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Analysis

Stream ecosystem service markets under no-net-loss regulation

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ABSTRACT

We analyze interactions between economics and ecology for ecosystem service markets under no-net-loss regulation. Previous studies of no-net-loss regulation address the ecological efficacy and valuation of restoration but largely ignore the effects of market dynamics. We link an economic model of free-entry equilibria with an ecological model that includes returns to scale and inefficiency of restored ecosystems and apply the result to stream mitigation banking in North Carolina. Intuition from ecology alone must be modified to account for economic processes, and vice versa. To implement no-net-loss regulation, one must not only account for ecological differences between restored and natural ecosystems, but also consider the effect of market entry on the number and size of restoration projects. In a purely economic model, free-entry equilibria are characterized by excess entry: the equilibrium number of firms is greater than the welfare maximizing number. Ecological considerations may exacerbate or ameliorate this, so that either excess entry or insufficient entry may occur, depending on the specific ecosystem services sought.

1. Introduction

Restoration of stream ecosystems is proliferating in the US and worldwide (Bernhardt et al., 2005), resulting in increased interest of both the underlying science (Wohl et al., 2005) and economics (Collins et al., 2005). To date the driver of restoration has often been site-specific needs (e.g., bio-engineered bank stabilization near infrastructure) or agency-specific programs (e.g., endangered species habitat recovery). However, restoration is now often being implemented under the auspices of Clean Water Act compensatory mitigation, in which impacts to streams (i.e., degradation) must be compensated by restoration of streams elsewhere (reviewed by Lave et al., 2008). This regulatory approach has, in essence, created a market for restored streams.

More generally, market mechanisms are increasingly used by policy makers for achieving the goals of a variety of environmental protection laws. Such environmental markets span water quantity for in-stream flows (Chong and Sunding, 2006), water quality (Woodward et al., 2002), air quality (Burtraw et al., 2005), wetlands (NRC, 2001), and endangered species habitat (Bonnie, 1999). The scope of these markets is considerable, and growing: the global carbon market exceeded $30 billion in 2006 (Capoor and Ambrosi, 2007), and within the US, Clean Water Act driven no-net-loss compliance has been approximated as a $2.9 billion business (ELI, 2007). Economic activity in environmental markets is only likely to increase as is the adoption of market-based approaches to streams and other environmental amenities.

A critical attribute of environmental markets is that ecological and economic processes may be linked in complicated ways. If ignored or unknown, such linkages can affect the potential success of markets in fulfilling environmental goals. For example, conservation programs often seek to preserve biodiversity by purchasing and protecting parcels of land with high biological value. As Armsworth et al. (2006) demonstrate, however, the interaction between the local market for land and conservation purchases may actually lead to a decrease in overall biodiversity. An important research agenda then needs to be joint economic and ecological studies, particularly theoretical studies that can identify linkages that may lead to unintended or non-intuitive consequences.

In contrast to the vast literature on pollution markets, there has been less research on markets in ecosystem services (Robertson, 2006) — despite the rapid expansion of such markets throughout the US. The extant literature generally falls into two areas. First, a number of studies have analyzed the problem of determining the value of ecosystem services (Holmes et al., 2004; Kuman and Kumar, 2008; Boyd and Banzhaf, 2007; Yang et al., 2008). Second, several studies have focused on the economics of site-specific ecosystem production decisions (Fernandez and Karp, 1998; Hallwood, 2007). In this paper, we develop an analytical approach from which to analyze an entire ecosystem service market. The value of ecosystem services and the economics of ecosystem production are components of our model, but not the main focus of the analysis.

In typical pollution market applications, the model is based on a single variable defined with respect to a metric of weight or volume (e.g., SO_2 or NO_x markets, Burtraw et al., 2005). It is not immediately obvious that a similar procedure can be used for ecosystem service

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markets, as one might argue that an ecosystem provides a variety of services that cannot easily be captured by a single variable. But it is not uncommon for analysts to simplify complex ecosystem characteristics or processes into single variables to facilitate modeling (e.g., Brock and Carpenter, 2007). Further, policies and regulations are widespread in which ecosystem services are commodified and traded, and the various services are typically bundled into a single unit represented by size (e.g., area or length). We therefore adopt as a starting point a simple ecological model of ecosystem function based on size, fully cognizant that this is a strong simplifying assumption.

We analyze ecosystem service markets in the context of no-net-loss regulation. We first consider a generic ecosystem market and then apply our approach to the specific example of stream mitigation banking. The oldest no-net-loss (NNL) regulatory program is the compensatory mitigation policy for NNL of wetlands. Under this policy, wetlands filled or damaged as part of development activities must be mitigated by the creation or restoration of wetlands elsewhere. The initial intent of this regulation was to ensure NNL of ecosystem function (NNLF), where function was intended to be the combination of a bundle of ecosystem services such as nutrient retention, flood attenuation, and wildlife habitat. However, because of the difficulty in measuring the loss or restoration of functions at specific wetlands sites, area was deemed a reasonable surrogate for function (MOA, 1990). This established NNL of size (NNLS, with area being the particular measure of size used for wetlands) as the primary implementation mechanism for wetland mitigation. Compensatory mitigation of stream ecosystems was subsequently developed, creating stream ecosystem service markets under NNL regulation, and regulators adopted similar approaches in that they established the goal of NNLF, but implement the regulation via NNLS (with length being the measure of size.).

In the early period of NNL regulation, developers tended to construct their own ecosystem mitigation sites. Since 1991, regulators and entrepreneurs have collaborated by creating regional credit markets in the US, hoping that market-based incentives will lower the cost of regulation (Robertson, 2006). In ecosystem mitigation banking, an ecosystem is created or restored by a for-profit company, and credits are assigned to the site by regulators (Corps of Engineers). These credits are then cataloged in the bank and developers may buy them to fulfill the conditions of the federal or state permit that allows them to damage ecosystems elsewhere. Thus there is a market between mitigators and developers. It is this market that forms the basis for our study.

The basic building blocks of our model are a standard economic model and a simple ecological model. The economic model is based on several observations about the features of the existing stream mitigation market. First, there are few institutional or ecological limits on the number of mitigators allowed to enter a given market. The markets are primarily regulated by and spatially constrained to one of the 38 districts of the Corps of Engineers; many of these overlap with state boundaries. In many states or districts, the market is statewide. In others, trades are limited to sub-watersheds within the Corps district. For example, in North Carolina, trades are constrained to 8-digit HUC watersheds, of which there are 54 in the state with an approximate area of 2500 km^2 each. As practiced, the effective geographic service area for trading is sufficiently large, containing hundreds to thousands of km of streams, giving ample opportunity for mitigators to select a project location. In North Carolina, the average geographic service area contains almost 2300 km of streams. This vast length of streams means that the availability of streams to restore is not a limitation for market entry.

