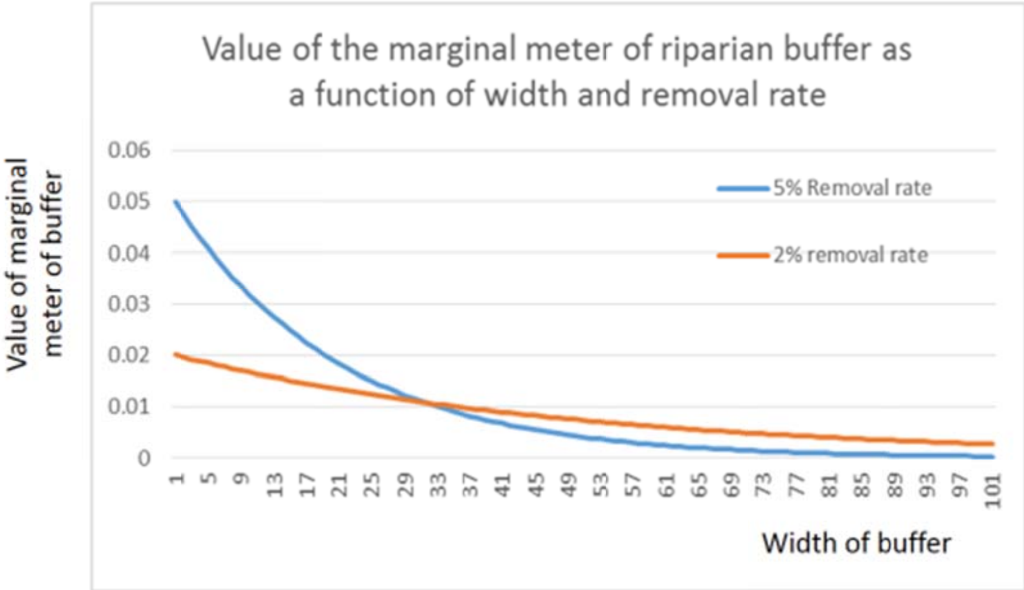
Other things being equal, the larger is the removal rate, γ, the more valuable will be small areas of riparian buffer. For reasons I will detail at the conclusion of this section, I will suppose that the marginal damage from pollution does not vary with the removal rate. Under this assumption, the value of the marginal meter changes with the removal rate, γ, as

$$\frac{\partial v}{\partial \gamma} \;=\; D'L_0 e^{-\gamma w}\left(1-\gamma w\right) \qquad (2\text{-}5)$$

The sign of (2-5) is determined by $1 - \gamma w$, and so the larger is γ, the lower will be the value of the marginal meter of a wide buffer. If one finds that the *first* meter of riparian buffer is very valuable, it may mean the w^{th} meter is not very valuable. Figure 1 illustrates this notion. When the marginal meter of riparian buffer is highly effective in removing pollution, its value is initially very high (blue curve). However, as a consequence of this high initial removal efficiency, there is less pollution remaining to be removed, and the value of the marginal meter declines rapidly. Because of the latter effect, the value of the marginal meter would be higher beyond a certain width – about 30 meters in this example – if a meter of buffer were *less* effective in pollutant removal (red curve).

Based on the results obtained over *c*. [*sic*] 30 years . . . The removal of [pollutants] has a nonlinear character: in the first part of the buffer (0–5m from the field–buffer borderline), significantly more material (20–60%) is retained [*i. e., removed from the flow*] than in the remote parts of the buffering ecosystem. [Mander 2008, p. 2047]

Figure 1



It is worth underscoring that the phenomenon illustrated in Figure 1 is not trivial. It is saying more than noting that demand curves slope down. The important point is that the curves in Figure 1 *cross*: a high removal rate would make the marginal product of an ecological asset more valuable when it is scarce, but make make it less valuable when it is abundant.

Upper bound

If one knew the pollutant load, L_0, buffer width, w, and marginal damage, D', but did not know the removal rate, she could find an upper bound on the marginal value of the w^{th} meter of buffer by setting the derivative in (2-5) to zero. Doing so, an upper bound would occur if

$$\hat{\gamma} \;=\; 1/w \;. \tag{2-6}$$

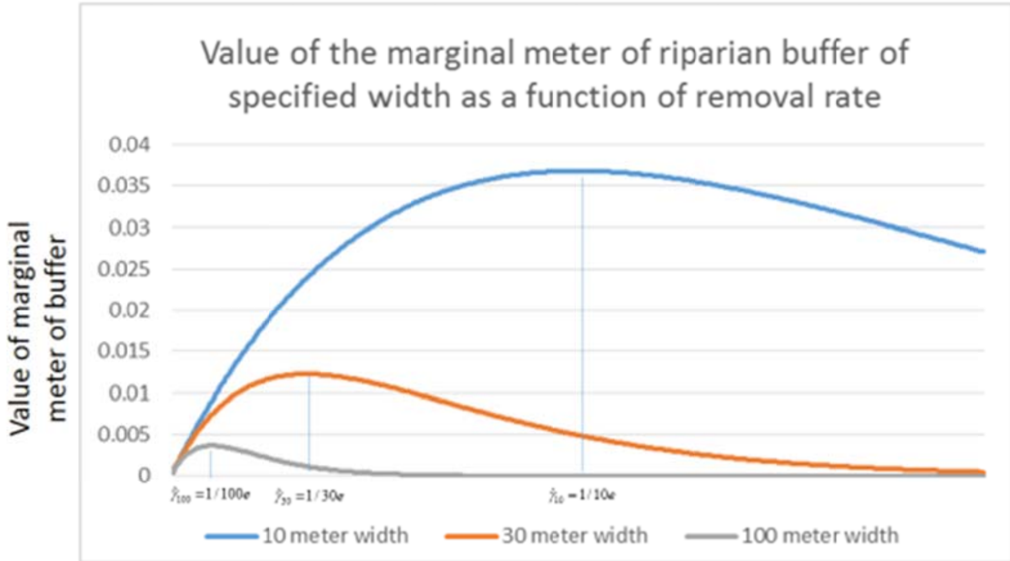
I will use "hats" over variables to represent the values they take on at the upper bound on marginal value. Substituting (2-6) in (2-5), the upper bound is

$$\hat{v} = D'(L_w)\hat{\gamma}L_0 e^{-\hat{\gamma}w} \;=\; D'(L_w)L_0/we \tag{2-7}$$

Given the marginal damage and the pollution load, then, the value of the marginal meter would be greatest if the removal rate were on the order of the inverse of the buffer width.

