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Ecological issues related to wetland preservation, restoration, creation and assessment

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Abstract

A wide range of local, state, federal, and private programs are available to support the national (USA) policy of wetland 'No Net Loss'. Implementation of programs, however, has resulted in the continued loss of natural wetlands on the premise that restored or created wetlands will replace the functions and values lost by destruction of natural wetlands. What are the ecological implications and consequences of these programs from a biodiversity and ecosystem perspective? From a biodiversity perspective, ongoing wetland protection policies may not be working because restored or created wetlands are often very different from natural wetlands. Wetland protection policies may also be inadequate to preserve and restore ecological processes such as nutrient cycling because they mostly focus on individual wetlands and ignore the fact that wetlands are integral parts of landscapes. Wetland mitigation projects, for example, often result in the exchange of one type of wetland for another and result in a loss of wetland functions at the landscape level. The most striking weakness in the current national wetlands policy is the lack of protection for 'dry-end' wetlands that are often the focus of debate for what is and what is not a wetland. From an ecological perspective, dry-end wetlands such as isolated seasonal wetlands and riparian wetlands associated with first order streams may be the most important landscape elements. They often support a high biodiversity and they are impacted by human activities more than other types of wetlands. The failings of current wetland protection and mitigation policies are also due, in part, to the lack of ecologically sound wetland assessment methods for guiding decision making processes. The ecologically based Hydrogeomorphic (HGM) approach to wetland assessment has the potential to be an effective tool in managing biodiversity and wetland ecosystem function in support of the national 'No Net Loss' policy. Published by Elsevier Science B.V.

Keywords: Wetlands; Biodiversity; Ecosystem function; 'No Net Loss'; Wetland policy; Restoration; Creation; Mitigation; Assessment; Hydrogeomorphic; HGM; Landscape

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1. Introduction

Most historical wetland losses that have occurred in the US resulted directly or indirectly from programs supported by government policy. Only in recent decades have policy debates and administrative programs resulted in a significant decrease in the rate of wetland loss (Dahl et al., 1991; Dahl and Allord, 1996; Opheim, 1997; Tzoumis 1998). In some states (e.g. Ohio and California), protection and restoration efforts came only after most wetlands were lost (e.g. Sibbing, 1997). In other states (e.g. Florida), historical policies related to water and wetland management resulted not only in considerable wetland losses, but, in threats to entire landscapes. In south Florida, for example, the Everglades and other hydrologically linked ecosystems are threatened primarily because of historical policies that allowed widespread wetland conversion and alteration of local and regional hydrologic regimes (Gunderson et al., 1995; Harwell, 1997). Currently, most types of wetlands are protected through a variety of federal and state programs (Vottler and Muir, 1996) and some programs (e.g. Wetlands Preserve Program) provide funding to support wetland protection and restoration.

We are currently in the era of 'No Net Loss'. Some argue that we have achieved the goal of 'No Net Loss' of wetlands because the rate of wetland loss has slowed and because many wetlands are being restored or created (e.g. Tolman, 1997). Others argue that, while wetland losses have slowed, wetlands losses continue at an unacceptable rate (Heimlich et al., 1997). The debate related to 'No Net Loss' is most often a numbers game and what is usually lost in the debate is the fact that wetlands continue to be lost or degraded and wetlands designed to replace them (e.g. restored, enhanced, or created wetlands) often have lower biodiversity and do not function similarly to the natural wetlands (Kentula, 1996; Street, 1998). Many wetlands, and the functions that they perform, are gone forever because a large number of wetland losses are never reported and restoration projects are often never completed or are failures (Smith, 1997). The scenario that emerges is one in which the US continues to lose natural wetlands without seriously considering the impacts of the losses under the guise of the 'No Net Loss' policy. One consequence of continued wetland losses, as described below, is that they will have a negative impact on wetland associated species.