Second, to enter the market, a mitigator must invest significant upfront time and effort to learn technical and bureaucratic requirements associated with this rather complex industry. Scientific and technical expertise must be acquired through staffing of hydrologists and ecologists, not to mention the expertise needed in real estate and legal issues associated with riparian land transactions that underlie the mitigation market.

It is standard in economic theory for a market with these features to be modelled using free-entry equilibria. The term "free-entry" deserves some explanation. It means that there are no restrictions on entry. Any potential mitigator may enter the market. It does not mean, however, that there are no costs associated with entry. A market entrant must pay fixed costs (which correspond to the upfront costs described above.) Once mitigators have entered, they compete with each other in the market. The key variable in a free-entry model is the number of mitigators that elect to enter the market and provide restored ecosystems. This variable is determined by the conditions of a free-entry equilibrium: the number of mitigators is such that each mitigator earns zero economic profit. Free-entry equilibria are typically characterized by excess entry. When a mitigator decides whether to enter, it considers its own profit, but it does not consider the effect that its entry will have on the output of the other mitigators. So more mitigators will enter the market than is desirable from the point of view of society.

The ecological model relates ecosystem function to the size of the ecosystem, and compares the ecosystem function of the restored system to the lost system. The functions often sought in restored ecosystems are both hydrological (e.g., flood attenuation, reduced sediment loads) and ecological (nutrient retention, habitat provision). Our model is based on two observations about the function of restored and natural (i.e., existing or undisturbed) ecosystems. First, restored ecosystems are inefficient compared to natural ecosystems. Consider a natural ecosystem and a restored ecosystem of equal size. The former provides at least as much ecosystem function as the latter (NRC, 2001). We allow for several possible reasons for this inefficiency. Second, the relationship between ecosystem function and ecosystem size is often non-linear, and this relationship will vary with specific ecosystem function. For example, the ability of streams to retain nutrients or provide habitat is a non-linear function of the length of the stream in restored or natural conditions (Carleton et al., 2001; Cedfeldt et al., 2000; Vaughn and Taylor, 1999; Doyle et al., 2003). Likewise, wildlife habitat for large mega-fauna exhibits a non-linear relationship to size. In this case, many mitigators creating many small restoration sites is generally inferior to a small number of mitigators creating a few large sites. Based on these observations, we use a power function to relate ecosystem functions to ecosystem size. If the size of an ecosystem doubles, then the ecosystem function may double, more than double, or less than double, depending on the value for the exponent in the power function.

A basic overview of our economic–ecologic model is shown in Fig. 1. Excess entry is a feature of free-entry economic models. We use the economic–ecological model to determine whether ecological considerations make things better or worse. In the top part of the figure, there are two mitigators and three developers. The mitigators generate the supply of ecosystem credits and this supply is matched to the demand from developers who must purchase credits from the market when they damage ecosystems. In the lower part of the figure, an additional mitigator has entered the market. As we shall see, this means that the size of restoration by each mitigator decreases, but the total size of all mitigations increases. So there is an increase in the total credits generated by the market and this in turn implies that the total size damaged by development increases as well. These changes in the market lead to changes in net ecological function. The effect on ecosystems damaged is straightforward — total size damaged by development increases, to the detriment of ecosystem function. The effect on restored ecosystems is more complicated and depends critically on the exponent in the power function that relates size to ecological function. Total size of restored ecosystems increases, but the size of each individual restoration decreases. If the exponent is large, then total ecosystem function of restored ecosystems will decrease. In this case ecological considerations exacerbate the economic problem of excess entry. If the exponent is small, however, then

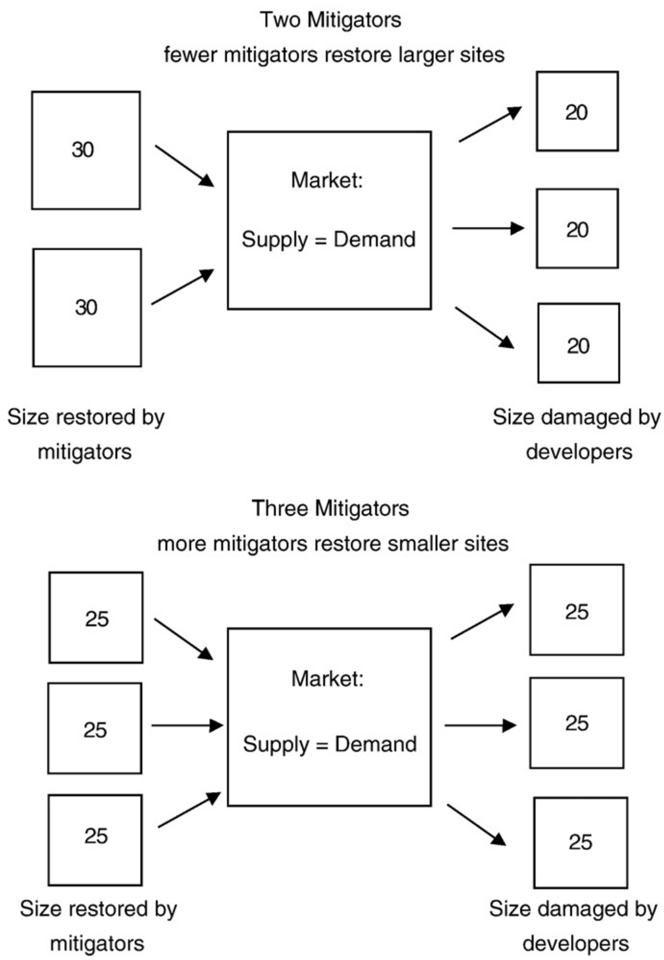

Fig. 1. Market equilibrium and ecological effect of an increase in number of mitigators.

total ecosystem function of restored ecosystems will increase. In this case, ecological considerations have an ambiguous effect and may even lead to insufficient entry.