Illustrative bounding results are depicted in Figure 2. As in Figure 1, the value of the marginal meter of riparian buffer width is on the vertical axis, but now each curve represents a buffer of a different width – 10, 30, or 100 meters – and the removal rate parameter, γ, varies along the horizontal axis. If a buffer were only 10 meters wide, the marginal meter could provide a valuable service under a relatively wide range of removal rates. As the buffer under consideration is made wider, however, the upper bound on marginal value declines, and that upper bound is achieved at lower and lower values of the removal rate. At a removal rate for which the 10th meter of a buffer might provide the most value, the marginal value of the 30th or 100th meter might be negligible.

Figure 2



Catch-22

To illustrate the *Catch-22* result, let me expand the problem somewhat. Let us suppose that the width of the buffer is fixed at w, but that we now have the option of reducing the loading, L_0. At what level would it be optimal to set L_0, and how would that level vary with the removal rate, γ?[8] We can subsume output and factor prices, geographical features, etc., in a reduced form and represent the profits of a firm or farm as a function $\pi\left(L_0\right)$.

[8] The reader might think of w being determined by other considerations – perhaps an appreciation of both the pollution treatment services we are considering now and other ecosystem service values such as carbon sequestration or animal habitat -- and the question now being how would that value change as the removal rate changes. Of course, one could optimize (2-8) with respect to both L_0 and w. The results are qualitatively the same as those of the simpler exercise I report now, but the calculations are more tedious.

The social objective would be to maximize the consumer surplus net of production costs and environmental damage. As is well known, this is, in the absence of any further market distortions, equivalent to maximizing the difference between profits and pollution damages,

$$\pi\left(L_0,\, w\right) - D\left(L_0 e^{-\gamma w}\right) . \tag{2-8}$$

Gross loading should be chosen so that

$$\frac{\partial \pi}{\partial L_0} - D' e^{-\gamma w} \;\; = \;\; 0 \; . \tag{2-9}$$

Assuming again that marginal damage is constant, totally differentiate expression (2-9) to find

$$\frac{\partial^2 \pi}{\partial L_0^2} \frac{dL}{d\gamma} + D' w e^{-\gamma w} = 0 \;\; , \tag{2-10}$$

or

$$\frac{dL}{d\gamma} = -\frac{D' w e^{-\gamma w}}{\partial^2 \pi \big/ \partial L_0^2} > 0 \;\; , \tag{2-11}$$

since increased loading would have diminishing returns to profits.

If we increase the removal rate, we could then also increase the loading, as the extra pollution would now be treated more effectively. Conversely, if the removal rate were *lower*, we would optimally set loading lower. This is not surprising. Increasing riparian buffer width and reducing gross loading are alternative approaches for doing the same thing: reducing delivered pollution load. They are substitutes. Other things being equal, the *less* effective riparian buffers are, the *less* loading would optimally be generated. Thus we have the *Catch-22*: The value of the last meter of a wide riparian buffer would be larger the lower is the removal rate. If the removal rate were relatively low, it would be more attractive to reduce gross loading. But if gross loading were reduced, the value of the marginal meter of buffer would be lower.

The value of pollution treatment services

I will not report detailed estimates of values for ecosystem services in this and the next two sections, but will note what the empirical implications of the considerations I raise might be.

It is difficult to estimate the marginal damage resulting from an additional kilogram of pollution loading, but in a number of places we have an alternative procedure for placing value on loading: authorities require reductions from other sources. If pollution reductions realized by riparian buffers can substitute for reductions from other sources, we can use the marginal cost of abatement from other sources as a proxy for the marginal value of pollution reduction. Perhaps the marginal cost of abatement from other sources does not reflect the true social cost of pollution, but it does measure the cost society incurs to achieve reductions from other sources. Studies of potential gains from water pollution trading programs establish that riparian buffers are cost-effective (see, e. g. CBC 2012; Ribaudo *et al*., 2014), which is equivalent to saying that the service they provide justifies their opportunity costs. In a sense, though, riparian buffers may be *too* effective to motivate extensive conservation. Because buffers reduce pollution very effectively, the value of the marginal meter of buffer declines quickly.

Of course, the amount of conservation an appeal to ecosystem services can justify depends not only on the value of the services provided by lands reserved for the provision of ecosystem services, but on the opportunity cost of such lands. If the opportunity cost is low, an appeal to ecosystem service values may be more compelling. By the same token, though, it is not difficult to justify the conservation of land whose opportunity cost is low (some suggest that much of modern "conservation" policy is simply "saving" resources whose isolation puts them at little risk of destruction, e. g, Asafu-Adjaye, *et al.*, 2015).

Some remarks on assumptions in this and the following sections

I am generally trying to lay out simple and schematic arguments, and so am relying on both parsimonious models and assumptions. This is true both in this section and those that follow, so let me briefly say something about my modeling choices. Classic contributions in both the social (see, e. g., Friedman 1953) and natural (see, e. g., Levins 1966) sciences address the appropriate level of complexity to employ in model-building. While detailed models with a multitude of parameters will certainly be useful in some circumstances, simpler specifications more clearly illustrate fundamental points. Results demonstrated using canonical models create a presumptive case that they will generalize to all-the-bells-and-whistles specifications.

Having said that, let me briefly defend two aspects of my models. The first concerns the marginal value of the ecosystem service provided, and the second the choice of the exponential functional form.

In the analysis above I assumed that marginal damages are constant. This is probably not literally true in many cases, nor will the analogous assumptions I make in the examples below likely hold literally. However, I make this assumption for three reasons:

- In many instances we are considering the value of a particular ecological asset, not some wide-ranging change in all quantities of the ecological asset everywhere. We might, for example, wonder about the value of an extra meter of riparian buffer strip width along a particular kilometer of stream, not that of adding a meter to all stream-kilometers in an entire watershed. Such local changes in the provision of ecosystem services would likely have little effect on the marginal damage from water pollution.