Wetland loss can be viewed in several ways. The most obvious type of wetland loss is the conversion of a jurisdictional wetland to a non-jurisdictional status. Some areas may be classified as wetlands based on biota, soils, and hydrology but are not considered to be jurisdictional because they are specifically excluded by state or federal regulations. Conversion of these types of wetlands also represents a wetland loss. Finally, it could be argued that conversion of one type of wetland to another represents a net loss of wetlands. The point that needs to be made, independent of how one defines wetland loss, is that wetland losses continue for many reasons. In Minnesota adequate wetland legislation does not protect wetland resources because political decisions resulted in regulatory structures that 'lead to a perpetuation of degraded wetland and upland habitat that is unrecognized but so prevalent on the landscape' (Svoboda, 1998). Some wetlands are not legally protected. Programs such as the US Army Corps of Engineers (COE) Nationwide Permit 26 (Albrecht and Connolly, 1998; Sibbing, 1998) allow wetlands, especially intermittently flooded headwater and isolated wetlands, to be converted to non-wetland habitats without any mitigation (Caputo, 1997). The COE has proposed to replace many of the nationwide permits with new permits, which would almost certainly allow for the continued loss of headwater and isolated wetlands. The proposed COE changes in nationwide permits were proposed even though a National Academy of Sciences study (National Research Council, 1995) concluded that 'the scientific basis for special permitting of wetlands in headwaters or isolated wetlands is weak'.

Undoubtedly debates will continue over issues such as: what is a wetland, how do you delineate the boundaries of a wetland, and how effective is the national 'No Net Loss' policy? While these debates continue I contend that because of the politically contentious nature of wetlands that 'No Net Loss' will remain a stated national goal but we will continue to lose natural wetlands. I also believe that wetland losses will continue to occur, this is based on the assumptions that destroyed wetlands can be replaced by restored, enhanced, and created wetlands.

What are the potential impacts of a continuation of the present wetland policy? I submit that wetland losses and associated declines in biodiversity and ecosystem function will continue for two primary reasons. First, the continued destruction of wetlands, particularly dry-end wetlands, will result in a loss of biodiversity and a reduced ability of wetlands to provide important landscape values such as water quality improvement. Second, the continued failure of the majority of wetland restoration efforts will result in a continued loss of wetland functions which directly or indirectly influence biodiversity. The purpose of this paper is to discuss four related topics that influence biodiversity and ecosystem function. First, I consider the issue of conservation of wetland-dependent species through efforts to preserve wetlands. Second, I consider the ecological consequences of the continued destruction of dry-end wetlands. Third, I ask whether or not there is any scientific justification to support the conclusion that restored and created wetlands function similarly to natural wetlands from the perspective of biodiversity and nutrient cycling? Fourth, I describe the Hydrogeomorphic (HGM) approach to functional assessment to develop the argument that most wetland mitigation will continue to fail unless they are based on sound ecological principles that include the use of reference wetland systems.

2. Wetland preservation

More than 50% of the nation's original wetland area has been lost and while the rate of wetland loss has declined, we continue to lose most types of wetlands (Opheim, 1997). Will the continued loss of wetlands have a negative impact on biodiversity? Boylan and Maclean (1997) have estimated that 46% of the nation's endangered species are wetland-dependent suggesting that the continued loss or degradation of wetlands will likely have a negative impact on the nation's biodiversity. One possible way to avoid the continued loss of wetland-dependent species is to conserve more wetlands. Can wetland preservation be used as a management tool at a national scale? I believe that wetland preservation will always be a desirable goal but that it is not likely to be a viable approach for conserving wetland-dependent species at a national scale. Noss and Peters (1995) identified areas of the country that contain ecosystems which contain federally threatened species. Most of the ecosystems identified by Noss and Peters occur over extensive geographic areas and it seems highly unlikely that it will be possible to preserve large areas in any of them (e.g. the entire state of Florida, riparian forests and wetlands throughout California, wetlands distributed across several Mid-western states). Noss and Peters found, for example, that many federally threatened species are associated with the nation's large rivers and streams. Large rivers and streams are all involved in interstate commerce and there is no possibility that they can be preserved or would we want them to be preserved in their current condition. Restoration of large rivers and streams is a more important issue than preservation (e.g. Sparks et al., 1998). Even if large areas associated with rivers could be preserved, it is unlikely that treats to wetland-dependent species could be eliminated because most landscapes associated with large rivers have been highly fragmented (e.g. Gosselink et al., 1990).