Our model applies to both NNLS and NNLF regulation. Under NNLF, the regulator uses an economic instrument to ensure that the change in net ecosystem function is zero. We use a trading ratio as the economic instrument. (A trading ratio of two, for example, implies that if a developer impacts 100 m of streams then they must purchase 200 m of restoration credits.) As we explain in detail below, it is often not sufficient to simply set the trading ratio in a manner that corrects for the inefficiency of restored ecosystems. The regulator must also account for the effects of entry on net ecosystem function. Thus the regulator needs a lot of information about the economic–ecologic system to select the proper trading ratio.

We illustrate these issues by conducting an analysis of stream trading in North Carolina. We show that under NNLS there is excess entry. Under NNLF, there may be either excess entry or insufficient entry. We also determine values for the trading ratio to satisfy NNLF and show how these values vary as a function of the underlying ecological parameters.

2. Model components and structure

There are three types of market participants, developers, mitigators, and a regulator. Developers degrade natural ecosystems and mitigators convert land into restored ecosystems. The regulator implements either NNLS or NNLF. To keep the model simple, we assume economic homogeneity: All the developers are the same (they have an identical demand for stream credits) and all the mitigators are the same (they have identical technology and costs). We allow, however, ecological heterogeneity through varying quality of ecosystems, as described below.

In our model, the number of developers, m, is fixed. We assume that each developer, as part of their development activities, degrades a stand-alone ecosystem, each of size ℓ. Development activities often lead to unavoidable impacts to streams or wetlands. However, the quantity of impacts for a given development can be changed with modifications to the development design, layout, or location of land. As the price of wetland and/or stream credits rises, developers tend to preserve those areas (usually as park areas), and adapt the footprint of their neighborhood designs. The demand curve for ecosystem use $p(\ell)$ summarizes this relationship between price and quantity of ecosystem degraded.

In our application, we will use length for the size unit of trade, which applies to streams, although the unit could just as easily be area, as in the case of wetlands or endangered species habitat. The number of mitigators in the market, n, is determined within the model, but we assume at least one mitigator enters the market, so that $n \geq 1$. (We also assume the welfare maximizing number of mitigators is in the interior of this region, so that the maximum can be described by first-order conditions.) Each mitigator that decides to enter must pay fixed cost F. We assume that the production from each mitigator is a stand-alone ecosystem of size q.

We explicitly sought to develop a modeling approach that allows for considering both NNLS and NNLF. Following the commonly employed practice (Brown and Lant, 1999), the regulator uses a trading ratio t for NNLF. Trading ratio's are also often used in pollution control regulation (e.g., Horan and Shortle, 2005). A developer that wants to degrade an ecosystem of size ℓ must purchase $t\ell$ units of restored ecosystem. Under NNLS, the trading ratio is simply equal to one.

The behavior of the mitigators is summarized by the quantity q of ecosystems that they produce. Mitigators select q such that their profit is maximized. Profit is determined by production costs C and the market price P. In turn, the market price is determined by the number of mitigators in the market and the trading ratio. We do not need to specify the exact details of how the price is determined, but simply assume that mitigators account for the effects that their actions will have on the market, given the actions of their rivals. Given that there are n mitigators in the market and the trading ratio is t, the profit maximizing output for each mitigator is $q(n,t)$, produced at a cost of $C(q(n,t))$. In other words, as the number of mitigators or the trading ratio changes, mitigators adjust the size of the restoration projects. In actual practice, there are several ways mitigators can adjust their production of mitigation credits. Many mitigators have options on several parcels of riparian land that allows them to restore streams, or to release the land back to the previous owners at various points in time. As well, mitigators with large sites already restored may divert parts of currently owned land to alternative uses; e.g., from restored wetlands that have not yet been sold to agricultural production. Bankers have arranged their legal instruments to allow this reversion to occur.

A given developer is not matched with a particular mitigator, but rather simply purchases the desired quantity of ecosystems from the market. That is, ecosystems are treated as a divisible good in the market. It follows that each side of the market can be characterized by a simple aggregation of ℓ and $q(n)$. For the developers we have a market demand curve $P(L,t)$ where $L = m\ell$ is the total quantity of ecosystems demanded. For the mitigators we have total production $Q(n,t) = nq(n,t)$. The two sides of the market are linked through a "supply equals demand" relation: $Q(n,t) = tL = tm\ell$. For streams, this implies that the total length of streams mitigated multiplied by the trading ration is equal to the length of stream impacted by developers. For example, if $t = 2$ and developers degrade 1000 m of streams, then the total length of streams produced by mitigators must equal 2000 m.

Given that n mitigators enter the market and the trading ratio is t, each mitigator earns profit

$$\pi(n,t) = P(Q(n,t)\,/\,t,t)q(n,t) - C(q(n,t)) - F.$$

In a free-entry equilibrium, each firm earns zero profit. Hence no other firms want to enter the market, and no existing firms want to leave the market. The equilibrium number of mitigators, n^{eq}, is the number of firms such that profit $\pi(n,t)$ is indeed equal to zero. Given a fixed t, it can be found by solving

$$\pi(n,t) = 0 \tag{1}$$

for n. In other words, n^{eq} is the variable that closes the economic model. We assume the parameters are such that there is a unique solution for n^{eq}.

Economic value $E(n)$ is defined in the standard way as consumer surplus less production costs and fixed costs. We have

$$E(n,t) = \int_0^{Q(n,t)} P(s\,/\,t,t)ds - nC(q(n,t)) - nF.$$

Let n^E be the value of n that maximizes economic value $E(n,t)$. Now, n^{eq} describes the outcome that actually occurs in the market and n^E is the hypothetical outcome that maximizes economic value. These quantities form the basis of our first welfare analysis. We stress that this first analysis does not account for ecosystem function, we are simply establishing a baseline result for the case of a normal free-entry market in which the commodity traded has no ecological implications.

Mankiw and Whinston (1986) analyze the relationship between n^{eq} and n^E and show that $n^E \leq n^e$. That is, there is excess entry: the equilibrium number of mitigators is larger than the number of mitigators that maximizes economic value. This result is obtained by assuming that n is continuous. It is modified slightly if n is restricted to be an integer. For ease of exposition we assume in the theoretical model that n is continuous. In the application section, we consider only integer values of n. A detailed mathematical analysis of their result in given in Appendix A. Here we develop the intuition. The key assumption of Mankiw and Whinston's model is that an increase in the number of firms leads to an increase in total output, but a decrease in output per firm. When a mitigator decides whether to enter, it considers its own profit, but not the decreased output from the other firms. So more firms will enter the market than is desirable from the point of view of society.