- My results concern bounds on values, and would hold *a fortiori* under the more likely alternative assumption that damages are convex in delivered pollution load.

- Most importantly, when I consider changes in the removal rate, or in the other examples below, changes in a parameter representing efficacy, I am suggesting a thought experiment, not contemplating a policy choice. One of the reasons for suggesting bounding results is that, while marginal damages may be a datum in some contexts, removal rates may be unknown. Taking the former as given, we can calculate the value of the latter for which the marginal meter of a buffer would command the greatest value.

Let me next address the exponential form the ecological production function takes in this example and those that follow. I have taken this assumption from a very extensive natural science literature on the topic. In the examples that follow the exponential form arises directly from first principles. While it certainly is convenient that the natural science literature supports this choice, I have not adopted the assumption for convenience. I might also note that this functional form is implicit in models put forward for the express purpose of valuing ecosystem services (InVEST 2015). If one assumes such a functional form, certain testable implications arise from it, and I am noting them here.

Having said this, however, one might still wonder what the implications would be if the sources I cite are wrong, and an exponential function is not an accurate depiction of the processes I consider. It is, of course, impossible to consider all the infinite alternatives one might adopt, but one generalization of the exponential function, (2-1), may be revealing.

Suppose that

$$L_w = \begin{cases} \left(L_0^{\beta} - \beta\gamma w\right)^{1/\beta} & \beta \leq 1, \beta \neq 0 \\ L_0 e^{-\gamma w} & \beta = 0 \end{cases} \tag{2-12}$$

The form for $\beta = 0$ results as β approaches zero in the limit.

As we have already seen results under the exponential form, suppose $\beta \neq 0$, and differentiate to find

$$\frac{\partial L}{\partial w} = -\gamma L_w^{\;1-\beta} \tag{2-13}$$

Again, we have a formulation in which the marginal product depends on a measure of the effectiveness with which the ecological asset provides the service, γ, and the amount of pollution remaining, L_w. Moreover, the three implications I have noted above follow in this more general formulation: If a little goes a long way, you don't need a lot, there is an upper bound on the value of the marginal meter, and *Catch-22*'s arise under circumstances in which the value of the marginal meter of a wide buffer is not negligible.

The more general form, (2-13), generates a value of the marginal product that spans the range between two extremes. The first extreme arises when $\beta = 1$ and load is reduced by γ kilograms per meter *until all pollution is removed*. Of course, once there is no more pollution to be removed, the marginal meter can contribute nothing more. The other extreme arises in the limit as β decreases without bound. In this case the first iota of riparian buffer would remove all save a speck of pollution, and any additional area would contribute miniscule value.

I have chosen the exponential form in the pollution treatment example, as well as the examples given below, because 1) the natural science literature supports the choice, and it may be derived from first principles; and 2) it generates compact analytical results. The general results I derive, however, do not depend greatly on exactly what functional form one adopts. The important thing is that ecological assets that perform treatments on some exhaustible stock inevitably have the property that the more they can do, the less that will remain to be done.

III. Flood protection

Many authors list the prevention and mitigation of floods as an important ecosystem service (see, e. g., Daily 1997).[9] Areas retained in forests, wetlands, and other natural systems may retain precipitation on or near the surface, conduct it into groundwater, or return it to the atmosphere through evapotranspiration. Water thus retained or diverted will not flood downstream areas. As one document on the *Economic Benefits of Wetlands* states, "Wetlands can play a role in reducing the frequency and intensity of floods by acting as natural buffers, soaking up and storing a significant amount of floodwater" (EPA 2006).

How might we think about the value of areas providing this service? Suppose that the total amount of precipitation over some period is a random variable θ, with probability distribution $f(\theta)$. While I will use the term "precipitation" for brevity, what I really have in mind is "the amount of water in a watershed at a given time," a quantity that might result from a single rainstorm, a series of rainstorms, or the melting of accumulated snowfall. We are interested in the role natural areas – let's call them "wetlands" in the interest of brevity, while acknowledging that forests, grasslands, or other ecosystems might perform a similar function – play in flood protection. Suppose an area of size A is retained in wetlands. One often hears statistics such as that an acre of wetland can retain up to three acre-feet of water cited in support of the flood protection services such wetlands provide (see, e. g., EPA 2006). Of course a wetland is, presumably, usually wet. The more relevant concern is how much *more* retention could such an area provide in the event of heavy rain or snow melt? I note this, as it is one motivation for deriving bounding results. If we do not know exactly what capacity wetlands are providing, we might ask for what capacity their marginal value would be greatest.

I will also suppose that capacity may be supplemented. Means for doing so might include construction of reservoirs or levees, or reinforcement and flood-proofing of structures by, for example, raising them on pilings, as one often sees in beach- or riverfront communities in both wealthy and developing countries. I will treat both alternative reservoir construction and making structures flood resistant as a sort of additional capacity, inasmuch as each would increase the amount of precipitation a community could experience before it would incur damage. Let me denote this additional capacity as K. I will treat it as fixed for now, but will consider the implications of variation in K in my discussion of the *Catch-22* aspect below.

Let us denote the water retention capacity of a hectare of wetland as k, and suppose flood damages are

[9] Another indicator of the salience of flood protection in discourse concerning ecosystem services is that the word "flood" appears four times in the Wikipedia entry on "ecosystem services" (http://en.wikipedia.org/wiki/Ecosystem_services accessed 3 April 2015).

$$D\left(\theta - kA - K\right) \tag{3-1}$$

if the volume of precipitation, θ, exceeds the retention capacity, kA, and zero otherwise. Note that expression (3-1) implicitly defines a "zero" point in the distribution of θ. The level of precipitation for which damage would first be recorded in the absence of any retention capacity would be where $\theta = 0$.[10]

Expected flood damages when an area of size A is set aside are, then

$$E\left(D \mid kA + K\right) = \int_{kA+K}^{\bar{\theta}} D\left(\theta - kA - K\right) f\left(\theta\right) d\theta \ , \tag{3-2}$$

Where $\bar{\theta}$ is the upper limit of the support of θ. We could interpret expression (3-2) as expected damage per unit time, and compute net present values by appropriate discounting.