Even where large areas of an ecosystem type have already been preserved in national parks (e.g. Everglades National Park) or preserved by a variety of conservation and planning mechanisms (e.g. Pine Barrens of New Jersey) many wetland-dependent species remain threatened because wetlands are impacted by indirect human activities such as changes in water quality, invasion of alien species (Flack and Benton, 1998), and physical alterations of watersheds which supply surface and groundwater to wetlands (Zampela and Bunnell, 1998). I conclude that wetland preservation should be viewed as only one part of our national effort to conserve wetland biodiversity. Conservation of wetland-dependent species will be most successful when the species in question have small geographic ranges or their life history needs can be met by preservation of specific areas.

3. Dry-end wetlands

Wetlands occur over a wide range of hydrologic conditions and many wetlands occur in areas where flooding is intermittent or where the substrate may be saturated to the surface for a relatively short period of time. These types of wetlands are often referred to as 'dry-end' wetlands. Many dry-end wetlands occur in headwater positions of drainage systems while others are distributed across landscapes, often as isolated depressional wetlands. Examples of isolated dry-end wetlands are vernal pools and playas in the arid west and southwest, potholes in the Prairie Pothole region, seasonally flooded depressional wetlands (e.g. Carolina Bays) and seasonally wet flats in the southeast. Many dry-end wetlands are isolated or are maintained by fluctuating groundwater and are not part of or adjacent to the surface tributary system of waters of the US.

Dry-end wetlands are often not protected because they can fall under the umbrella of the COE Nationwide Permit 26 (Caputo, 1997) and they will likely remain unprotected by any permit that replaces the Nationwide Permit 26. Final decisions have not been made on the specific nature of future COE nationwide permits but proposals, to date, appear to be based more on streamlining the regulatory bureaucracy rather than on scientific knowledge. The argument is also made that dry-end wetlands are not important because they are typically small and perform few important ecological functions. There is no ecological basis for these conclusions (National Research Council, 1995) and they may be more valuable than other types of wetlands because of important landscape and biodiversity functions that they perform. Several examples of the importance of dry-end wetlands serve to support these contentions.

Small isolated wetlands are often important from the perspective of landscape hydrology and biodiversity. In the Prairie Pothole region, seasonal wetlands are important sites for feeding and breeding success of migratory waterfowl (Weller, 1998; Kantrud et al., 1989). Vernal pools in California play an important role in the maintenance of regional biodiversity (Zedler, 1987).

Dry-end wetlands associated with first order streams are the most abundant types of wetlands in watersheds and they are the first wetlands in the landscape to intercept runoff of surface and groundwater from source areas such as agricultural fields (Holland et al., 1990; Lowrance et al., 1995). Several studies have demonstrated that riparian forests (including wetland and non-wetland habitats) play an important role in removing sediments in nutrients from agricultural runoff (Peterjohn and Correll, 1984; Jacobs and Gilliam, 1985; Cooper, 1990). Boundaries between uplands and depressional wetlands have also been shown to play an important role in the maintenance of biodiversity of plant and animal species (Kirkman et al., 1998). Connectivity between habitats is also an important component of biodiversity maintenance in landscapes and fragmentation of landscapes has been shown to result in the decline of even common species (e.g. Jules, 1998). The available data, while still scarce for most types of wetlands and landscapes, demonstrates that every effort should be made to retain and restore headwater and isolated wetlands rather than allow their continued conversion to non-wetland habitats.

4. Wetland restoration and creation

Wetland restoration and creation are growing fields of scientific endeavor and societies (e.g. Society for Ecological Restoration), journals (e.g. *Restoration Ecology*, *Wetlands*, *Wetland Journal*), books (e.g. Kusler and Kentula, 1990; Hammer, 1992; Kentula et al., 1992; Kadlec and Knight, 1996) and annual meetings (e.g. Society of Wetland Scientists, Hillsborough Community College Ecosystem Restoration and Creation meeting) are devoted partially or completely on the topic of wetland restoration. While the science of wetland restoration and creation is young, one can ask whether or not enough information is known about wetlands to restore or create them with a high degree of success. Success in the context of wetland restoration and creation is a relative term and it depends, in part, on the goals of each restoration project. If the purpose is to create wetland habitat without regard to any particular objective other than to provide more wetland area, the answer is yes as long as hydrologic conditions are present to support wetland biota. The answer is also yes if the objectives are fairly broad in scope and if the hydrologic conditions are appropriate. The Olentangy River Wetland Restoration Research Park on the campus of the Ohio State University is a good example of a successful wetland creation project with broad objectives (Mitsch, 1997). The project objectives were to: (1) create riverine wetland habitat for purposes of conducting research on ecological processes in constructed and natural wetlands; (2) conduct research for the development of proper design criteria for wetland restoration and creation efforts; (3) develop criteria for measuring the success of wetland restoration and creation efforts; (4) teach wetland ecology to a wide range of audiences; and (5) demonstrate wetland creation and restoration techniques. If the restoration goals are directed toward emplacement of specific wetland biodiversity and function(s) at mitigation sites, success is possible (e.g. Marcus, 1998) but failures far outnumber successes (e.g. Street, 1998).