We now turn to how this economic problem of excess entry is affected by ecological considerations. To do this, we must add an ecological model to Mankiw and Whinston's economic model. The ecological model captures the inefficiency of restored ecosystems relative to natural ecosystems as well as returns to scale. First we discuss returns to scale. We use a simple power function to describe the relationship between ecosystem size and function. The ecosystem function of natural ecosystem k is $\beta_k \ell^x$ where x is positive. A value for x greater than one indicates increasing returns to scale, a value for x equal to one indicates constant returns to scale, and a value for x less than one indicates decreasing returns to scale. For example, if $x<1$, then doubling the size of the ecosystem less than doubles the ecosystem function. The parameter β_k indicates the quality of natural ecosystem k. The total ecosystem function of all natural ecosystems lost to development is

$$\sum_{k=1}^{m} \beta_k \ell^x = m\beta\ell^x$$

where β is the average quality of the natural ecosystems.

Now consider the inefficiency of restored ecosystems. We assume that the ecosystem function of a restored ecosystem is less than or equal to the ecosystem function of the average natural ecosystem. So the ecosystem function of restored ecosystem i can be expressed as $\alpha_i \beta q^x$ where α_i is a number between zero and one that captures the efficiency loss of ecosystem i. The total gain of restored ecosystem function due to mitigators is

$$\sum_{i=1}^{n} \alpha_i \beta q^x = n\alpha\beta q^x \tag{2}$$

where α is the average efficiency loss. This formulation is deterministic and allows heterogeneous restored ecosystems; some restored streams are ecologically superior to others. There are several other ways, however, to interpret Eq. (2) and, in particular, the variable α. Consider a probabilistic model with homogeneous restored ecosystems. Suppose that a given restoration project either succeeds or not, and the probability of success is α. Then $n\alpha$ is the expected number of successful restorations. Yet another interpretation of α comes from consideration of the effectiveness of regulators at determining restoration quality. Under this interpretation, α is the conditional probability of a successful restoration, given regulatory approval. At present, the success criteria for evaluating restored streams seems quite relaxed (US Army Corps of Engineers, 2003). If this is the case, then an appropriate value for α may be significantly less than one.

To place the ecological model in the context of a market for ecosystems, we must specify net ecosystem function $B(n)$. This is defined as the total ecosystem function of all the restored ecosystems minus the total ecosystem function of all the degraded ecosystems. We have

$$B(n,t) = n\alpha\beta q^x - m\beta\ell^x. \tag{3}$$

Using the supply–demand relation gives

$$B(n,t) = \beta\left[n\alpha q(n,t)^x - m\left(\frac{Q(n,t)}{mt}\right)^x\right].$$

Thus both components of net ecosystem function depend on the number of mitigators in the market. Notice that if $B=0$, then trading has no net effect on ecosystem function in the trading area. This is, of course, the goal of NNLF regulation.

We define welfare as the sum of economic value and net ecosystem function (Ludwig et al., 2003):

$$W(n,t) = E(n,t) + B(n,t),$$

Subsequent welfare analyses will be based on W, rather than E, so that we account for the effects of both economic value and ecosystem function. For example, we will compare the number of mitigators that maximizes welfare with the number that results from a free-entry equilibrium, and the details of this comparison depend on whether regulation is via NNLF or NNLS.

3. NNLS regulation

The policy goals of NNLS regulation are relatively easy to obtain. The regulator simply requires that a developer purchase a quantity of ecosystems from the market that is equal to the quantity of ecosystems the developer damaged. In other words the trading ratio is equal to one. This insures that the net size of ecosystems remain constant (i.e., the total length of streams within a particular region). However, net ecosystem function does not remain constant because trading may lead to a different number of ecosystems with different individual sizes and trading reduces natural ecosystems and increases restored ecosystems. We account for these changes in the welfare analysis of NNLS regulation.

We have seen that, accounting only for E, there is excess entry ($n^{\mathrm{E}} \leq n^{\mathrm{eq}}$.). How does the addition of B change this? Once again we place the mathematical details in the appendix and simply present the intuition. With $t=1$, the welfare maximization problem is

$$max\ W(n,1) = E(n,1) + B(n,1).$$

Let n^{W} be the welfare maximizing number of mitigators. This value may differ from n^{E} because of the $B(n,1)$ term. So lets study it in detail.

The formula for B has two terms. The effect of the second term,

$$-m\left(\frac{Q(n,1)}{m}\right)^{x},$$

is unambiguous: it exacerbates the problem of excess entry. An increase in n leads to an increase in total output Q. The total size of ecosystem degraded by development increases, to the detriment of net ecosystem function. And of course a mitigator does not account for these effects when they decide whether to enter.

The effect of the first term in B,

$$n\alpha q(n,1)^{x},$$

is in fact ambiguous. As n increases, output per mitigator decreases (i.e., the size of individual restored ecosystems decreases). But ecosystem function of restored ecosystems may increase or decrease, depending critically on the returns to scale parameter x. If there are increasing returns to scale, (i.e., that one large restored ecosystem gives more function than two small ones), then function of restored ecosystems are likely to decrease as n increases. In this case, the first term in B exacerbates the problem of excess entry as well. Alternatively, when there are decreasing returns to scale, the function of restored ecosystems are likely to increase as n increases. This generates a benefit to society which at least partially ameliorates the problem of excess entry.

Putting these results together, then, leads us to conclude that for large values of x, the overall effect of $B(n,1)$ is to exacerbate the economic problem of excess entry, so that we have $n^{\mathrm{W}} < n^{\mathrm{E}} \leq n^{\mathrm{eq}}$. Mankiw and Whinston show that, in the absence of ecological considerations, there is excess entry. Thus, a key finding of our linked analysis is that if large restored ecosystems provide greater functions than small restored ecosystems (reflected in the returns to scale parameter x being large), then accounting for ecological considerations makes things worse. As discussed above, the provision of wildlife habitat is likely to be characterized by large values of x. In this case, we would expect that NNLS regulation leads to excess entry.