Differentiating (3-2) with respect to A to find the value of the marginal hectare, and again changing the sign of the resulting derivative to follow the convention that a *reduction* in damages represents a positive value, we have

$$v\left(A\right) = -\frac{\partial E\left(D\right)}{\partial A} = kD\left(0\right) \ + \ k \int_{kA+K}^{\bar{\theta}} D'\left(\theta - kA - K\right) f\left(\theta\right) d\theta \tag{3-3}$$

The first term following the second equal sign in (3-3) is zero, as no damage results when precipitation is less than the retention capacity, and so we may rewrite the value of the marginal hectare of wetland for the flood protection service it provides as

$$v\left(A\right) = kE\left[D'\left(\theta - kA - K\right) \mid \theta \geq kA + K\right]\left[1 - F\left(kA + K\right)\right] \tag{3-4}$$

We thus have the generic three-factor representation of the value of the marginal unit of an ecological asset again. In this case, the value of the marginal hectare of wetland for the flood-protection service it provides consists of: the amount of precipitation retained by a hectare of wetland, k; the conditional expectation of marginal damage given that a flood occurs, $E\left[D'\left(\theta - kA - K\right) \mid \theta \geq kA + K\right]$; and the probability that a flood occurs, $1 - F\left(kA + K\right)$. Let us now turn to the three implications I laid out above, before concluding with some brief observations on value:

[10] It is also worth noting that water retention by upstream ecosystems can have *negative* consequences. For example, Lélé, et al. (2008) note that upstream water retention can reduce the availability of irrigation water.

"If a little goes a long way, you don't need a lot"

The value of the *first* hectare of wetland preserved to provide the flood protection service would just be

$$kE\left[D'(\theta)\,|\,\theta > K\right]\left[1 - F\left(K\right)\right]. \tag{3-5}$$

To proceed further let me make three specific assumptions concerning functional forms and offer justifications for them:

1. *Expected marginal damage in the event of flooding is constant.* This assumption is not, of course, literally true, but I will adopt if for reasons analogous to those presented above for treating marginal damage as constant in the case of pollution treatment.

2. *Flooding is sufficiently unlikely that I can employ the approximation* $1 - F \approx -\ln F$. If large floods causing extensive damage were anticipated on a regular basis, we would have to wonder if people would build expensive structures or establish residence in so dangerous a place. It is worth noting also that the approximation will be reasonably accurate over a fairly wide range; if $F = 0.90$ – there is a ten percent chance of a damaging flood occurring – the approximation yields about a 10.5% chance of flooding.

3. *Precipitation follows a type I* (*Gumbel, or double-exponential*) *probability distribution.* This is a natural assumption to make in this context. The extreme value distribution is the limiting distribution that obtains for the maximum value of a set of random variables – like precipitation or floods. For this reason, it is often employed by empirical researchers studying such phenomena (see, e. g., Yue, *et al*., 1999). Moreover, the precipitation events of concern would likely be those associated with the largest rain- or snowfalls of a year or season: that is, the extreme values.[11]

From the third assumption, let us suppose that

$$F\left(kA\right) \;=\; \exp\left[-\exp\left(-\frac{kA + K - \mu}{\sigma}\right)\right] \tag{3-6}$$

where μ and σ are parameters that may be estimated from data.[12] From the second assumption, we have

[11] Moreover, even should mulitiple floods occur in the same time period, the most severe among them would likely explai most of the damage.

[12] The mean of the Gumbel distribution is $\mu + \gamma\sigma$,. The variance is $\sigma^2\pi/6$. Just to avoid confusion with symbols used elsewhere in the paper, let me note that γ here denotes Euler's constant, approximately 0.5772, and π here means "3.1415 . . ."

$$1-F\left(kA+K\right)\;\approx\;-\ln\left\{\exp\left[-\exp\left(-\frac{kA+K-\mu}{\sigma}\right)\right]\right\}\;=\;\exp\left(-\frac{kA+K-\mu}{\sigma}\right). \qquad (3\text{-}7)$$

From the first assumption, suppose that *D'*(·) is constant at δ. Combining these assumptions, then, I may write the value of the marginal hectare in providing flood protection as

$$v\left(A\right)=k\delta e^{-\left(kA+K-\mu\right)/\sigma} \qquad (3\text{-}8)$$

With these preliminaries out of the way, let us consider next how the value of the marginal hectare varies if we fix the area providing the service, *A*, and perform the thought experiment of varying the retention capacity of each hectare. Differentiating (3-8) with respect to *k*, we have

$$\frac{\partial v}{\partial k}=\left(1-kA\right)\delta e^{-\left(kA+K-\mu\right)/\sigma}\;. \qquad (3\text{-}9)$$

When a large area of wetlands is available, the marginal hectare will be *less* valuable the *more* water it is capable of retaining.

Upper bound

Setting expression (3-9) to zero, we find that an upper bound is reached when

$$\hat{k}=1/A\;. \qquad (3\text{-}10)$$

At this level of the retention parameter, the value of the marginal hectare would reach its upper bound

$$\hat{v}\;=\;\left(\partial/A\right)e^{-\left(\sigma+K-\mu\right)/\sigma}\;, \qquad (3\text{-}11)$$

which I can also write under the assumptions I've proposed above as

$$\hat{v}\;\approx\;\left(\partial/Ae\right)\left[1-F\left(K\right)\right] \qquad (3\text{-}12)$$

So the upper bound on flood damage is marginal damage divided by the product of wetland area and *e*, the base of the natural logarithm, times the probability that enough precipitation would fall to cause flooding in the absence of any wetland retention capacity.

Catch-22

In this section I will again expand the problem as I did in the preceding section to consider a broader objective. One natural way to think of flood damage is as the amount of replacement investment that would be required to again restore the damaged area to its optimal condition. To calculate this value, we might construct a dynamic programming model in which a stock of capital investment – the value of structures at risk from flooding, in this case – might be augmented by new investment and decimated by floods. I will not do this in detail, but rather argue by extension from well-known results.