Zedler (1997) found that many restoration and construction projects are not successful or are only partially successful because of failures to recognize that wetlands are parts of larger landscapes. She demonstrated that failed attempts at restoration and creation were often due to design parameters that excluded important features of natural wetlands. Scatolini and Zedler (1996) found that invertebrate species assemblages were different in constructed coastal wetlands compared to natural reference wetlands. They suggested that the differences were mostly due to the fact that the substrates in the constructed wetland were coarser and had lower organic matter, which resulted in a lower rate of colonization by plants. Zedler provided four principles that should be used to guide coastal wetland restoration efforts:

1. Large systems support and maintain the highest biodiversity. Thus, restoration work with large systems should have greater potential for sustaining regional biodiversity;
2. Coastal wetlands support greater biodiversity if there are good linkages with adjacent ecosystems and few barriers to water flow and movement of animals. Restoration projects should remove barriers and improve connectivity;
3. Specific ecosystem types will develop best if located near or adjacent to an existing ecosystem of the same type; and
4. Small habitat remnants are likely to have reduced resilience and less resistance to natural and man-made perturbations.

Restoration of non-tidal habitats may be more problematic because it is harder to restore hydrologic conditions, particularly when wetlands are created or restored in topographic locations which are different than those in which the natural wetland occurred. After 3 years, restored wetlands in Iowa had fewer species than natural wetlands in all zones but the submersed zone (Galatowitsch and van der Valk, 1996). They also found that many species guilds were different between natural and restored wetlands and concluded that the seed bank is an important source of colonizers and that fewer species occurred in the seed bank of restored wetlands. Brown and Bedford (1997) and Stauffer and Brooks (1997) had suggested that restoration of soil conditions is important in restoration efforts in non-tidal wetlands. They found that vegetation recovery occurred faster when restored wetlands received salvaged wetland soils that contained propagules of wetland species compared to sites, which had no soil remediation. Ashworth (1997) found similar results in a Wisconsin study. She used a multivariate approach to evaluate the success of a restoration and found that, while revegetation occurred in the restored wetlands, that the dominant genus in the restored site was *Salix* while *Carex aquatilis* and *Calamagrostis canadensis* were the dominant vascular plant species in the natural reference site. Ashworth concluded that vegetation dissimilarities between restored and natural sites were due to differences in soil depth and hydrology.

The literature now contains several studies in which characteristics of restored wetlands have been compared to natural wetlands that are often referred to as reference wetlands (e.g. Keddy et al., 1993; Scatolini and Zedler, 1996; Streever et al., 1996; Brown and Bedford, 1997). The use of reference wetlands in restoration efforts is highly desirable for at least three reasons. First, information from reference wetlands can be used in planning restoration efforts to decrease the likelihood of failure and partial successes. Second, data from natural reference wetlands can be compared to restored wetlands to determine the degree to which restoration efforts are successful in restoring biodiversity and ecosystem functioning. Third, when restorations have failed or have been only partially successful, information from natural reference wetlands can be used to guide efforts to make the restoration successful. These and other uses of reference information are an integral component of the ecologically based Hydrogeomorphic (HGM) approach to wetland functional assessment.

5. The Hydrogeomorphic (HGM) approach to wetland functional assessment

In this section I provide a brief overview of the Hydrogeomorphic (HGM) approach to wetland assessment that is currently being developed in the USA. HGM is different from previous assessment systems because it is based on ecological principles and on the use of reference wetland systems. The use of reference wetland systems is especially relevant to the current discussion because they provide the information needed to make ecologically based decisions related to wetland conservation, restoration, and creation. Brinson (in press) provides a useful summary of the rationale and uses of reference wetlands.