The results are not nearly as clear-cut for other values of x. Here the overall effect of B is ambiguous. If the benefit generated by the first term of B is large enough, then ecological considerations may overwhelm the economic problem of excess entry and we actually have insufficient entry. To understand conditions under which this can occur, observe that the fixed costs F and the number of developers m have opposite effects on the equilibrium number of firms. It is possible for both of these numbers to increase in such a way that the equilibrium number of firms is unchanged. Such an increase would, however, have an effect on the output levels of the mitigators, and therefore the welfare maximizing number of firms. One can easily construct examples in which insufficient entry occurs for large values of m and F; if the costs of entering in the market are too large (e.g., large technical requirements or legal fees), then there are not enough mitigators to provide restored streams to meet the requirements of the economic–ecologic system.

4. NNLF regulation

In contrast to NNLS, it is not straightforward to meet the policy goals of NNLF regulation. Intuitively, one might think that the regulator can simply select $t = 1/\alpha$, to correct for the fact that restored ecosystems give less function that natural ecosystems. This is only true, however, in the special case of constant returns to scale. More generally, the regulator must account for linkages between the economics of entry and ecosystem function. As the trading ratio changes, the demand for mitigation changes, and this leads to a change in the equilibrium number and size of restoration projects, which in turn leads to a change in net ecosystem function.

The regulator wants to select t such that

$$B(n,t) = 0. \tag{4}$$

Solving this yields

$$t = \left(\frac{1}{\alpha}\right)^{\frac{1}{x}}\left(\frac{n}{m}\right)^{\frac{x-1}{x}}. \tag{5}$$

If there are constant returns to scale ($x=1$) then the regulator should indeed select $t=(1/\alpha)$. That is, if restored ecosystems are half as ecologically functional as natural ecosystems and there are no effects of scale of restoration projects, then the trading ratio should be set at 2. But for other values of returns to scale parameter, the analysis is more complicated. From Eq. (1), we see that the equilibrium number of mitigators depends on the trading ratio. And from Eq. (4) we see that the trading ratio depends on the equilibrium number of mitigators. Thus we must solve this system of two simultaneous equations for n and t.

In summary, the solution to the system of Eqs. (1) and (4) describes the market equilibrium outcome for NNLF regulation. To obtain this outcome, the regulator must in essence anticipate the equilibrium behavior of the mitigators. In other words, the regulator solves Eqs. (1) and (4) for n and t and then announces this t as the trading ratio. Given this t, the mitigators make independent entry and production decisions, and in equilibrium the number of mitigators will turn out to be equal to n, and net ecological function will turn out to be zero. If the regulator does not properly account for the economics of market entry and the ecology of returns to scale, then the resulting equilibrium may not satisfy NNLF.

There is more than one way to conduct a welfare analysis of ecosystem markets with a trading ratio. First consider a method that is most consistent with Mankiw and Whinston and our own analysis of NNLS regulation. Mankiw and Whinston compare the equilibrium n that solves Eq. (1) to the n that maximizes economic value. For NNLS regulation, we compared the equilibrium n that solves Eq. (1) with the n that maximizes welfare, which accounted for economic value and ecosystem function. For NNLF regulation, the obvious parallel procedure is to determine the equilibrium n (and t) that solve Eqs. (1) and (4) with the n (and t) that solve

$$W^{B} = max\ [E(n,t) + B(n,t)] \text{ such that } B(n,t) = 0.$$

Denote the n that solves this problem as $n^{\mathrm{W}B}$. Also, we use the variable W^{B} to indicate the maximum value of welfare attained by accounting for the net-ecosystem-function constraint but ignoring the zero-profit equilibrium condition. We obtain a similar result as with NNLS regulation. The welfare maximizing number of mitigators may be greater or less than the equilibrium number of mitigators, as we show by example in the next section.

Alternatively, we might be interested in the total value obtained by society under various assumptions rather than the relation between the equilibrium and welfare maximizing number of mitigators. Let W^{U} be the unconstrained welfare maximum. We have

$$W^{\mathrm{U}} = max\ E(n,t) + B(n,t).$$

This represents the maximum welfare that can be generated by the economic-ecological system, without regard to the zero-profit equilibrium condition and the net-ecosystem-function constraint. Next define

$$W^{\pi} = max\,[E(n,t) + B(n,t)]\,\text{such that}\;\pi(n,t) = 0.$$

Here W^{π} corresponds to the maximum value of welfare attained by accounting for the zero-profit equilibrium condition but not the net-ecosystem-function constraint. Finally, let W^{eq} be the welfare generated by the n and t that solve Eqs. (1) and (4). A comparison of the quantities W^B, W^U, W^{π} and W^{eq} reveals the relative welfare loss associated with the zero-profit condition, the net-ecosystem-function constraint, or both. We perform such a comparison in the next section.

5. Application: stream trading in North Carolina

To apply our theory, we provide a more detailed description of the demand for ecosystem services and the behavior of the mitigators. The model developed in the previous section applies to a variety of market structures. Here we analyze the case of a classic Cournot oligopoly. For a fixed value of n, each mitigator selects output q given the output choices of their rivals. The equilibrium value for n is determined by the zero profit condition. Each developer has quadratic utility function for developing ecosystems $U(\ell) = a\ell - (b/2)\ell^2$ where a and b are positive constants (see Vives, 2002 for an example of this model.) The utility functions give rise to the market demand for ecosystems. In a Cournot oligopoly, the mitigators select output given the output choices of their rivals and the market demand. Let the marginal cost of restoring a unit size of ecosystem be equal to a constant $c>0$.

Given the trading ratio t and the price p for purchasing ecosystem credits, each developer maximizes net utility (utility minus the cost of obtaining the sufficient quantity of credits):

$$\max_{\ell}\; a\ell - (b/2)\ell^2 - p(t\ell).$$

The solution to this problem gives the individual demand curve $p = \frac{a}{t} - \frac{b}{t}\ell$ and the market demand curve is

$$P(L,t) = \frac{a}{t} - \frac{bL}{mt}. \qquad (6)$$

Using the relation $Q = tL$, we can express this demand in terms of the total output of the mitigators as $P(Q/t,t) = \frac{a}{t} - \frac{bQ}{t^2 m}$.

Given this demand, each mitigator selects q to maximize profits:

$$\max_{q}\; P(Q/t,t)q - cq.$$

The solution to this problem is $q(n,t) = \frac{mt(a-ct)}{b(n+1)}$. Thus we can write Eq. (1) as

$$(n+1)^2 = (a-ct)^2\frac{m}{bF}. \qquad (7)$$

Using Eqs. (7) and (5), we can determine the equilibrium number of mitigators and the trading ratio that gives this equilibrium and keeps net ecosystem function equal to zero.