Suppose that $\pi(X)$ is the value of a cumulative investment of *X*. Let *r* be the opportunity cost of investment, that is, the discount rate, and suppose Δ is the proportional rate of depreciation; a fraction Δ of cumulative investment, *X*, is lost every year. Then it is well known that in a steady-state optimal investment program

$$\pi'(X) = r + \Delta \tag{3-13}$$

Now let us suppose that depreciation is a function of the amount of flooding (we could incorporate nonstochastic depreciation due to normal wear and tear or obsolescence by subsuming it in the discount rate). Then we could express the condition for optimal investment[13] as

$$\pi'(X) = r + E\left[\Delta\left(\theta - kA - K\right)\right] = r + E\left[\Delta\left(\theta - kA - K\right) \mid \theta > kA + K\right]\left[1 - F\left(kA + K\right)\right]. \tag{3-14}$$

Expression (3-14) yields a *Catch-22*. The value of flood protection is greater the greater is the expectation of damages from flooding, but the greater is the expectation of damages from flooding, the less investment would be placed at risk. This is a rather obvious point; the fact that less investment would be made in areas prone to flooding underlies the logic of hedonic exercises for estimating the value of risk reduction (see, e. g., Bin and Polasky 2004; Kousky 2010; Whitehead and Wilson 2011).[14] Still, it suggests a question a researcher might ask herself in conducting such exercises: if she finds very high values for risk reduction, are they consistent with the fact that valuable properties have been constructed in a high-risk area? I might note that, for reasons I will consider momentarily, they may well be, but this does not obviate the utility of posing the question.

We might also consider the implications of a different kind of investment. I have incorporated an expression for alternative retention capacity, *K*, in my expressions above. For the sake of clarity let us return to the simple formulation in which investment at risk is fixed, and consider the minimization of the sum of expected flood damage and cost of alternative retention investment,

$$\int_{kA+K}^{\bar{\theta}} D\left(\theta - kA - K(c)\right) f(\theta)\, d\theta + c \tag{3-15}$$

differentiating (3-15) with respect to investments made in alternative retention capacity, *c,* we have:

$$-D(0)\, f\left[kA + K(c)\right] K'(c) - K'(c) \int_{kA+K(c)}^{\bar{\theta}} \frac{\partial D}{\partial \theta} f(\theta)\, d\theta + 1 = 0 \ . \tag{3-16}$$

Since there is no flood damage when there is no flooding, the first term on the left-hand side of (3-16) is zero, and comparing (3-16) with (3-3) and (3-4), we have

13 I am implicitly assuming that the optimal response to flood damage is what is known in the optimal control literature as a "bang-bang" solution: immediately rebuild what was lost. For present purposes this does not seem unreasonable, though it is one of many assumptions that might be varied in a more complete treatment.

14 I thank Edward Barbier for pointing out the flaws in this section in an earlier version of this paper, as well as providing these references.

$$v(A) = \frac{k}{K'(c)} \tag{3-17}$$

Equation (3-17) is a simple arbitrage condition: the flood protection value afforded by the marginal hectare in retaining its *k* liters should be the same as the cost of purchasing the same *k* liters of retention capacity by making alternative investments.[15]

If we differentiate (3-17) totally with respect to the retention parameter, we find

$$\frac{\partial v}{\partial k} = \frac{K'(c) - kK''(c)\,dc/dk}{\left[K'(c)\right]^2}\ , \tag{3-18}$$

or, using (3-17),

$$\frac{dc}{dk} = \left(1 - \frac{\partial v/v}{\partial k/k}\right) \Big/ vK''. \tag{3-19}$$

From (3-8) and (3-9), $\left(\partial v/v\right)/\left(\partial k/k\right) = 1 - kA$, and so we can restate (3-19) as

$$\frac{dc}{dk} = kA/vK'' < 0, \tag{3-20}$$

where the inequality holds under the assumption that there are diminishing returns to investment in retention capacity.

The intuition underlying (3-20) is straightforward. Investments in alternative retention capacity are substitutes for retained wetlands. Arbitrage implies that the value of wetlands would optimally be equal to the marginal cost of alternative retention capacity. We have seen that, when wetlands are relatively abundant, the flood-protection value afforded by the marginal hectare of wetland will be *higher* the *less* precipitation that hectare can retain. Yet the less capacity a hectare of wetland has for retaining precipitation, the *more* attractive is investment in alternative retention capacity. And of course, the more investment is made in alternative retention capacity, the less valuable is the marginal hectare of wetland. Again, we have a *Catch-22*: the conditions under which the value afforded by a marginal hectare of wetland would be most valuable might also be those under which dependence on natural ecosystems would be obviated by investment in alternative measures.

The value of flood protection services

While there are certainly limits on the value of flood-protection services performed by natural ecosystems, it remains an open question as to how valuable such services might be in practice. There are reasons to suppose that they might afford a more compelling argument for conservation than do

[15] I am grateful to Steve Newbold for discussions clarifying this point.

some other ecosystem services. Ecosystems providing flood protection services are an example of a sort of action-at-a-distance that does not characterize all ecosystem services. In our first example, of pollution treatment by riparian buffers, only buffers located adjacent to areas from which pollution is released are likely to be of any appreciable value. With flood protection, however, water retained thousands of kilometers upstream of vulnerable cities or fields might afford them some protection against flooding. Moreover, such floodwater storage may protect a whole string of downstream assets. In calculating expected marginal damages, one would want to include all of the place that might be affected. Water retained on one hectare upstream might prevent a flow that could endanger a succession of hectares downstream.

As I noted above in deriving expression (3-14), it begs a question: if more profitable investment might be undertaken elsewhere, why place structures and people in harm's way by building in a flood plain? This is a question worth considering as a way to check the plausibility of value estimates, but there may very well be good answers to it, such as:

- Some areas – the confluence of rivers comes to mind – have advantages of location that justify a concentration of investment and population because they generate substantial scarcity rents.

- Investments are not necessarily made under the same conditions or expectations that will later prevail. If natural infrastructure might protect the centuries-old *palazzi* of Venice or *ghats* of Varanasi against the risk of destruction, retaining it should certainly be carefully considered. Similarly, anyone who has waded the knee-deep streets of Mexico City or Delhi in the aftermath of a spring cloudburst will appreciate that those cities' urban infrastructures were not installed with perfect foresight for the hydrological consequences of rapid urbanization. Such differences in conditions between the times investments were made and floods occur might become more pronounced if climate change leads to more precipitation in some heavily populated areas.