The HGM approach (Smith et al., 1995) is based on the premise that wetland can be categorized into distinct classes (Brinson, 1993) and that characteristics of the classes can be used to describe and quantify specific wetland functions such as maintenance of groundwater recharge, nutrient cycling, and biodiversity. The six classes of wetlands that are used to develop HGM models are riverine, depressional, slope, flats, estuarine fringe, and lacustrine fringe. Wetland functions for each class are based primarily on position of a wetland in the landscape, the dominant source of water to the wetland, and the direction of water movement within the wetland.

HGM is still under development but several publications are available which describe and evaluate the strengths and weaknesses of the approach (Brinson et al., 1994, 1995, 1997, 1998; Brinson, 1995, 1996; Brinson and Rheinhardt, 1996; Hruby, 1997, 1998; Rheinhardt et al., 1997). In the context of the present paper, HGM is relevant because the methodology that underpins the approach can be used to provide solutions to many of the problems and issues related to wetland biodiversity and ecosystem functioning that I have described.

As an example, I return to problems associated with dry-end wetlands. I provided evidence to demonstrate that dry-end wetlands are important and that efforts should be made to change the regulatory environment that permits their continued destruction. While there is a clear need for policy change and for additional research on the ecology of dry-end wetlands, it is unlikely that either will occur quickly. Decisions will continue to be made on the fate of dry-end wetlands without a complete understanding of their ecology. The HGM approach provides input into the decision making process by offering quantitative information on the ecological status of wetlands. The approach, thus, provides the opportunity to make decisions that are based on ecologically sound principles. In the Prairie Pothole Region, for example, teams of experts working on a regional HGM guidebook identified 11 ecological functions for temporary and seasonal dry-end wetlands: (1) maintain static surface water storage; (2) maintain dynamic surface water storage; (3) retain particulates; (4) maintain elemental cycling; (5) remove imported elements and compounds; (6) maintain characteristic plant community; (7) maintain habitat structure within the wetland; (8) maintain food webs within the wetland; (9) maintain habitat interspersion and connectivity among wetlands; (10) maintain taxa richness of invertebrates; and (11) maintain distribution and abundance of invertebrates.

To quantify the 11 functions, a team of wetland experts developed a list of variables that could be used to assess ecological conditions at any wetland assessment site. Short descriptions of the 15 variables follow and examination of them demonstrates that most can be easily measured or observed in the field. Other variables can be quantified by collecting data that are readily available on aerial photographs. The list of variables also demonstrates that the team of experts decided that it was important to assess each wetland in a landscape context. The 15 variables fall into three categories.

5.1. Variables related to conditions within an individual wetland

(a) Presence of litter in several stages of decomposition; (b) presence or absence of a natural or constructed outlet; (c) abundance of woody and herbaceous plants in all vegetation zones; (d) physical integrity of the soil; (e) ratio of native to non-native plant species; and (f) dominant land-use and condition in the wetland.

5.2. Variables related to relationships between a wetland and surrounding upland areas

(a) Condition of the grassland buffer surrounding the wetlands. (b) Continuity of the grassland buffer surrounding the wetland. (c) Width of the grassland buffer surrounding the outermost wetland edge. (d) Extent of sediment delivery to wetland from anthropogenic sources including agriculture. (e) Area surrounding the wetland that defines the watershed of the wetland. (f) Presence of a constructed subsurface or surface outlet outside the outermost vegetated zone of the wetlands. (g) Dominant land-use of the upland watershed.

5.3. Variables relating a wetland to surrounding wetlands in the landscape

(a) Ratio of total area of temporary and seasonal wetlands to the total area of semi-permanent and permanent wetlands within a 1-mile radius of the wetland. (b) Absolute density of wetlands within a 1-mile radius of the wetland.

The scaling (e.g. quantification) of the variables listed above is done by collecting information from a series of wetlands (Reference Wetlands) that represent the range of conditions found in nature for a given class of wetlands within a defined geographic area. The collection of material from the reference wetlands includes information that can be used to develop descriptive profiles of the wetland subclass and data that are used to develop and scale HGM models. The entire set of information collected from reference wetlands is called a Reference Wetland System.