We use this specific model for firm behavior and demand to analyze ecosystem trading in North Carolina. In particular, consider the market for streams mitigation credits. We calibrate the model by taking a snapshot of the current market. Based on conversations with private mitigators and state mitigation program regulators, we approximated the marginal cost c to be \$250 per linear foot of stream, the fixed costs F to be \$250,000, and the equilibrium price P to be \$350 per foot. By analyzing a database of actual trades (BenDor et al., 2008), we determined the number of developers m to be 150, the equilibrium quantity of linear ft of stream traded Q to be 40,000 ft, and the equilibrium number of mitigators n^{eq} to be 10. From this information, and assuming a trading ratio of one, we then determined the demand parameters a and b by solving the system of Eqs. (7) and (6). This gives $a = 887.651$ and $b = 2.016$. Finally, based on Holmes et al. (2004) we selected $\beta = \$89.5$ (β can be interpreted as the dollar value of a foot of average quality stream).

With these parameters in hand, we first consider NNLS regulation. The equilibrium number of mitigators n^{eq} is of course 10. The number of mitigators that maximizes economic value, n^E, is equal to 4. In Table 1 we show the welfare maximizing number of mitigators as a function of x and α. For values of $x>1$, ecological considerations exacerbate the economic problem of excess entry and so $n^W < n^E$. And it is always the case that the equilibrium number of mitigators is larger than the welfare maximizing number of mitigators. Although it is possible to have insufficient entry, this does not occur for the particular set of parameters that characterize stream trading in North Carolina.

For NNLF, both the equilibrium number of mitigators and the welfare maximizing number of mitigators n^{WB} are a function of x and α, as shown in Table 2. The results are in contrast to those found for NNLS. Here it is possible to have insufficient entry, in particular for $\alpha = 1$ and $x = 0.85$. In this case the equilibrium number of firms is 6 and the welfare maximizing number of firms is 11. Also shown in Table 2 are the welfare losses associated with the zero-profit condition (Loss π), the net-ecosystem-function constraint (Loss B), or both (Loss eq). Each welfare loss is calculated as a percentage loss from the unconstrained welfare maximum. For example, we have

$$\text{Loss eq} = \frac{W^U - W^{eq}}{W^U}.$$

For $\alpha = 1$ and $x = 0.85$, the welfare loss due to the net-ecosystem-function constraint (76%) is larger than the welfare loss due to the zero-profit condition (11%). This relationship holds for most of the other cases as well.

The results for welfare losses are interesting from a theoretical perspective. On a more practical level, our model can also be used to guide the choice of a trading ratio to satisfy NNLF. Table 3 shows the solution to Eq. (5) as a function of returns to scale x and efficiency of restored ecosystems α. In the row corresponding to $x = 1$, the trading ratio is simply the inverse of α. For other values of x, the trading ratio accounts for the dynamics of market entry through the effect of the equilibrium value for n in Eq. (5). For this application and range of parameters, as α decreases the trading ratio increases. Recall that one interpretation of α is the effectiveness of regulatory oversight. Table 3 shows how a decrease in effectiveness can be accounted for by increasing the required trading ratio. Also for this application and range of parameters, as x increases, the trading ratio decreases.

We conclude the application section with a sensitivity analysis of the six parameters for which we determined numeric values. As shown in Table 4, we used the Loss eq variable to conduct this analysis. For each parameter in turn, we analyzed the percent change in Loss eq that resulted from a 10% increase in the parameter, keeping

Table 1
NNLS: welfare maximizing number of mitigators ($n^{eq} = 10$).

x	n^W ($\alpha = 1$)	n^W ($\alpha = 0.5$)
0.4	4	4
0.6	4	4
0.8	4	4
1.0	4	3
1.2	3	2
1.4	2	2
1.6	2	1
1.8	1	1

Table 2
NNLF: equilibrium and welfare maximizing number of mitigators; welfare losses associated with equilibrium, zero-profit condition, and net-ecosystem-function constraint.

$\alpha=1$						$\alpha=0.5$				
x	n^{eq}	n^{WB}	Loss eq (%)	Loss π (%)	Loss B (%)	n^{eq}	n^{WB}	Loss eq (%)	Loss π (%)	Loss B (%)
0.85	6	11	77	11	76	[a]				
0.95	9	5	54	12	52	3	3	94	13	94
1.05	10	3	33	15	26	7	2	70	16	64
1.15	11	3	18	8	6	8	2	38	28	23
1.25	11	3	58	7	51	9	2	36	13	21
1.35	12	3	83	10	80	10	2	71	14	64
1.45	12	3	93	16	92	11	3	89	18	86
1.55	12	3	97	22	97	11	3	96	23	94

[a] An equilibrium does not exist for this parameter combination.

the other parameters fixed at their base value. The marginal cost of the production of streams c seems to be the most sensitive parameter.

6. Conclusions

Our analysis of ecosystem service markets links the economics of market entry with the ecological principles of returns to scale and restored ecosystem inefficiency. We derive two important insights. First, ecological considerations may exacerbate or ameliorate the economic problem of excess entry, regardless of whether a regulator pursues NNLS or NNLF. Second, to properly execute NNLF, a regulator must account for both the economic and ecological processes. Failure to do so may actually lead to a decrease in net ecosystem function. Regulators are currently wrestling with the attainment of NNLF by selecting various trading ratios. Our model offers theoretically grounded insights into the proper choice of trading ratios, accounting for market entry, efficiency of restored ecosystems, and returns to scale.

Our model could be extended in several ways. First, it would be interesting to allow developers and mitigators to have heterogeneous costs and demand functions. Second, although we considered heterogeneous ecosystems, we did not explore the possibility that a given ecosystem might interact with other ecosystems. An important extension of our model would be to incorporate a specific geo-physical mapping of stream function to account for these interactions. Third, the only economic instrument we considered was the trading ratio. Other possible instruments include a restriction in the number of mitigators in the market or a restriction on the size of each mitigation project. Perhaps the use of one of these other instruments to attain NNLF would lead to smaller welfare losses.