- Some people "choose" to occupy dangerous floodplains because they have no better choices. As is always the case in economic valuation, economic "values" may not always reflect social or moral priorities.

These observations underscore the need for considerably more research. I am only trying to suggest some standards we might apply to determine when estimates of value are plausible. As this example shows, these "reality checks" do not necessarily preclude the possibility that some of those values could be cited in defense of greater conservation of natural habitats.

IV. Pollination

Another ecosystem service that has garnered considerable attention is pollination (see, e. g., ESA, no date). A number of different types of animals pollinate plants. That is, they collect pollen from male flowers or parts of flowers and transfer it to the female counterparts, where the sperm in the pollen can fertilize an egg cell and form a seed. This service is economically valuable when the seed itself becomes a marketable product, such as almonds, or more commonly, when a marketable fruit forms around the seed or seeds, such as apples or melons. While many commercially important crops are fertilized by rented colonies of European honeybees, much of the emphasis on pollination as an ecosystem service stems from the observation that native animals may also perform similar services (see, e. g., Ricketts, *et al*., 2007). The value of this service might, then, be cited as a motivation for preserving landscapes that promote the survival of native pollinator populations (Ricketts, et al., 2004), as well as restricting use of pesticides and limiting other threats to native animal populations.

Let me outline a schematic model of pollination. The ecological asset whose value is under consideration here is a population animals, which I will refer to as "bees," though a variety of different types of animals may perform this service. It is worth underscoring that the asset under consideration here is different than those considered in the previous two examples. A population of pollinators would itself depend on the availability of other assets, such as suitable habitat for their propagation, nutrition, and protection. For this reason, valuing the "marginal pollinator" would be an incomplete exercise for informing conservation policy. We would also want to know the how much the "marginal hectare" of land retained in natural cover would contribute to the population of pollinators.[16] Be that as it may, however, we would still need to know the value of the marginal pollinator to compute the value of the marginal hectare of land supporting it, so let us focus now on valuing the marginal bee.

Suppose there is a population of B bees, each of which may visit and pollinate ϕ flowers. Let us suppose that a flower, if pollinated, will produce a fruit, which can be sold at a price of p. We can consider this as a net price, after subtracting costs of cultivating, harvesting, and marketing the portion of the crop that comes to fruition. Suppose farmers plant a crop of size Φ, where we can think of Φ as the number of flowers that might potentially be fertilized and hence produce marketable products. The cost of planting Φ flowers is $c(\Phi)$. If bees visit flowers at random, the probability that any bee will visit a particular flower is ϕ/Φ, and the probability that any particular flower will *not* be visited, and hence not pollinated, by *any* bee is

$$\left(1-\phi/\Phi\right)^{B} \tag{4-1}$$

Thus, the probability that a flower *is* visited and pollinated is

[16] There is a literature on the foraging behavior of pollinators that can be used to relate land use to pollinator populations (see, e. g., InVEST 2015, and the literature cited there). Another concern for the valuation of ecosystem services may arise I this context, however. If such foraging ranges are large enough, there may be little reason to preserve habitat in close proximity to farm fields. Conversely, if foraging ranges are small enough, there may not be a public policy problem, in that farmers would find it in their own interest to maintain habitat for the pollinators on which they depend.

$$1 - (1 - \phi/\Phi)^B \tag{4-2}$$

The expected profits of a farmer who chooses to plant *Φ* flowers when *B* bees are available to service her crop are

$$p\left[1 - (1 - \phi/\Phi)^B\right]\Phi - c(\Phi) . \tag{4-3}$$

We can write

$$(1 - \phi/\Phi)^B = \left[\left(\frac{\Phi/\phi - 1}{\Phi/\phi}\right)^{\Phi/\phi}\right]^{\phi B/\Phi} \tag{4-4}$$

I have noted the expansion in (4-4), as $\lim_{x\to\infty}\left(\frac{x-1}{x}\right)^x = e$, and so, if ϕ – the number of flowers that may be visited by any single bee – is small relative to *Φ* – the total number of flowers that might be pollinated –we can employ the approximation

$$(1 - \varphi/\Phi)^B \approx e^{-\varphi B/\Phi} \; ; \tag{4-5}$$

of course, if the number of flowers that may be visited by any single bee were *not* small relative to *Φ*, and there were more than a few bees, the marginal bee would not be scarce relative to the demand for its services, and so its value would be negligible.

Thus, to what is likely to be a very close approximation (close enough that I will use a simple "=" sign, rather than the approximation sign "≈" henceforth), the farmer's profit function is

$$\pi(B, \Phi) = p\left[1 - e^{-\phi B/\Phi}\right]\Phi - c(\Phi) \tag{4-6}$$

Differentiating with respect to *B* to find the value of the marginal bee,

$$\frac{\partial\pi}{\partial B} = v(B) = p\phi e^{-\varphi B/\Phi} \tag{4-7}$$

The "treatment" interpretation is apparent in expression (4-7). The value of the marginal bee is the price of the fruit that may result should it pollinate a flower, *p*; times the effectiveness with which it pollinates flowers, which is ϕ, the number of flowers it visits; all times the probability that each of the flowers it visits would not be pollinated by any of the other *B* bees.

"If a little bit goes along way, you don't need a lot"

Turning now to the three implications of the "treatment" interpretation, note that the marginal value of the *first* bee[17] would be, from (4-7),

$$v(0) \;=\; p\phi \;: \tag{4-8}$$

the more flowers bees can visit when bees are scarce, the more valuable they would be. Fixing the number of bees available to serve a crop of a certain size and doing the thought experiment of varying the number of flowers any one bee could visit, we have

$$\frac{\partial v}{\partial \phi} \;=\; pe^{-\phi B/\Phi}\left(1-\frac{\phi B}{\Phi}\right) \tag{4-9}$$

For a large enough value of *B*, the larger is ϕ relative to the size of the crop requiring fertilization, Φ, the less valuable is the marginal bee. So, when the *first* bee is very valuable, the B^{th} may not be.