The reference concept is a cornerstone in the HGM approach but reference sites have been used successfully in other assessment approaches. The Index of Biological Integrity (IBI) approach to stream assessment is based on conditions in relatively unaltered or pristine reference sites. IBI has been used to assess the ecological health of fish (Karr, 1981; Angermeier and Karr, 1986) and invertebrate communities in stream ecosystems (Karr and Kerans, 1992; Kerans and Karr, 1994). Zampela and Bunnell (1998) have used components of IBI and reference in the HGM context to compare fish assemblages in degraded streams in the New Jersey Pinelands. The term ‘reference’ in the context of HGM is used as a basis for comparison. The common property of wetland reference systems, defined as a group of two or more wetlands of the same class or subclass, is that they represent examples of a particular type or complex of wetlands independent of their ecological conditions (Brinson and Rheinhardt, 1996).

As indicated above, in HGM, assessments of wetland sites are based on application of models of wetland functions (Rheinhardt et al., 1997). HGM models are developed from information compiled from a range of sites in a Reference Wetland System that are chosen to represent the range of conditions found for a particular class or subclass of wetlands in a defined geographic area. At the two ends of a continuum within a Reference Wetland System used to development of HGM models, are sites that are pristine or relatively undisturbed and sites that are highly altered or degraded.

One of the strengths of wetland reference systems is that they have many potential uses beyond development and application of HGM models. Brinson (in press) had identified four categories of uses: mitigation, education, research, and preservation. There are two sets of reasons or justifications for the use of reference wetland systems in HGM. First, data from reference systems allow everyone to use the same framework of comparison for assessment and the data can be standardized and scaled for better resolution and comparison between sites. Development of reference systems for HGM models also allows for better efficiency in time for applying the assessment models, and greater consistency in application of the assessment methodology. Second, the application of federal, state, county or municipal regulatory guidelines for assessment almost always require comparisons of existing conditions at a wetland site with conditions that will result from a proposed action. This type of comparison can not be adequately made unless it is possible to compare the proposed project site in its existing condition with similar types of wetlands that are relatively undisturbed and sites which represent a wide range of conditions, including those activities that are proposed for the project site. In addition, regulation of activities in wetlands routinely calls for avoidance and/or minimization of impacts, for restoration design, and for monitoring or contingency activities that document compliance. Each of these steps in the regulatory process require users to develop, articulate, and defend reasonable conclusions concerning proposed activities that can be referenced directly to wetlands on real property (e.g. reference wetlands). Furthermore, users of assessment methods need to be able to rapidly distinguish between existing conditions in the wetland in question and the impacts to ecosystem functioning that result from the proposed project. Comparisons of this sort, to be effective and accurate, require information from and the possibility of visiting specific wetland sites within a reference system.

Reference wetland systems can be used for training. Many wetland regulators are burdened with large numbers of applications and wetlands (e.g. dry-end wetlands and wetlands smaller than some critical area necessary to trigger wetland assessment procedures) are not even visited prior to issuance of permits. The availability of locally described reference wetlands, in conjunction with HGM models, would allow regulators to visit actual wetland sites that span the range of ecological conditions that might be encountered. At minimum, the availability of reference wetlands would allow new untrained regulators to become familiar with the resources that they are in charge of protecting.

6. Conclusions

Current wetland policy in the US will result in the continued loss of wetlands. From a landscape and biodiversity perspective, the focus of wetland regulation needs to be changed to provide greater protection for dry-end wetlands. Dry-end wetlands need to be protected to a greater degree because they provide important ecological functions, including the maintenance of biodiversity, at the level of individual wetlands and, most importantly, at the landscape level. It is difficult to argue for greater protection of dry-end wetlands without placing them in an appropriate ecological context. The development of reference wetlands systems offers the potential to provide the necessary context to support conservation of biodiversity and ecosystem functioning. Reference wetlands provide many opportunities beyond application in HGM models. They can be used in a regulatory context as well as for training and education. Currently, many decisions about the fate of wetlands, particularly dry-end wetlands, are made in the absence of adequate ecological knowledge. The use of reference wetland systems offers the potential for an ecologically sound approach to wetland management.

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