The choice of trading ratio is highly sensitive to the relationship between ecosystem function and size of the restoration project, reflected in our analysis by x. However, values of x depend on the specific ecosystem function of interest. Maximizing nutrient retention may be best accomplished by numerous small restored streams or wetlands located in the headwater regions of a watershed, while habitat for endangered species may necessitate single, larger restoration sites. These would be reflected in different values of x for different ecosystem functions, and raises the classic ecological conservation question of SLOSS, single large or several small (Schwartz, 1999). Our linked analysis follows current regulatory practice in using a single variable to capture ecosystem function. In practice, such a procedure will likely result in no-net-loss of some functions, but not others. This suggests the need to more accurately understand the relationship between size and success of restoration, and how to deal with managing multiple ecosystem functions through single bundled commodities like area or length.

Table 3
Values for the trading ratio such that NNLF is satisfied as a function of returns to scale and efficiency of restored ecosystems.

		α			
		0.4	0.6	0.8	1.0
x	0.9	[a]	2.85	1.83	1.38
	1.0	2.5	1.667	1.25	1
	1.2	1.32	0.97	0.77	0.65
	1.5	0.75	0.59	0.50	0.43

A value of $x=1$ implies constant returns to scale and a value of $\alpha=1$ implies perfect restoration efficiency.
[a] An equilibrium does not exist for this parameter combination.

Table 4
Sensitivity Analysis: Percent Change in Loss *eq* assuming 10% change in each parameter, *ceteris paribus*.

Parameter	$x=0.95$ (%)	$x=1.35$ (%)
f (fixed costs)	0.11	−0.15
m (number of developers)	−0.78	0.24
c (marginal cost production)	8.01	−2.70
a (demand parameter)	−10.32	0.66
b (demand parameter)	0.05	−0.82
β (value of average quality stream)	−1.32	1.76

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Appendix A

We now give a detailed mathematical derivation of Mankiw and Whinston's (1986) result and show how it is modified when we account for both economic value and net ecological function. Because this analysis holds for the case in which $t=1$ we suppress the explicit dependence on t and simply refer, for example, to $Q(n)$ rather than $Q(n,t)$.

Mankiw and Whinston show that $n^E \le n^e$. That is, there is excess entry: To obtain this result, Mankiw and Whinston make three assumptions about the economic model.

B1. Total output $Q(n)$ is increasing in n
B2. Output per mitigator $q(n)$ is decreasing in n
B3. The price $P(Q(n))$ is at least as large as marginal cost $C'(q(n))$.

Their argument proceeds in the following manner. The first-order condition for n^E can be written as

$$\frac{\partial E}{\partial n} = \pi(n) + n(P(Q(n)) - C')\frac{\partial q}{\partial n} = 0. \tag{A.1}$$

From assumptions B1–B3, it follows that the second term is non-positive and that $\pi(n)$ is decreasing in n. So, loosely speaking, $\frac{\partial E}{\partial n}$ lies below $\pi(n)$, and hence $n^E \le n^{eq}$. In a purely economic model, a free-entry equilibrium leads to excess entry.

Now, in our model, welfare depends on both E and B. It is helpful to express B as

$$B(n) = \beta\left(\underbrace{n\alpha q(n)^x}_{\phi} \underbrace{-m\left(\frac{Q(n)}{m}\right)^x}_{\theta}\right).$$

We have

$$\frac{\partial W}{\partial n} = \frac{\partial E}{\partial n} + \frac{\partial B}{\partial n} = \frac{\partial E}{\partial n} + \beta\frac{\partial \phi}{\partial n} + \beta\frac{\partial \theta}{\partial n} \tag{A.2}$$

The partial derivative of E is given in Eq. (A.1). We have

$$\frac{\partial \phi}{\partial n} = \alpha q(n)^{x-1}\left(q(n) + nx\frac{\partial q}{\partial n}\right) \tag{A.3}$$

and

$$\frac{\partial \theta}{\partial n} = -x\left(\frac{Q(n)}{m}\right)^{x-1}\frac{\partial Q}{\partial n} \tag{A.4}$$

From assumption B1, $\frac{\partial Q}{\partial n}$ is positive, so clearly $\frac{\partial \theta}{\partial n}$ is negative. From Eq. (A.3), we see that the sign of $\frac{\partial \phi}{\partial n}$ is determined by the sign of $q(n) + nx\frac{\partial q}{\partial n}$. From assumption B2, $\frac{\partial q}{\partial n}$ is negative, so $q(n) + nx\frac{\partial q}{\partial n}$ may be positive or negative. It will be negative if

$$x > \frac{-q(n)}{n\frac{\partial q}{\partial n}}. \tag{A.5}$$

In summary, when Eq. (A.5) holds, then $q(n) + nx\frac{\partial q}{\partial n}$ is negative, and hence $\frac{\partial \phi}{\partial n}$ is negative. For large values of x, the net function of restored ecosystems decreases as n increases.

Having specified the effect of n on net ecosystem function, we now compare the welfare-maximizing number of mitigators n^W with the equilibrium number of mitigators n^{eq}. The first-order condition for n^W is

$$\frac{\partial W}{\partial n} = \frac{\partial E}{\partial n} + \frac{\partial B}{\partial n} = 0. \tag{A.6}$$

From Eqs. (A.1) and (A.2) we have

$$\frac{\partial W}{\partial n} = \pi(n) + n(P(Q(n)) - C')\frac{\partial q}{\partial n} + \beta\frac{\partial \phi}{\partial n} + \beta\frac{\partial \theta}{\partial n} = 0. \tag{A.7}$$

Recall that n^{eq} is found by solving $\pi(n) = 0$ and n^E is found by solving $\pi(n) + n(P(Q(n)) - C')\frac{\partial q}{\partial n} = 0$. These expressions are components of Eq. (A.7), suggesting it may be possible to analyze the relationship between all three variables n^{eq}, n^E and n^W. Toward this end, consider another assumption:

B4. Marginal economic value $\partial E / \partial n = \pi(n) + n(P(Q(n)) - C')\frac{\partial q}{\partial n}$ is decreasing in n.

If this assumption holds, then we can conveniently characterize the effect that ecological considerations have on the economic problem of excess entry. It is not necessary, however, to make this assumption if we simply want to compare n^{eq} to n^W. It is straightforward to adjust our arguments to compare n^{eq} to n^W directly for cases in which assumption B4 is not satisfied.