One suspects that this point may not have been fully appreciated by some conservation advocates. For example, the Ecological Society of America (ESA, no date) publishes a "pollination fact sheet" that states "a single southeastern blueberry bee pollinates approximately $75 worth of berries by visiting nearly 50,000 blueberry flowers in a year." If bees are, in fact, prodigious pollinators, it might well be that the marginal product provided by any one bee would be negligible.

Upper bound

We can find a upper bound on the value of the marginal bee by setting expression (4-9) to zero, which would occur if

$$\hat{\phi} \;=\; \frac{\Phi}{B} \tag{4-10}$$

Substituting from (4-10) into (4-9), the upper bound on the value of the marginal bee would be

$$\hat{v}(B) \;=\; \frac{p\Phi}{Be} \tag{4-11}$$

Expression (4-11) has a very straightforward interpretation: $p\Phi$ would be the gross value of the crop if all flowers were fertilized, so the upper bound on the value of the marginal bee would be this gross value divided by the product of the population of bees and *e*, the base of the natural logarithm.

Catch-22

[17] The reader may reasonably object that bees are social insects, and so positing the contribution of a single bee is not a very credible thought experiment. More generally, however, we can think of pollinators as being more or less scarce. Expression (4-8) is simply noting that when pollinators are relatively scarce, the marginal pollinator will be more valuable the more flowers they can visit.

The *Catch-22* aspect can be illustrated very clearly in this context. Define

$$L(\varphi, \Phi, B) = e^{-\varphi B/\Phi} \tag{4-12}$$

as the fraction of the crop *lost* for want of pollination services. Note, then, from expression (4-7), that the value of the marginal bee can only be substantial if farmers anticipate some nonneglible loss of output for want of pollination. Thus an alternative restatement of (4-7) is

$$v(B) = p\phi L \tag{4-13}$$

It is easy to show – and implicit in the derivation of $\hat{\phi}$ in expression (4-10) above – that the value of the marginal bee would be greatest when a fraction 1/*e* = 0.368 of the crop were lost for want of pollination. One has to wonder how many crops would be grown if farmers expected losses of such a magnitude, or if farmers would not seek a more advantageous location if they experienced such crop losses for want of pollination.

To put this another way, suppose a farmer takes the number of bees available to pollinate her crop as given, and chooses what size of crop to plant. That is, suppose she maximizes her profit, (4-6), by choice of crop size, Φ. She would then choose Φ so that

$$\frac{\partial \pi}{\partial \Phi} = p\left[1 - e^{-\phi B/\Phi}\right] - c'(\Phi) - p\frac{\phi B}{\Phi}e^{-\phi B/\Phi} = 0 \tag{4-14}$$

Using (4-12), this is equivalent to

$$p(1 - L) - c'(\Phi) + pL\ln L = 0 \tag{4-15}$$

or

$$\frac{p - c'(\Phi)}{p} = L(1-\ln L) \tag{4-16}$$

The right-hand side of (4-16) approaches zero in the limit as *L*, the fraction of the crop lost for want of pollination, vanishes, and increases toward one as *L* nears one. Recall from (4-13) that the value of the marginal bee *increases* in the fraction of the crop lost for want of pollination. As just noted, the expression on the right-hand side of (4-16) also increases in the fraction of the crop lost.

We might interpret this result by noting that left-hand side of (4-16) is known in the industrial economics literature as the Lerner index. It is a measure of market power: how much price can be marked up over marginal cost. Expression (4-16) is telling us that the value of the marginal bee is related to the scarcity rents that arise in agricultural production from access to pollinators that are in short supply. If pollinators remain abundant in other areas, they can only be valuable in those areas in which they are in short supply if other advantages of location prevent the relocation of production.

The value of pollination services

While the expressions above define conditions under which pollination services could be of substantial value, they also suggest that such conditions might only be encountered rarely. In order for pollination services to be valuable, there would have to be nontrivial expected crop losses for want of pollination, as well as substantial rents accruing to producers in areas in which pollinators were scarce. Yet large areas remain around the world in which natural habitats remain largely intact. Instead of reestablishing habitats to support pollinators where they have become scarce, farmers might clear small fields amid forests in which pollinators remain abundant. The ecologist Jaboury Ghazoul (2005) has made this argument in noting that many crops requiring native pollinators are, in fact, grown in areas of the world in which pollinators remain abundant.[18]

The model of pollination I have sketched above might be considered deficient in that it does not address the contributions of pollinators in stochastic environments. One argument often made for preserving native pollinator populations even when rented honeybees might also be employed is that the natives may provide pollination services when honeybees cannot; for example, Brittain, *et al*., (2012) find that natives may be able to pollinate crops when honeybees are virtually grounded by high winds. On reflection, however, such observations only underscore the points I have brought out above. First, the marginal native pollinator is only valuable when losses are expected to be high (e. g., when high winds are likely). Second, the fundamental diversification principle of portfolio allocation applies: an asset – in this case, native pollinators – is more valuable to the extent that its contribution is negatively correlated with the performance of the "portfolio" as a whole – all pollinators. This cannot occur if the number of natives is high relative to that of other potential pollinators.

While it may be difficult to identify scenarios in which the ecosystem services provided by native pollinators are likely to contribute large marginal values, the example of pollination shows how arguments concerning ecosystem services may sometimes proxy for broader concerns. As I write, new reports are circulating concerning the severity of Colony Collapse Disorder (CCD), a phenomenon in which commercial honeybee colonies die out or disappear (ARS 2015). The large-scale disappearance of commercial pollinators would, of course, be a calamity. Different modes of analysis may be appropriate for considering the value of such consequences (see, e. g., Bauer and Sue Wing 2011). The only thing I would add here is that the local land use decisions that have often motivated concern with valuation of pollination services (e. g., Ricketts, *et al*., 2004) are probably not the key consideration in thinking about the survival of commercial pollinators.

[18] My own interest in pollination was spurred by a similar observation. I attended a meeting on the Caribbean island of Trinidad, where my hosts noted that pollinators were becoming scarce in the vicinity of Port of Spain, the largest city on the island. There are lush forests with a few dozen kilometers of the city, however, and I noticed during a sight-seeing trip along the island's backroads that small, scattered, agricultural fields had been carved from the dense forest in places. This would seem to be the logical development if pollinators are essential to growing valuable crops and pollinators are becoming scarce in densely settled areas, but they remain abundant elsewhere.

V. Other examples

The general three-factor structure I have proposed and illustrated with the above examples might be applied to the provision of a number of ecosystem services. Let me briefly consider several other examples.

In earlier work, two colleagues and I demonstrated how an upper bound may be placed on the value of the "marginal species" for its use in the search for new pharmaceutical products (Simpson, Sedjo, and Reid, 1996). We derived a three-factor formula for value that consisted of the probability that the marginal species would yield the product for which search was being conducted, the payoff in the event the product was developed, and the probability that the product would *not* be developed from any of the other species available to serve as research leads. Arguments analogous to those I made in Section IV on pollination led to an exponential form for the probability of discovery, and we developed similar bounding results. One might also wonder if the *Catch-22* implication has also played out in practice. Enthusiasm for "bioprospecting" seems to have waned considerably since the 1990's (see, e. g., Firn 2003), and one reason may be that the probability of finding useful products was low enough as to make synthetic chemistry or other approaches seem more profitable choices.

A couple of the examples I gave above could be easily adapted to slightly different circumstances. While I have focused on the flood protection service provided by upstream wetlands, an entirely analogous approach could be taken to the valuation of coastal ecosystems. Storms striking the coast can be characterized by the probability distributions of their wind speeds or storm surges. So, deeper coastal defenses, like larger upstream wetlands, would provide greater protection against less likely events, and so we might expect a similar pattern of diminishing returns.

It has been suggested that tree foliage may provide a service for the filtration and separation of airborne pollutants in a fashion analogous to the way ground-level vegetation traps water-borne pollutants (see, e. g., Nowak, et al., 2006). The same general principle applies in both cases: whatever pollution is treated by assets already in place would not remain to be treated by the marginal asset. Similar conclusions might also be drawn with respect to carbon sequestration through re- and afforestation, although this case might be the exception that proves the rule: the stock of carbon to be removed from the atmosphere may be so large relative to our means for removing it that, regrettably, diminishing returns would be unlikely soon to be evident.

Natural pest control would be another example. The larger the population of predators that is maintained in the vicinity of crops, the less likely it would be that any pest that would otherwise eat the crop would itself escape being eaten. The three-factor formulation would, then, be that the value of the marginal predator would be the value of the amount of the crop that a pest might eat times the number of pests a predator might consume, all times the likelihood that the pests the predator consumes would not have been eaten by any other predator.

One might well continue with such examples virtually *ad infinitum*.[19] The message of this paper is that a common underlying structure may be identified among them, and this structure can be used to focus and bound results concerning their values.

[19] I might also note in this context that exponential models isomorphic to those I have developed above have also been employed in the ecology literature to demonstrate implications of species and functional diversity; see, e. g.,

VI. Conclusion

I have sketched some simple models and shown how each gives rise to three important implications. How might these implications be exploited to inform policy choices? There are many potential applications:

- The models of ecological production functions I have sketched are, in some instances, somewhat scaled-down versions of more elaborate models already in the public domain. For example, the Natural Capital Project has a suite of "InVEST" (Integrated Valuation of Ecosystem Services and Tradeoffs; 2015) models which provide very detailed treatments of a variety of ecosystem services. Similarly, the United States Environmental Protection Agency's Office of Research and Development is assembling a "library" of ecological production functions for use in applied research (Bruins, et al., 2012). My results might be used as a sort of "reality check" on analyses done with them.

- My results also suggest scoping studies that might be conducted to determine when it is worth the effort to construct more elaborate and detailed models. If a simple model strongly suggests that the upper bound on value would not be large, there may be little point in putting in the extra expense and effort to find a more precise estimate of a small value.

- "Benefit transfer" exercises have often been conducted in which results estimated at one time and place are applied to another. My results can be useful for evaluating such benefits transfers in two ways. First, I identify "sufficient statistics" for valuation in the sense of Chetty (2009): what information would we need to have in order to construct value estimates? Second, my bounding results could be applied to determine if transferred estimates are plausible.

- There are alternative ways to estimate values. One of the purported advantages of hedonic valuation models, for example, is that the researcher may simply establish a statistical relationship between the amounts of ecological assets available and the value of properties thought to benefit from them, without specifying explicitly how she believes them to be related. Of course, such exercises always raise the question of whether the relationship found represents real causation or only spurious correlation. My results could provide a way to distinguish between these possibilities.

- The *Catch-22* results I sketched point to the importance of some other considerations in doing ecosystem service valuation. The value of ecosystem services depends on things such as the amount of pollution to be treated, the size of a crop to be pollinated, or the value of structures at risk from flooding. Each of these quantities depends in turn on economic choices that might also inform the analysis. Analyses of the types I have sketched above might be conducted to

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Tilman, *et al*., 1997 and Naeem 1998. Another interesting analogy is to the "reserve site selection problem" (see, e. g., Pressey, *et al*., 1993; Ando *et al*., 1998). This is the problem of selecting a set of nature reserves so as to maximize the probability of endangered species survival. In such a context the value of the "marginal reserve site" in preserving a particular species is proportional to the probability it *will* protect the species times the probability that all others will *not*.

determine which investment and location decisions might also be modeled to better inform valuation estimates.

My goal in this paper is both modest in some respects and ambitious in others. It is modest, in that the models I have developed and presented here are simple and schematic, and I have not made any attempt to reduce them to practice by calibrating expressions with data.[20] It is ambitious in that I propose a generic framework for the valuation of many types of ecosystem services. Perhaps in this regard I have put the cart before the horse. A paper of this type might be more useful if it constituted a survey of completed work than as a prospectus for an extensive research agenda. My point in composing the paper at this stage, however, is to suggest that such a research agenda has great promise.

[20] In earlier work (Simpson, *et al*., 1996) my coauthors and I performed such calibration exercises in the analogous context of bounding the value of native biodiversity for its use in new product research. That work might be regarded as a proof-of-concept exercise for the other examples I have given here.

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