For large values of x there is excess entry: the equilibrium number of mitigators is greater than the welfare maximizing number of mitigators. To see why this occurs, suppose that x satisfies Eq. (A.5). As discussed above, this implies $\frac{\partial \phi}{\partial n}$ is negative. Because $\frac{\partial \theta}{\partial n}$ is always negative, it follows that $\frac{\partial B}{\partial n}$ is negative. From Eq. (A.6) we see that $\frac{\partial W}{\partial n}$ lies below $\frac{\partial E}{\partial n}$. By assumption B4, $\frac{\partial E}{\partial n}$ is decreasing in n. Hence $n^W < n^E \leq n^e$.

If Eq. (A.5) is not satisfied, then $\frac{\partial \phi}{\partial n}$ is positive, $\frac{\partial B}{\partial n}$ may be positive, and it may be the case that $n^W > n^E$. If this does indeed occur, then ecological considerations, at least to some extent, ameliorate the economic problem of excess entry. Furthermore, if $\frac{\partial \phi}{\partial n}$ is large enough, then it may even be the case that $n^W > n^{eq}$ which implies insufficient entry.

References

Armsworth, P., Daily, G., Kareiva, P., Sanchirico, J., 2006. Land market feedbacks can undermine biodiversity conservation. Proceedings of the National Academy of Sciences 103, 5403–5408.

BenDor, T., Doyle, M.W., Sholtes, J., 2008. Spatial characteristics of an emergent stream mitigation banking program: economic and ecological implications. Paper presented at the Annual Meeting of the Society of Wetland Scientists.

Bernhardt, E.S., Palmer, M.A., Allan, J.D., National River Restoration Science Synthesis Working Group, 2005. Restoration of US rivers: a national synthesis. Science 208, 636–637.

Bonnie, R., 1999. Endangered species mitigation banking: promoting recovery through habitat conservation planning under the Endangered Species Act. Science of the Total Environment 240, 11–19.

Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standarized environmental accounting units. Ecological Economics 63, 616–626.

Brock, W.A., Carpenter, S.R., 2007. Panaceas and diversification of environmental policy. Proceedings of the National Academy of Sciences 104, 15206–15211.

Brown, P.H., Lant, C.L., 1999. The effect of wetland mitigation banking on the achievement of no-net-loss. Environmental Management 23, 333–345.

Burtraw, D., Evans, D.A., Krupnick, A., Palmer, K., Toth, R., 2005. Economics of pollution trading for SO_2 and NO_x. Annual Review of Environment and Resources 30, 253–289.

Capoor, K., Ambrosi, P., 2007. State and trends of the carbon market 2007. World Bank, Washington, DC.

Carleton, J.N., Grizzard, T.J., Godrej, A.N., Post, H.E., 2001. Factors affecting the performance of stormwater treatment wetlands. Water Research 35, 1552–1562.

Cedfeldt, P.T., Watzin, M.C., Richardson, B.D., 2000. Using GIS to identify functionally significant wetlands in the Northeastern United States. Environmental Management 26, 13–24.

Chong, H., Sunding, D., 2006. Water markets and trading. Annual Review of Environment and Resources 31, 239–264.

Collins, A., Rosenberger, R., Fletcher, J., 2005. The economic value of stream restoration. Water Resources Research 41, W02017. doi: 10.1029/2004wr003353.

Doyle, M.W., Stanley, E.H., Harbor, J.M., 2003. Hydrogeomorphic controls on phosphorus retention in streams. Water Resources Research 36, 1147. doi: 10.1029/2003WR002038.

ELI [Environmental Law Institute], 2007. Mitigation of impacts to fish and wildlife habitat: estimating costs and identifying opportunities. ELI, Washington DC.

Fernandez, L., Karp, L., 1998. Restoring wetlands through wetland mitigation banks. Environmental and Resource Economics 12, 323–344.

Hallwood, P., 2007. Contractual difficulties in environmental management: the case of wetland mitigation banking. Ecological Economics 63, 446–451.

Holmes, T., Bergstrom, J., Huszar, E., Kask, S., Orr, F., 2004. Contigent valuation, net marginal benefits, and the scale of riparian ecosystem restoration. Ecological Economics 49, 19–30.

Horan, R., Shortle, J., 2005. When two wrongs make a right: second-best point-nonpoint trading ratios. American Journal of Agricultural Economics 87, 340–352.

Kuman, M., Kumar, P., 2008. Valuation of the ecosystem services: a psycho-cultural perspective. Ecological Economics 64, 808–819.

Lave, R., Robertson, M.M., Doyle, M.W., 2008. Why you should pay attention to stream mitigation banking. Ecological Restoration 26, 287–289.

Ludwig, D., Carpenter, S., Brock, W., 2003. Optimal phosphorus loading for a potentially eutrophic lake. Ecological Applications 13, 1135–1152.

Mankiw, N.G., Whinston, M., 1986. Free entry and social inefficiency. Rand Journal of Economics 17, 48–58.

MOA, 1990. Memorandum of Agreement Between the Environmental Protection Agency and the Department of the Army Concerning the Determination of Mitigation Under the Clean Water Act Section 404(b)(1) Guidelines, 6 February 1990.

NRC, 2001. Compensating for Weland losses under the Clean Water Act. National Academy Press, Washington, DC.

Robertson, M.M., 2006. Emerging ecosystem service markets: trends in a decade of entrepreneurial wetland banking. Frontiers in Ecology and the Environment 4, 297–302.

Schwartz, M.W., 1999. Choosing the appropriate scale of reserves for conservation. Annual Review of Ecology and Systematics 20, 83–108.

US Army Corps of Engineers, Wilmington District, 2003. North Carolina stream mitigation guidance, Wilmington, NC.

Vaughn, C.C., Taylor, C.M., 1999. Impoundments and the decline of freshwater mussels: a case study of an extinction gradient. Conservation Biology 13, 912–920.

Vives, X., 2002. Private information, strategic behavior, and efficiency in Cournot markets. Rand Journal of Economics 33, 361–376.

Wohl, E., Angermeier, P.L., Bledsoe, B., Kondolf, G.M., MacDonnell, L., Merritt, D.M., Palmer, M.A., Poff, N.L., Tarboton, D., 2005. River restoration. Water Resources Research 41, W10301. doi: 10.1029/2005WR003985.

Woodward, R.T., Kaiser, R.A., Wicks, A.-M.B., 2002. The structure and practice of water quality trading markets. Journal of the American Water Resources Association 28, 967–979.

Yang, W., Change, J., Xu, B., Peng, C., Ge, Y., 2008. Ecosystem service value assessment for constructed wetlands: a case study in Hangzhou, China. Ecological Economics 68, 116–125.